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PROCEEDINGS

RAMIRAN 2015

16th International Conference
Rural-Urban Symbiosis

RAMIRAN 2015 – 16th International Conference
Rural-Urban Symbiosis

Proceedings book

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Hamburg University of Technology, Germany

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Preface

Residues originate in rural and in urban systems. Many contain biogenic ingredients, e.g. food and green waste, livestock manure, feces, sewage and other sludge's, harvest residues, digestates, agro-industry residues, and more. Commonly residues are known to pollute environments, cause hygienic problems or they are cost-intensive to collect and treat. Since fossil raw materials are becoming scarce, biogenic residues are more and more a topic of utilization. Application options are manifold. They range from energetic utilization, e.g. to provide electicity and heat for households, industries and agricultural facilities, up to substantial applications ranging from bulk products such as composts and specific products like mineral fertilizers or biochemicals. A special challenge is the interface between rural and urban systems. In this context the mutual understanding is often limited, usually due to a lack of knowledge or contradicting interests.

The proceedings book provides information on actual research and on innovations on these sectors.
The book contains 4-page papers of oral and poster contributions of the conference. Each paper was reviewed by two experts. The contributions cover following areas:

- Keynotes on interdisciplinary issues (K: 3 oral presentations)
- Quality fertilizers from residues (TA: 29 oral presentations and 22 posters)
- Sustainable soils (TB: 8 oral presentations and 6 posters)
- Advances in emission prevention (TC: 25 oral presentations and 17 posters)
- The bioresource challenge (TD: 21 oral presentations and 8 posters)
- Sustainable regions (TE: 22 oral presentations and 12 posters)
- General thematic lectures (G: 5 oral presentations)

To deepen networking and to strengthen interdisciplinary not only among researches, but also with practitioners, politicians and public, some further initiatives were included in the conference (see www.ramiran2015.de):

- Podium discussion on rural-urban symbiosis
- Science and art presentations
- Manure management post-conference workshop
- Urban gardening post-conference workshop

The conference was attended by more than 200 delegates from more than 30 countries. Additional to the reviewed 4-page-papers an abstract book is available (www.ramiran2015.de). Selected contributions were invited for a more detailed publication in a special issue of the open-access online Journal „Energy, Environment & Sustainability". This special edition is titled "The bioresource challenge – A broader view on sustainable bioenergy systems coupled with substantial applications", edited by Ina Körner, Michael Narodoslawsky and Dagmar Fiedler (www.energsustainsoc.com).

Ina Körner
Conference Chair

About RAMIRAN

The "Recycling of Agricultural, Municipal and Industrial Residues in Agriculture Network (RAMIRAN)" is a research and expertise network dealing with environmental issues relating to the use of livestock manure and other organic residues in agriculture. RAMIRAN evolved in 1996 from the much smaller FAO Animal Waste Network, that had been active since 1978, and the scope was expanded to include other organic residues (industrial and municipal) which are used on land as organic manures and soil amendments. It is in principal a European network, but it is also open to interested experts from other parts of the world.

The network provides an invaluable means of exchanging ideas, information and experiences on topics that are becoming increasingly important at a national and international level. The main objectives of the network are to:

- Promote the exchange of methodologies, materials and processes;
- Progress knowledge on the environmental assessment of organic residues recycling in agriculture;
- Identify research priorities and initiate innovative collaborative activities that make use of the synergies resulting from the international network.

The main activity of RAMIRAN is a scientific conference organized every two years, usually attended by 150-250 participants. The RAMIRAN conferences are respected as the leading event in the field of manure and other organic residues used in agriculture in Europe. They provide an extensive overview of ongoing research and knowledge transfer activities concerning manure and other organic residues. This overview of *who is who* and *who does what* is an important prerequisite to the networking activities that RAMIRAN wants to foster.

With its participants, RAMIRAN holds a tremendous resource of knowledge and expertise in a wide range of topics across the whole of Europe and some countries in Northern America, Asia and even Oceania. The network represents a unique opportunity to mobilise this resource through network activities above and beyond the regular conferences. To use this potential, RAMIRAN fosters task groups, short-term teams with a clear task that can be achieved in a defined time of ideally 1-2 years and maximum four years. These tasks make use of the potential of RAMIRAN arising from its membership of experts. This means that, for example, surveys about management techniques, environmental, economic or social issues in connection with manure and other organic residues or interdisciplinary studies are ideal topics for such tasks. Past examples include residual Nitrogen effects from organic residues, anaerobic digestion and utilization of digestates. In 2003 and 2011 a group produced a "Glossary of Terms on Livestock Manure Management" which has proved very valuable in harmonizing the use of terms relevant to organic residues and their environmental relevance. At the 2013 conference in Versailles, it was suggested that the Glossary should be translated into different languages (a Russian version is now available) and that RAMIRAN should support its members to produce "Country Manure Profiles" providing an overview of the current practices and knowledge concerning organic residue management in the different countries.

With the special topic "Rural-Urban Symbiosis" the 16th RAMIRAN conference focused on closing the loop linking rural production and urban consumption systems and on the development of more sustainable solutions for the handling of residues. Once again this reflects the changing perception from waste and emissions towards benefits and resource use efficiency that has occurred throughout the lifetime of RAMIRAN. As Co-chairmen of the Network we thank the organizers for arranging this exciting and successful conference!

Tom Misselbrook and Harald Menzi
Network Coordinators

Thematic areas

TA: Quality fertilizers from residues

Agricultural production depends on the supply of plants with nutrients. Efficiency in agricultural production considers not only yields, but also product qualities and fertilizer footprints. Fertilizers provide nitrogen (N), phosphorous (P), potassium (K), calcium (Ca), magnesium (Mg), and sulphur (S) as macronutrients in varying proportions and forms. Furthermore micronutrients are needed in trace amounts, which are valuable not only for plant production, but also in follow up chains such as food consumption or anaerobic digestion. Trace nutrients in many foods have declined over the last half century and rock phosphate as the main source of P fertilizers will deplete in 50-100 years. In some locations, over-fertilization leads to water contamination, while in others high fertilizer prices leads to nutrient deficiencies in soils. The main source for N fertilizers is ammonia generated via the energy intensive Haber-Bosch process from atmospheric N. It is estimated that this process alone demands around 1.4% of the world's total energy consumption. Agricultural, municipal and industrial residues contain varying quantities of N, P and other nutrients and trace elements. They are often disposed of with environmentally damaging effects or through costly treatment processes e.g. by waste water treatment or incineration.

TB: Sustainable soils

Soil is a living body. It is a complex medium comprising mineral particles, organic matter, water, air and living organisms. Soil is an essential, very slowly-renewable resource, which provides many vital ecosystem services such as food and the production of other bioresources as well as filtration and retention of toxic substances and nutrients. Demands on soil are increasing as the world population and the per capita food demand continue to grow. In addition, the pressure to reduce consumption of fossil resources has led to a growing demand to provide bioresources as alternative sources for energy and raw materials. Soil overuse is increasingly leading to soil degradation, both in the EU and at a global level up to desertification. In line with sprawling urbanization, arable land is decreasing in quantity as well as in quality. Lacking direct legislation, soil degradation is now having trans-boundary impacts along with high economic costs. One means of improving soil quality is the use of organic residues generated by human activities as soil amendments for enhancing soil carbon levels and soil structure. However this practice is not without risks, namely the introduction of harmful substances such as antibiotics and other pollutants or unwanted nutrient losses.

TC: Advances in emission prevention

Farming is a source of emission of pollutants to the atmosphere and to water. A well-known problem is nutrient leaching and surface run-off, which may cause eutrophication of surface and groundwater bodies and is detrimental to drinking water quality and human health. The most-studied climate relevant gases are methane, carbon dioxide and nitrous oxide. Their atmospheric concentrations have increased in the last centuries due to human activities, including agriculture. Another important rural emission pathway is ammonia volatilization, arising largely from livestock manures and urea-based fertilizers. Together with other reactive nitrogen compounds, e.g. NO_X from processes in transport and industry, it leads to N deposition that damages susceptible ecosystems and leads to soil acidification Particulate matter originates from a range of agricultural sources, in particular the formation of secondary particulates from ammonia emissions, and may lead to a variety of health problems and associated social costs. In the future emissions may also be caused by new anthropogenic substances/compounds such as nanoparticles from nanomaterials. Urban emissions are numerous and may lead to the introduction of polluting substances (antibiotics, pharmaceuticals, heavy metals etc.) into agricultural chains with a feedback on urban systems.

TD: The bioresource challenge

The sustainable use and the protection of natural resources are essential for enduring food production and quality of life. In this context, bioresources will play a key role. Bioresources are non-fossil biogenic resources which can be used for multiple purposes: to produce food, substantial products such as paper, biobased plastics, biochemicals and composite materials or energy carriers such as bioethanol, biogas and heat. Bioresources are renewable, but they are not available in unlimited quantities and have limits to their utilization. Biobased economy encapsulates the vision of a future society no longer wholly dependent on fossil resources. The basics are bioresources originating from plants, animals, microorganisms or residues. In biorefineries they are converted into a multitude of products such as chemicals, materials, feed, fuels, and other energy carriers. Biorefineries are complex and integrated systems consisting of many process units. They take advantage of the various components contained in bioresources such as cellulose, hemicelluloses, starch, lignin, proteins, fats, oils, extractives and their intermediates. To date, the biorefinery industry is still in a nascent state, mostly using ligno-cellulosic feedstocks on larger scale. However, many concepts and approaches exist. Frequently discussed biorefinery systems with a connection to agriculture include sugar, starch, vegetable oil, lignocellulose, green, synthesis gas and biogas biorefinery.

TE: Sustainable regions

A sustainable agricultural system aims to deliver sufficient productivity, through the use of minimal and non-hazardous inputs, while maintaining soil quality and contributing to the reduction of environmental problems. The recycling of residues for fertilizing and soil quality improvement is still limited in practice. But urban and rural residues are increasingly not only a topic of disposal but of utilization. This provides an opportunity to bring rural and urban systems closer together again. However, practices involving recycling of residues might also cause environmental problems and lead to the evolution of unwanted compounds and pests.

Zero Waste is a visionary goal connected with changing people’s lifestyle and behaviour and traditional waste management practices. A holistic and integrative approach for their improved utilization is the “Civilization biorefinery” - a system aiming for complete and efficient utilization of secondary, tertiary and quaternary regional bioresources in a rural-urban symbiosis. It consists of three major parts - collection of the local bioresources, their conversion in a local network of centralized and decentralized technical units into material and energy products and the utilization of these products.

Towards Nitrogen neutrality at RAMIRAN 2015

Nitrogen (N) is an essential element for food provision - plants need to be fertilized and animals as well as humans need N as a nutrient too. But N can also cause manifold problems. There are *problems of too much N* - losses into environment contribute to eutrophication, acidification, global warming, and more. But there are also *problems of too little N* - soil resources depleting and endangering the livelihood of farmers, and threatening food security. A lot of effort is needed to better balance N-management.

A large event like RAMIRAN 2015 causes a considerable N-footprint. Various activities at RAMIRAN 2015 showed that institutional and individual responsibilities can contribute to a better balanced N-management. By these activities we want to raise awareness of the topic and show possibilities for progression towards N-neutrality. The "N-neutrality" approach suggested by the European Commission's Joint Research Centre (JRC, Institute for Environment and Sustainability Monitoring of Agricultural Resources) was considered.

At RAMIRAN 2015 the following activities were taken into consideration to lower the footprint or reactive nitrogen (Nr):

1. **Provision of tasty food with reduced Nr impact**

   Our first aim regarding food was to provide tasty food in sufficient amounts. But we also selected the menus regarding their N-footprint. For the lunch break we evaluated 28 meals and selected 10. In the coffee breaks we provided various selections of fresh fruits, which have generally a low N-footprint. For the gala dinner a table served menu was chosen instead of buffet to reduce food waste. Furthermore we asked for special diets of the participants (mixed cost, vegetarian, vegan, allergies and intolerances) in the registration procedure and considered the results in food provision, also to reduce food waste generation.

2. **Calculating of the Nr-impact**

   The N-impact of food provided at RAMIRAN 2015 was calculated on the basis of the N-footprint approach by Leip et al. (JRC). For that purpose we collected data regarding type and amount of all menu ingredients. For instance the average N-footprint of the meals prepared for lunch had a 9 % smaller Nr-footprint compared to an earlier conference in the same canteen and same working days. Additionally we studied the waste generation and the waste whereabouts in order to find out about used, recovered or lost Nr amounts and to define measures for improved N-management.

3. **Compensating the Nr-impact**

   All participants of RAMIRAN 2015 were asked to contribute a voluntary fee to donate to a sustainable food project in Indonesia (BEST, Institute for Integrated Social Economic Development, NGO). It focused on demonstration of vertical gardening as a special urban farming solution. The installations have the potential to be widely used in urban areas contributing to food provision, but mainly helping 'reconnect' people with their food systems, and provide more pleased and environmentally friendly sourroundings.

More information to the N neutrality approach and the calculations for the lunch meals are to be found in this proceedings book (Leip et al., 2015, page 26). More information on the overall evaluation of food provision and final whereabouts of residues is available on www.ramiran2015.de. Also information on the progress of the urban gardening project can be found on this page.

Art at RAMIRAN2015

"Art is the queen of all sciences communicating knowledge to all the generations of the world."
Leonardo da Vinci

The conference is not only an interface between rural and urban regions as well as between scientists, practitioners and politicians. It also shall connect the practical world with culture and art. Following artistically activities were carried during the conference to give it an communicative, but also enjoyable and relaxing atmosphere:

Acting for sustainability: A group of young researchers and practitioners with interdisciplinary experience in environmental governance uses theatre to promote intercultural dialogue on sustainability in the context of academic and public conferences. By combining scientific knowledge with artistic expression they appeal to the emotions, thus engaging their audience at a deeper level than can be achieved through mere intellectual argumentation and create a level of communication that engages participants with the heart as well as the mind.
(http://scientific-theatre.org/; Freiburg Science Theatre, t.floerkemeier@scientific-theatre.org)

Art from tetrapak: Christiane Lüdtke is a Hamburg artist. Sculpturing is one line of her activities. At RAMIRAN 2015 she presents a further line: etchings from tetrapak materials. Etching is traditionally method of printmaking where a metal surface is used to create a relief, which delivers the printing matrix. Mrs. Luedke is using Tetrapak as printing matrix which gives the pictures a very lively structure. She presents funny etching from various human situations as well as book marks.
(http://christianeluedtke.de/; kunst@christianeluedtke.de)

Rural-urban colors: Photographic images – original and artistically edited – were arranged to relaxing films for the conference breaks and upgraded with some statistical data to the conference for information. The focus of the images is on structures and colors from urban and rural environments taken from various distances. It ranges from extreme close-ups, where very small subjects appear in the photograph greater than life size up to photos taken with wide perspective.
The presentation is available under: https://www.youtube.com/watch?v=MbC44Aw4Ad8

(http://www.bioresource.eu; BioResourceInnovation, BRI, i.koerner@bioresource.eu)

Art at the Hamburg University of Technology: TUHH hosts various artworks ranging from photographs, over paintings up to sculptures, partly from internationally known artists (e.g. Hanne Darboven, Berto Lardera, Chui Wang, Alfred Mahlau). They are distributed within the buildings and the campus park. Some of the most impressive artworks were explained via a tour through the university. Information includes the manifold ways they came to the university, the techniques used and the partly difficult standing of art in a technical environment.
(http://kunst-tuhh.de/; stieglitz@tuhh.de)

The cell factory: Biorefineries are the foundation of the biobased economy. The major actors in biorefinery systems are microorganisms. The complexity of the processes within a microbial cell was visualized via a 3-D-model with an almost 1-meter-diameter. The model represented a fungal cell including their organelles. Also various enzymes were visualized in 3-D-form. The exhibition unit is accompanied by biorefinery feedstockes and products.
(http://www.tuhh.de/ibb/home.html; aze@tuhh.de).

Examples are shown on pages (1, 14, 30, 145, 230, 265, 291, 395, 461, 553, 581, 673)

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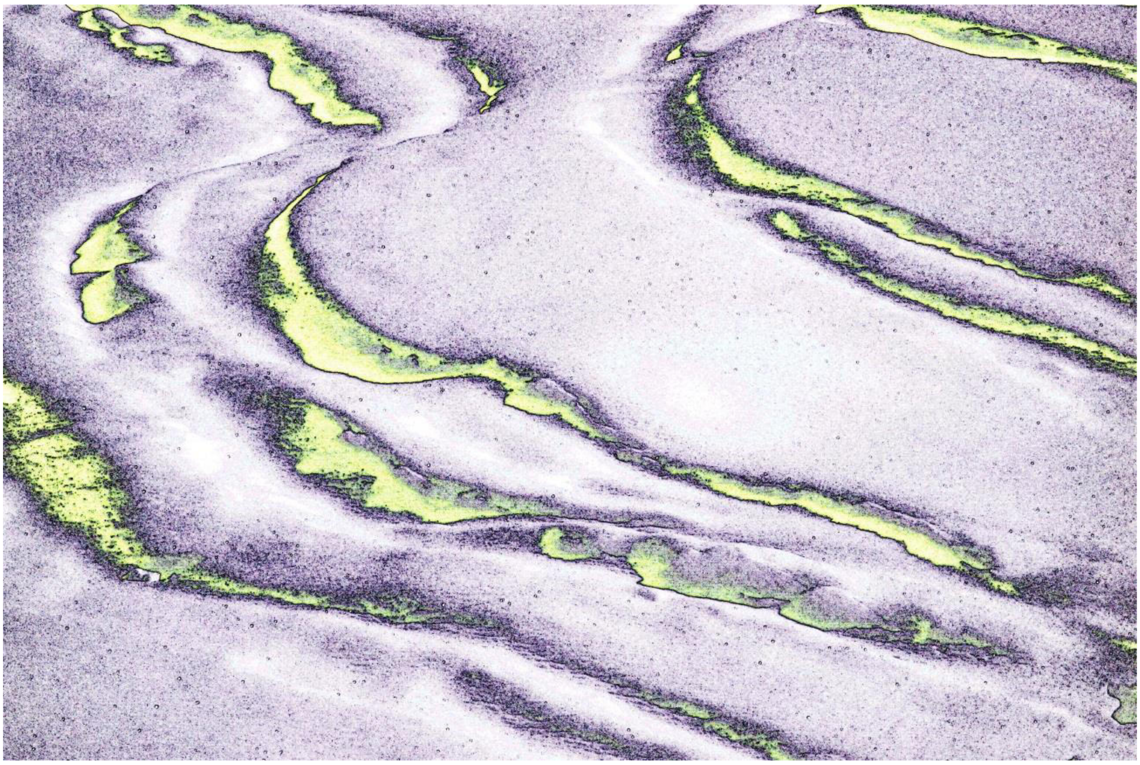

Ina Körner: Snow, processed photograph

K_01 Biobased economy in Germany – Sustainable supply of bioresources

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1. Policy Strategy Bioeconomy in Germany

In 2013, the Federal Government adopted the National Policy Strategy on Bioeconomy [1]. In doing so, the Federal Government is supporting the shift to a resourceefficient economy based on renewable resources that either makes less use of fossil resources or dispenses with them entirely. This change is mainly driven by the biobased economy.

The Policy Strategy on Bioeconomy builds upon the Federal Government's Sustainability Strategy. This dovetails with the "National Research Strategy Bioeconomy 2030 – our route towards a biobased economy", adopted in 2010, providing the foundation for innovations in the bioeconomy by means of research and development. In addition to other strategies and concepts formulated by the Federal Government the following programmes describe further points of policy orientation and conclusions with a direct effect on the bioeconomy:

- the "Energy Concept for an Environmentally Sound, Reliable and Affordable Energy Supply" (2010),
- the "Raw Materials Strategy" (2010),
- the "German Resource Efficiency Programme" (2012),
- the "Biorefineries Roadmap" (2012).

The bioeconomy affects various specific policy areas, such as industry and energy policy, the policy on agriculture, forestry and fisheries, climate policy and environmental policy, in addition to research and development policy. In aiming to give coherent structure to policy, the political framework conditions for the bioeconomy must be arranged so that, within the limits of what is possible, a contribution is made to securing world food supplies, reducing dependence on fossil-based raw materials, protecting the climate and using the renewable resources sustainably, while safeguarding both biodiversity and the functions performed by soil. In part, these requirements give rise to conflicts between goals, which need to be resolved by means of suitable framework conditions.

The concept of the bioeconomy takes natural cycles of materials as its point of orientation. It encompasses all sectors of the economy that produce, work and process, use, and trade with renewable resources, such as plants, animals, microorganisms, and their derivatives. Materials used include not only raw materials produced in the agricultural, forestry and fisheries sectors, as well as in aquaculture or in microbial production; increasingly, biogenic waste materials and residual materials are also used. The bioeconomy is thus also resource-efficient recycling. The renewable resources are worked and processed to form a variety of products (e.g. raw materials for chemical processes, insulating materials), also increasingly by means of industrial application of biotechnological and microbiological processes. Aside from its use for the production of materials, the use of sustainably produced biomass also acts as a significant renewable source of energy – with preference given to using it at the end of the cascading process of use.

2. Research Strategy Bioeconomy

In a global context as well as nationally, food security takes priority over the production of raw materials for industry and energy. Support must be given to the use of synergy effects between food production and the provision of raw materials for energy and industry.

Natural resources form the basis for the bioeconomy. Sustainable management of agricultural and forestry areas, and also of seas and other bodies of water, acts as the basic prerequisite for providing most of the raw materials in the bioeconomy. It is only by sparing resources and thus operating resource-efficiently that the necessary increase in agricultural production can be harmonised on a lasting basis with the protection of the environment, of the climate and of nature. This demands efforts that take into account all the factors involved

in the production systems, including location specific requirements and the aspects of sustainability.
Topics of research and development providing sustainable biomass include:

- improvement of the availability of traditional biomass,
- increasing productivity and efficiency,
- improvement of the availability of nontraditional biomass sources,
- diversification of agriculture and forestry (cultivation and usage),
- new plants and sources incl. residues and waste,
- adapting agriculture and forestry to climate change,
- breeding for adapted plants with high and stable yields and
- sustainable production of biomass.

In Germany current focus in research are new and improved industry and energy plants (breeding, cultivation and use), algae as feedstock, lignocellulose as feedstock, residues/waste usage, sustainable biofuels feedstock, biomass certification and sustainability and soil, water, air and nutrients aspects.
Topics of research and development regarding biobased chemicals and materials include:

- complete use of all biomass components and usage of the synthesis capabilities and preliminary performance of nature,
- development of biobased products having new functionalities and properties,
- improving the efficiency of known conversion processes and development of new, innovative technologies,
- process- and product-orientated RTD along the complete value chain and
- integrated conversion processes (Biorefineries).

Current focus in research are synthesis and use of biobased fine and speciality chemicals, chemical, biotechnological and technical use of starch and other carbohydrates, biopolymers and biomaterials, usage of lignin as feedstock, RRM (renewable raw materials) in housing and construction and wood usage.
Topics of research and development regarding bioenergy include:

- improving the efficiency of known conversion processes and development of new, innovative technologies (electricity, heat, biofuels),
- intelligent solutions for the combined usage of bioenergy and other renewable energies,
- improving the efficiency of decentralised bioenergy concepts,
- integrated conversion processes,
- biorefineries and
- sustainable bioenergy.

Current focus in research are solid bioenergy carrier, microbiology in biogas plants, residues from bioenergy plants, biofuels from lignocellulose, biotechnological production of hydrocarbon fuels and sustainability of biofuels.

3. Sustainable Biomass and Biorefineries

A biorefinery is characterised more by an explicitly integrative, multifunctional overall concept that uses biomass as a diverse source of raw materials for the sustainable and simultaneous generation of a spectrum of different intermediates and products (chemicals, materials, bioenergy/biofuels), whilst including the fullest possible use of all raw material components. The co-products can also be food and/or feed. By closing material cycles and through cascade utilisation and recycling, efforts are also made to prevent resource loss. Along the entire biorefinery value chain, all processes for the treatment and conversion of biomass should be resource efficient in terms of the use of materials and energy and in the consumption of media (e.g. water) and auxiliaries, and should avoid adverse environmental impacts. From the perspective of resource efficiency, goods and products of long lifetime are also of particular value [2].

There are two basic approaches to the implementation of a biorefinery concept: *'Bottom-up'* and *'top-down'*. If the biorefinery concerns the expansion of an existing biomass processing facility (e.g. sugar, starch, pulp mill, oil mill, ethanol plant), it is referred to as a *bottom-up* approach. In contrast, the term *top-down* is used when the emphasis is on newly conceptualised, highly integrated systems designed for the use of various biomass fractions and the (zero-waste) generation of a variety of products for different markets.

Sugar, starch and vegetable oils are the dominant commercial biorefining platforms. Primary refining is commercially established. Secondary refining is also commercially common, but mainly orientated to food and feed products including integrated biorefining. Established biorefining sites seek for a broader product portfolio with special focus on non-food products (special products, fine chemicals). This is the case for integrated sugar/starch biorefineries (e.g. Offstein, Barby, Zeitz, Krefeld, Hamburg) and integrated plant oil biorefineries (e.g. Düsseldorf, Hamburg). Hence, there is a focus on secondary refining. Lignocellulose (cellulose, hemicelluloses, lignin) is the platform of the pulp & paper industry. Nevertheless, additional value creation is targeted through additional secondary biorefining (e.g. fuel, new pulp-based materials, lignin separation, value-driven lignin products).

Primary and secondary refining exist on all technology readiness levels (TRL), but demonstration and commercial plants as integrated biorefineries are missing. Top-down concepts are still in the development stage for integrated biorefining.

The priority area for German R&D activities towards integrated biorefinery concepts will be to emphasise the *top-down* development approaches of lignocellulosic biorefineries (including the green biorefinery) and synthesis gas biorefineries. Work here will be spread out over a number of major projects. *Bottom-up* development approaches are being pursued for the sugar/starch biorefinery and the vegetable oil biorefinery, while the lignocellulosic biorefinery, which is based on existing structures in the pulp industry in Germany, has played almost no role in this area to date (e.g. in contrast to the Scandinavian countries), except some activities in Blankenstein and Arneburg. There have been comparatively few activities in the context of major projects with the *bottom-up* approach to development, as these often expand on existing structures in the sugar-, starch- or vegetable oil industries. For this reason, the dominant projects here are oriented towards providing support for such biorefineries, or are concerned with aspects of these biorefineries and/or with the conventional conversion of sugar, starch and vegetable oils into bio-based products or bioenergy.

4. Outlook

Agriculture, forestry, fisheries, and aquaculture, but also the biotechnological use and conversion of biomass, in addition to biogenic waste materials and residual materials: these are the central starting points for the bioeconomy's value chains and valueadding networks, which are interlinked in a multitude of ways. Downstream sectors work and process renewable resources to form a variety of products, partly also through industrial application of biotechnological and microbiological processes, particularly in the chemical industry. This also includes food producers, and the wood, paper, construction, leather, and textile industries, as well as parts of the pharmaceutical industry and the energy sector. They are as involved in the build-up of a bioeconomy as are the associated areas of retail, distribution and commercial service sectors. It is characteristic of the bioeconomy, firstly, that the value chains of its products in the various business sectors are increasingly networked, or respectively are able to be networked, and secondly that by-products and residual materials are used in a way that yields the highest possible value. Accordingly, the bioeconomy system also attaches particular significance to recycling and waste-management processes that can avoid residual materials and waste materials, or respectively direct them to a use that derives the highest possible value from them.

The bioeconomy has the potential to further expand this economic output, through the development and further processing of the various biomass-based raw materials – in some instances new ones – to form high-value, innovative materials and products, through increasing numbers of coupled uses and cascading uses, as well as through the optimisation and intelligent linking up of various value-adding networks.

The raw materials mix of the German chemical industry will shift in favour of renewables. Today, the German chemical industry uses slightly less than 19 million tonnes of fossil raw materials (petroleum products, natural gas and coal), 2.7 million tonnes of renewable raw materials and approximately 20 million tonnes of mineral raw materials. Renewable raw materials are used in the production of plastics, fibres, detergents, cosmetics, paints and coatings, printing inks, adhesives, building materials, hydraulic fluids, lubricants and medicines. In the coming years the corresponding segments of the chemical industry will experience higher rates of growth than those devoted to basic organic chemicals and the standard polymers. By 2030 the chemical industry will use about 50 per cent more renewable raw materials than today. This estimate represents merely the lower limit, because the potential of bioplastics cannot be foretold reliably [3].

References

[1] Federal Ministry of Food and Agriculture (2014): National Policy Strategy on Bioeconomy - Renewable resources and biotechnological processes as a basis for food, industry and energy. Brochure. Berlin, Germany (http://www.bmel.de/SharedDocs/Downloads/EN/Publications/NatPolicyStrategyBioeconomy.html)

[2] The Federal Government (2014): Biorefineries Roadmap. 2nd edition. Berlin, Germany (http://www.bmbf.de/publikationen/?pag=31?hilite=Biorefineries+Roadmap)

[3] German Chemical Industry Association (VCI) (2012): The German Chemical Industry in 2030. Frankfurt, Germany (https://www.vci.de/vci/downloads-vci/2012-12-10-vci-prognos-study-chemistry-2030-short-version.pdf)

K_02 Synergistic interaction of sanitation, biowaste utilization and energy systems towards water and food security

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1. Introduction

It has been found that most large civilizations were breaking down after degradation of large parts of their arable lands [1]. Modern day city life depends to a large extent on sufficient living soil for water renewal, food security and a balanced climate. Rural regions are the key to water- and food resilience. Over one-third of all arable land has been strongly degraded or destroyed globally [2] with the consequence of major changes of local climate, more drought and flooding and famine. Soil degradation is often occurring due to agro-chemical agriculture, non- adapted grazing and deforestation often for firewood. If meat and dairy production will be converted to adapted grazing patterns it could build humus on a large scale [3], give animals a good life and avoid the many downsides in concentrated industrial animal keeping. Rural populations usually lack support, but they can act with productive restoration methods and boost the local economy. Most of the powerful restoration solutions are not well known, some even contradict common beliefs. A stunning example is given by "Micacle Water Village" in India, a system that is now expanding [4]. Restoring local economy will often require building up living soils. In order to keep soil fertile, the returning of bio-waste and agricultural residues to the land is a key element. Thus it should as a basic rule better not be used for other purposes. The same is true for sanitation; organic matter and the nutrients mainly from human food consumption should be processed for soil building, ideally in a non-food sector. Bioenergy utilization has to be revised as it is often very inefficient, tends to weaken soils and in combination with agro-chemical agriculture contributes to leak toxic substances into the groundwater. An extensive study by the German National Academy of Science, Leopoldina has shown this very clearly, according to the study for most bioenergy with the exception of the utilization of wood for energy generation [5]. Terra Preta Sanitation was developed to combine the upsides of woodgas technology from stoves and heat/cooling-charcoal co-generation units. The charcoal has a potential for the improvement of soils in combination with proper composting processes in sanitation [6,7,8] as in biowaste management.

2. Downsides in the utilization of bio-resources and pathways forward

Utilization of bio-resources and even agro-chemical bio-energy was until recently often perceived as something positive. However, the recent boosting of a food-to-bio-energy by heavy subsidies in Germany has numerous downsides besides rather few upsides. One of the side effects is, that farming for food production becomes a challenge for many farms. Massively rising land prices by subsidies mainly for corn production often by investor's models are too high for normal food production. Recent adaptation of subsidies has led to the defaulting of many of those operations. This short article will only highlight some points of a far more complex picture. This list is intended to trigger a debate and help to get funding not only for positivistic research within a given context by political leadership over professional lobby groups.

- Biogas is most often a steady year round available renewable energy.
- Anaerobic methane production / biogas technology can be good as a treatment of residues that require thorough treatment and/or sanitization
- A major part of all bio-waste including excreta is needed to keep and strengthen the humus layer of arable lands with proper treatment and application
- Biogas production will often lead to humus deficiencies by converting most of the potential 'soil fodder' into energy. This is worst for corn monoculture without crop rotation. However, biogas operaors and researcher have put efforts into crop alternatives like the cup-plant (*Silphium perfoliatum)* and intercropping. Legal requirements for soil monitoring are put into practice, too.

- Net energy gains from biogas or ethanol production are very low; consideration of an energy equivalent of productive soils may well turn the balance to the negative
- Most energy crops are not grown organic (non-bio!) so there is water- and soil pollution by pesticides and mineral fertilizers coming along with it. A large part of commercial phosphate products are meanwhile of very poor quality and cause water- and irreversible soil contamination with uranium (700 tons/year in Germany alone, in [9] with information from Prof. Dr. Ewald Schnug, Julius-Kühn-Institut in Braunschweig, Germany) and cadmium
- In a world with hundreds of million people starving, children developing permanent brain damage through malnutrition all bio-energy and bio-material systems require rethinking and eventually, phase-out
- Wood utilization seems to offer many options for energy production with overall benefits [5], with the combination of woodgas stoves ans woodgas units for power, heat-cooling and charcoal production can be beneficial in the context of Terra Preta (Dark Soils) systems learning from the historic examples in the Amazon.
- Tree-Crops for erosion prevention and slope-farming [11] is a highly interesting and long ignored combination for the production of food, fodder, wood for many purposes. There are also many opportunities for soil improvement by tree-crops intercropping with field fruits. Totals yields of 150% have been reported from France.
- Instead of utilization of manure for energy production and secondary resources concentration of cattle cages with too little land should be reversed towards free grazing patterns [10]. This will also respect animals as fellow beings on Earth.
- Flush sanitation with its mixing of excreta into the wastewater is a major leak for the human food cycle; nutrients are destroyed or replaced from the land to the seas. Source separation technologies are available as high- and low-tech options [7]
- Land use patterns should be revised in order to revive the vast deteriorated lands. As a suggestion by the author built on old models [12] there can be New Towns consisting of organic micro-farms installed

The following paragraphs give some more background on how to shape a future that meets all demands and is not only serving sectors at the costs of others. Safe and clean water in good quantity alongside with local food production are a major topic for the resilience of the rural and urban areas around the world. Energy potentials from arable land are very small and come along with a high risk for water and food security. Direct solar energy and wind power generation is by far more efficient, especially with large scale grid connections between suitable regions.

3. Nexus Humus, Water and Food

One of the key parameters of the watershed is the humus content in the soils of the catchment. As stated above already between 1950 and 1990 over one-third of all arable land globally has been strongly degraded or destroyed [2]. The water profession is only now starting to acknowledge the high value of humus rich land in the catchment. Far sighted water utilities are supporting organic agriculture in the water catchment to avoid pollution with nitrate and pesticides while keeping or increasing the humus contents. Humus rich soils can absorb enormous amounts of water. Long and intensive rain events that could lead to devastating flooding in a catchment with degraded soils can be soaked up by living soil. The same water is retained and will contribute to restoring aquifers. Soils can remain moist for long periods of time, avoiding drought and reducing the need for irrigation. Just as the now common consideration of a ‘watershed’ there is the idea of looking at a suitable area around a city as the ‘foodshed’ [14]. Humus building besides crop rotation and leguminous plants can build on the full reuse of clean biowaste and further with resources oriented sanitation [15] like low-cost options of Terra Preta Sanitation [16]. Research on plant-microbes interaction has shown that contrary to common belief plant roots can take up macromolecules and even life microorganisms by endocytosis [17]. With these findings we must be even more careful with reuse of waste and wastewater, on the other hand there is an enormous potential for improving plant nutrition though well fed humus in organic systems.
When it comes to discussion on water efficiency in scientific, political and public debates there is a strange repetition of issues that are already well known. Drip irrigation is still described as an innovation, even though sub-surface systems can work with much less water (not all systems work

well though and will requiring water absorbing root soil). The very far reaching possibilities of rainwater harvesting on catchment level with contour swales, check dams, humus building and reforestation especially of uphill areas are mostly not mentioned. Even within the rainwater harvesting community the most efficient tool is mostly forgotten: improvement of the top soil quality in the whole catchment. Another crucial issue is to grow drought adapted plants where water is scarce instead of water intensive cash crops.
Reforestation is an important part of improvement of living top soils. Ideally and wherever suitable this should include food trees such as Moringa trees or sweet chestnut. The leaf and fruit of Moringa is excellent food and fodder with fast growing wood that can be used also for producing fuel for woodgas stoves. Theses stoves will produce plenty of charcoal for Terra Preta compost while cooking efficiently. Holistic Planned Grazing is a method of keeping larger-than-usual numbers of grazing animals for soil recovery in the way of high-animal-density-short-impact-time. Industrial style mass "production" of animals, manure pollution and hormone poisoning can thus be phased out. In exchange humans get animals which can express their life the natural way, contribute to soil improvement, get fed with healthy grass. This is a realistic base measure for restoring savannah regions on a large scale and in a very profitable way. This type of system has a factor four of productivity over the conventional grazing while restoring land and humus. See "Holistic Planned Grazing" by Allan Savory or Joel Salatin, Polyface Farm, USA.
Water utilization in cities and towns should also become more efficient. Many water efficient devices for households are on the market. Unfortunately, in many dry regions like Berlin in Germany there is a lack of political will to have long term plans for water efficiency. For example, savings in hot-water utilization like wash basins and showers are not encouraged even where politicians make the impression to really care for the limitation of fossil energy utilization. There are only very few shower heads that do fulfill hygiene regulations with 6 liters per minute (instead of the usual 18 and possibly far more) and allow pleasant showers. A small German company invented a unit with magnificent performance, whereby a vortex swirl inside the shower head producing a sort of bubble rain, with water drops filled with air. Another system is using intermittent feeding of the holes and is also pleasant and safe. However, the usual spray units that are on the market are unpleasant, produce dangerous aerosols where the presence of legionella becomes most dangerous. In addition it will be beneficial to manage rainwater runoff on site with infiltration to avoid the need for expensive central sewerage systems. Infiltration, too, opens many ways for producing water locally. Water efficiency can be an energy issue if water is produced by reverse-osmosis plants from seawater, as it happens more and more in the dry regions of the world.

4. Energy systems that restore soil

Supply of energy can be combined with soil restoration. Soil quality is far more important than the energy issue in the long run – food, water and a balanced local climate are crucial for all life. Terra Preta systems for soil improvement require good amounts of clean charcoal. Interestingly, woodgas stoves as well as woodgas-to-heat/cooling-and-electricity units use woody in a highly efficient way. In addition, they produce charcoal. This opens pathways to regenerative energy that is actually helping to improve top soil. The German NGO Climatefarming, founded by Jörg Fingas, has developed efficient movable metal woodgas stoves in Senegal and Burkina Faso which can actually be operated with briquettes made from woody waste and invasive reeds. Users will get a discount on new briquettes by handing in the charcoal produced while cooking. It makes sense to give new fuel in the weight equivalent what will equal a price reduction of around 25 to 30%. The company ProLehm (Mud-Brick-Producers in Germany) has developed a mud woodgas stove, the NOAH stove. It has a larger pyrolysis chamber to assure longer cooking times and is made for fixed installation. The NGO Climatefarming (Joerg Fingas) and the author are consultants to a huge rice farm in northern Senegal. This farm had a persistent problem of decreasing productivity after 20 years of agro chemical production. The soil had no more earthworms, which is highly alarming to the local farmers. We started co-composting of organic materials from the region with charcoal from a woodgas-to-electricity unit. It was installed by Climatefarming for the rice factory of the farm, running on the otherwise unusable waste material rice husk and stabilizing power supply in blackouts. Interestingly, the addition of charcoal and some organic material into the paddy fields could raise productivity on average by around 30% over 4 harvests in large scale trials. In order to restore soil quality further we have recommended introducing an organic system including intercropping and planting of Sesbania Rostrata or to switch to the dry rice system SRI. Charcoal application should always be part of composting processes, which can also avoid wind or water erosion of the charcoal particles. Woodgas devices that co-produce power, heat/cooling and charcoal are now available on the market in scales from around 20 KW up to several MW of electricity.

However, some woodgas units burn the charcoal for more energy production. It has to be considered that small units require quite some maintenance.

5. Re-Localization, Reverse-Migration and New Town development

Globalization of economy has turned to a disaster for many rural regions of the world. Re-localization of production, taking care of soil and water resources are not trade barriers but a basic method of good housekeeping. It is in the hands of rural communities to reverse their fortunes, reverse rural-to-urban migration and come up with integrated concepts. The methods described above are a great base for improving the local economy and become more resilient. The town of Hiware Bazar in India [4] has shown this in a stunning way, mainly with rainwater harvesting combined with reforestation, using water efficient crops and irrigation systems. The small town is thriving in a region where other towns are losing population through migration due to water scarcity and drought. New Town development in a kind of Gartenring is an initiative of the author to promote pathways to develop rural areas with a number of proven sustainable solutions. Unlimited urbanization has an enormous risk potential. Actual estimates for 70% of urban population by 2030 can be read as a horror scenario; considering that urban dwellers are mostly 100% dependent of outside supplies. Supply with clean and sufficient water and food are a consequence of land use. It is crucial to establish interesting new rural lifestyles e.g. with part time commercial gardening, only then there will be sufficient numbers of people working for and with living soils.

References

[1] Montgomery, David R.: Dirt: The Erosion of Civilizations, University of California Press (2007)

[2] UN Millennium Ecosystem Assessment, UNEP 2005

[3] Savory, Allan: Holistic Planned Grazing, TED Talk: how to restore soil on a large scale with keeping many more animals, http://youtu.be/vpTHi7O66pI (access June 2015)

[4] D'Costa, Robert: 'Miracle Water Village', India: Hiware Bazar, Integrated Wateshed Management led this and other villages out of water scarcity and poverty (https://youtu.be/9hmkgn0nBgk accessed July 2015) This and meanwhile many more projects where initiated by Hermann Bacher and Robert D'Costa (personal information June 2015)

[5] NN: Bioenergy: Chances and Limits, Leopoldina Nationale Akademie der Wissenschaften http://www.leopoldina.org/de/publikationen (2012)

[6] *Bettendorf, T., Wendland, C., Otterpohl, R. (edts): Terra Preta Sanitation, Deutsche Bundesstiftung Umwelt, ISBN* 978-3-00-046586-4 (2014)

[7] Factura, H.; Bettendorf, T.; Buzie, Ch.; Pieplow, H.; Reckin, J.; Otterpohl, R.: Terra Preta sanitation: re-discovered from an ancient Amazonian civilisation . Water Science & Technology, 61(10), 2673-2679 (2010)

[8] Otterpohl, Ralf: Boosting compost with biochar and bacteria. Nature, 486, 14 June 2012

[9] Cords, Detlev: NDR Dokumentation "45 Min - Gefährliches Trinkwasser" (Dangerous Drinking Water documentary in Northern German Public TV NDR documentary), 26. April, 2014

[10] Savory, Allan: Holistic Planned Grazing, TED Talk: how free grazing larger herds in planned rotational patterns reverse desertification, www.youtube.com/watch?feature=player_embedded&v=vpTHi7O66pI (access 4.3.2015), scientific material in www.savoryinstitute.com

[11] Schmith, Russel J.: Tree Crops – A permanent Agriculture, Harcourt, Brace and Company, USA, 1929, online since 2009, public domain, Soil and Health Library

[12] Howard, Sir Ebenezer: Garden Cities of Tomorrow, United Kingdom, 1902 (reprited from "To-morrow: A Peaceful Path to Real Reform", 1898)

[13] Otterpohl, Ralf: TEDx TUHH "New Town" https://youtu.be/_M0J2u9BrbU access 2015, (2013)

[14] Ackerman-Leist, Philip: REBUILDING THE FOODSHED – How to create local, sustainable and secure food systems, A Community Resilience Guide, Chelsea Green Publishing, 2013

[15] Otterpohl R. and Buzie C (2013): Treatment of the solid fraction. In: Source Separation and Decentralization for Wastewater Management Editor(s): Tove A. Larsen, Kai M. Udert and Judit Lienert Feb 2013 • ISBN: 9781843393481 Pages: 520 • Hardback, IWA Publishing, London

[16] Schober, Sabine: WTO/TUHH Award winning Terra Preta Toilet Design: http://youtu.be/w_R09cYq6ys (access June 2015)

[17] Paungfoo-Lonhienne, C.; Rentsch, D.; Robatzek, S.; Webb, R.I.; Sagulenko, E; Näsholm, T.; Schmidt, S.; Lonhienne, T.: Turning the Table: Plants Consume Microbes as a Source of Nutrients, PLoS ONE 5(7), July 30, 2010

K_03 Energy recovery and added value molecules extracted from agricultural bioresources and organic waste – A review of research needs, perspectives and prospects

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Waste generation is closely linked to population, urbanization and affluence. It remains a major challenge for municipalities to collect, recycle, treat and dispose of increasing quantities of waste (Bogner et al. 2007). There is indeed a number of existing waste management practices developed over the last twenty years which can provide effective solutions: composting and anaerobic digestion are among well-known and well-studied processes.

In addition, conservation of raw materials, improved energy and resource efficiency and fossil fuel avoidance are new requirements of the so-called "bio-based economy" which suggest to design and develop a new vision for the future of sustainable organic waste management. Strategic research has a key role to play in such a context.

In this brief overview, we will particularly describe two promising topics: energy recovery and added value molecules.

1. Existing waste management practices

Whatever their origin and processing scheme, the final valorisation of organic waste is definitively and hopefully the agronomic utilization and recycling through soil-plant systems. The aim is therefore to transform organic residues in compost or pellets, considering the easiest handling transport and a better agronomic efficiency. Indeed, 45% of European soils, show a deficit in organic matter content (Favoino, 2008), and organic wastes represent a resource well adapted as soil amendment and/or organic fertilizer. In France the overall supply of organic matter is estimated at 15 million tons from the agro-food industry, 25 million tons from municipal waste (from which only 3% are biologically treated (Ademe, 2008) and roughly 160 million tons of livestock wastes mostly in a solid form.

The waste characteristcics and their subsequent ability for dedicate processing being widely variables, any waste management scheme should consider:

- the physico-chemical analysis and classification depending on the origin of wastes (residual municipal wastes, sludges, agriculturals wastes),
- the development of methods to assess their biodegradability (respirometry and modelling),
- the optimization of dedicated processes including composting, drying, pelletizing, nutrients recovery (P and N).

Within the actual societal context showing a growing concern about energy supply and concurrently the need to tackle the climatic change, the valorisation of organic waste through anaerobic digestion (AD) technology is expanding. Indeed this technology has a combination of environmental, economical and societal advantages. It allows a capture of methane from waste through the production of renewable energy and heat. This technology allows therefore to reduce greenhouse gases from a number of sources and to reduce the consumption of fossil energy and compare well in terms of environmental impacts with landfilling (EEA, 2008) or even with direct composting (Ademe, 2007, Mata-Alvarez et al., 2000).

A predictive study published by the EEA (2008) envisage a reduction of GhG's from municipal waste management from 55 million tons equivalent CO_2 by the end of the 1980 to 10 million tons equivalent CO_2 by 2020. Similar results are expected from livestock effluents which share about 35% of greenhouses gases from agricultural origin. Moreover AD should help to a better balance of energy production through more decentralised systems helping to reach more independency and to reach the objective of 20% (or even 25%) of renewable energy production by 2020, considering environmental efficient systems and feasibility.

For the situation in France, a quick estimate of sources and potential methane production shows the following figures:

- from agro-food industrial streams : 0.5 to 1 Mtep (million tons of oil equivalent)
- sewage sludge : 0.1 to 0.2 Mtep
- agricultural wastes : 1 to 5 Mtep
- crop residues: 0 to 4 Mtep.

Additional Mtep could be consider if energy crops are include in the assessment although such development is under debate in Europe.The various estimates show therefore potentials ranging from 3 Mtep to 10 Mtep, unless one study arise estimates of 7 to 16 Mtep. Considering a realistic target (based on actual knowledge) about 6 Mtep could be produced through AD.

The challenge is therefore mostly environmental and if any energetic challenge it is rather local and particularly focus on the agricultural systems (in France the target is to built up to 1000 treatment plants by 2020).
However, unless the sewage sludges which could be treated alone, the manures do not present a methanogenic potential sufficient to be economically effective and needs addition of energetic co-substrates and then the development of appropriate and efficient collection and “sorting” schemes. The proper agronomic management of the digestates produce from the plant should be taken into account too.

The research needs on this topic will concern the four levels of the anaerobic digestion scheme:

- to evaluate and make reliable the estimates on the various waste sources streams
- to optimize methane production through the characterization and the transformation of mixed waste and co-substrates
- to improve the knowledge of microbial communities involved in the hydrolysis steps
- to develop strategies for enhance better valorisation of biogas

2. Future processes and solutions

2.1. Bioethanol production

Biofuel production from organic wastes is strongly supported by various EU and National legislations, which gives an aim of 10% of renewable energy consumption within the transport sector by 2020 while the situation is a 98% dependency of fossil fuel (OPECST, 2008). According to the road map for renewable energy sources, about 195 millions tons equivalent petrol (Mtep) of biomass would have to be used in 2020 to reach such goal. The utilization of every kind of biomass including biowastes is therefore needed, and the development of new routes is required (De Wit & Faaij, 2010).

Therefore, the production of bioethanol from secondary biomass like municipal wastes should attain in the future a, economical and environmental balance much more favourable than agro-ethanol of fossil fuel (Kalogo et al. 2007). Various companies have already anticipated such increase in demand and have elaborate patents in the USA (Process Masada Oxynol TM and Genesyst Internaational Inc. for ethanol production from residual municipal wastes. However the processes proposed are designated in a way similar to that used for agroethanol and include heavy and costly pre-treatments with the aim of optimizing the ethanol production (vapour explosion, acid/alkaline treatment, high pressure….). Such design seems inappropriate and un-adpated to the constraints related to the municipal wastes and require high investments and a long running time to obtain a result rather low. The estimated provide by Kalago et al. (2007) show that ethanol production from organic matter extracted from municipal wastes should represent only few percents of the biofuel needs in the USA.
The challenge is therefore to achieve bioethanol production from wastes, within a modified process derived from existing anaerobic digestions of solid wastes. Beyond the technological challenge, the implementation of such step necessitate to solve several unknown mechanisms arounf the production of a molecule through complex microbial communities from complex and non ster-

iles substrates! Indeed the alcoholic fermentation from yeast or from pure bacteria has been intensively studied. Whereas, there is a lack of documentation from alcoholic fermentation linked with complex anaerobic microbial communities as that found in municipal wastes.
Some authors like Ren et al. (1997 and 2007) and Temudo et al. (2007 and 2008) have experimentally demonstrated the possibility of producing ethanol from a mixed culture reactor receiving molasses or simple substrates (glucose and glycerol). Experimental conditions abale to allow ethanol production from waste have been observed by Bouchez (personal communication). On overall all this findings suggest a high potential but also the need to investigate particularly on:

- the definition of most favourable mixed wastes to optimize ethanol production,
- the optimization of the process parameters favouring ethanol production,
- the modelling of processes which deserve a new generation of mathematical models including thermodynamics parameters

2.2 Added value molecules

Unless the agronomic and energetic valorization which is the main routes, several others possibilities for an enhance valorisation of organic matter are being studied including the production of molecules of high added value or of economical interest. Such transformation will be encouraged in the future due to various reasons:

- The depletion of mineral phosphates resources (mines) which may strongly push towards the recycling of organic phosphates,
- The reduction of oil activities. More that 7.8% of fossil fuel energy is used within the chemical industry (OECD/IEA, 2005). Today, the substitution of fossil fuel by renewable energy is only seen from an energetic point of view without considering that the chemical production of molecules is connected to the refineries activities (Banhofer et al., 2008; Brehmer et al., 2009).
- The need to supply with biodegradable molecules in place of synthetic molecules used nowadays in the industry (Moiibroek et al., 2007; Brehmer et al., 2009).

The implementation of such valorization processes require however research efforts which vary according to the target substances.

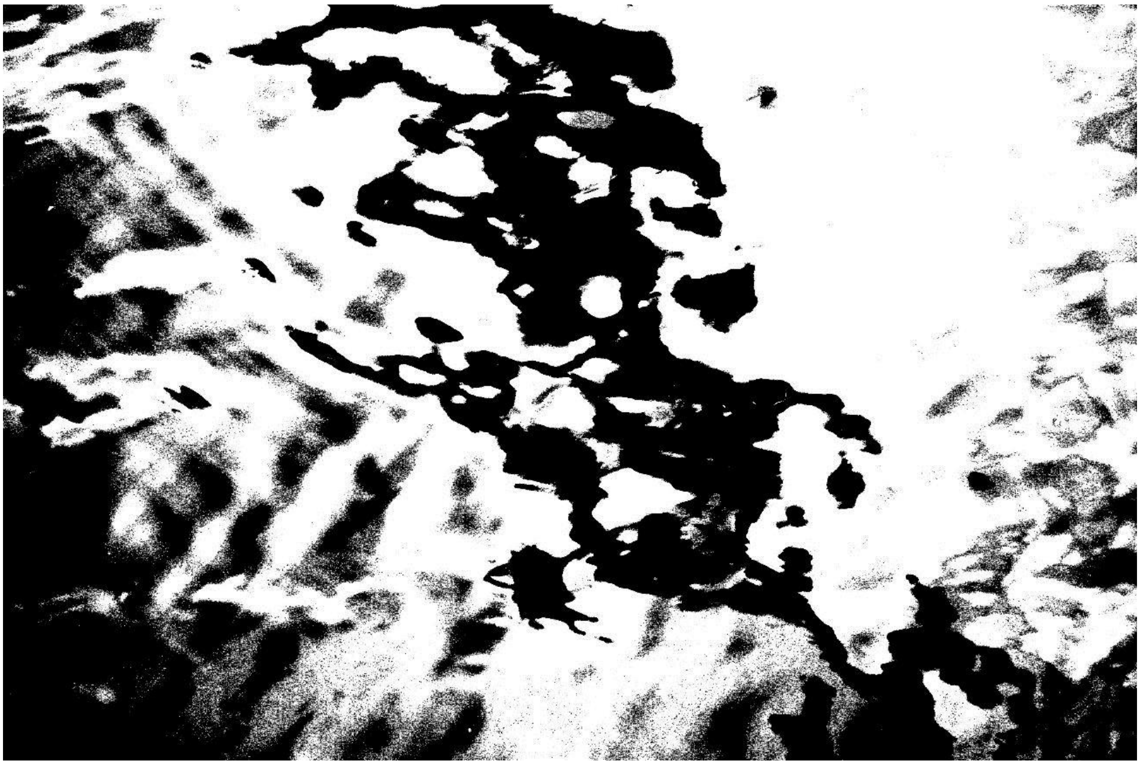
A first research line may focus on the extraction of phosphorus or nitrogen from liquid effluents. Work has already been developed in this way from landfill leachates, anaerobic digested liquors or animal effluents. The processes involved precipitation stages of phosphorus or struvites, to the stripping of ammonia or even to the use of membranes (Renou et al., 2008). Some of such processes have been implemented at the lab bench scale, other are at the field scale, but not necessarily with the aim of recycling nutrients.
A second research line may concern the development of production/extraction of industrial added value substances. To fulfil this line a prerequisite is the characterization (lignin content, cellulose, carbohydrates, amino-acids etc) from vegetables and wastes used which becomes thereafter a source of “renewable chemical compounds”.
This line is sub-divided into:
-the development of fermentation processes from wastes allowing the production of target substances like polyhydroxyalkanoates use to produce biodegradable plastics (Rabetafica et al., 2006; Du & Yu, 2002; Ujang et al., 2007; Koller et al., 2008) or amino-acids (Mooibroek et al., 2007).

Reference

[1] ADEME (2007). "Les déchets en chiffres."
[2] ADEME (2008). "Les installations de traitement d'ordures ménagères - Résultats 2006."
[3] Banhofer, W.F., Watson, K.J., *et al.* (2008) How might biofuels impact the chemical industry? Chemical Engineers Progress, 104(3): S7–14.
[4] Bogner, J., Abdelrafie Ahmed, M., Diaz, C., Faaij, A. Hashimoto, S. Mareckova, K. Pipatti, R. Zhang, T. waste management, In Climate Change 2007. : Mitigation. Contribution of Working Group III to the fourth Assessment Report of the intergovernmental Panel on Climate Change (Metz, B, Davidson, O.R., Bosch, P.R., Dave, R. Meyer, L.A. eds), Cambridge University Press, United Kingdom and new York, NY, USA.
[5] Brehmer, Boom, Sanders (2009) Maximum fossil fuel feedstock replacement potential of petrochemicals via biorefineries chemical engineering research and design 8, 7: 1103–1119.
[6] De Wit, M., Faaij, A. 2010. European biomass resource potential and costs. Biomass and Bioenergy 34, 188-202.
[7] Du G. and Yu J. (2002) Green Technology for Conversion of Food Scraps to Biodegradable Thermoplastic Polyhydroxyalkanoates. Environ. Sci. Technol., 36, 5511-5516.

[8] EEA, E. E. A. (2008). "Better management of municipal waste will reduce greenhouse gas emissions." Support document to EEA Briefing 2008/01.
[9] Favoino, E. (2008). "Strategies for the management of biowaste in the EU - optimising the C-cycle." Communication at the ECN/Orbit Workshop 2008 The future for Anaerobic Digestion of Organic Waste in Europe.
[10] Kalogo, Y., S. Habibi, H. L. Maclean, and S. V. Joshi. (2007). Environmental implications of municipal solid waste-derived ethanol. Environmental Science & Technology 41:35-41.
[11] Koller, M., A. Atlić, M. Dias, A. Reiterer and G. Braunegg (2008) Novel biotechnological processes for production of polymers, chemicals, and biofuels from waste Workshop on Biofuels and Bio-based Chemicals Trieste, 18-20 September 2008 Organized by ICS – UNIDO. Cf. http://www.ics.trieste.it/ VideoStore/BioWorkshop/2008.09.18_13.00-13.59/presentations/2_ Koller.ppt
[12] Mata-Alvarez, J., S. Mace, *et al.* (2000). "Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives." Bioresource Technology 74(1): 3-16.
[13] Mooibroek H; Oosterhuis, N; Giuseppin, M; et al. (2007) Assessment of technological options and economical feasibility for cyanophycin biopolymer and high-value amino acid production Applied Microbiology and Biotechnology. 77 : 257-267
[14] OECD/IEA. (2005) in Tréanton, K. (ed) (p. 351).
[15] OPECST (2008). Les biocarburants, p. 4. *In* A. nationale (ed.), vol. 2. OPECST - Assemblée nationale.
[16] Rabetafika, H., M. Paquot, P. Dubois (2006) Les polymères issus du végétal : matériaux à propriétés spécifiques pour des applications ciblées en industrie plastique. Biotechnol. Agron. Soc. Environ. 10 (3), 185 – 196.
[17] Ren, N. Q., B. Z. Wang, and J. C. Huang. (1997). Ethanol-type fermentation from carbohydrate in high rate acidogenic reactor. Biotechnology and Bioengineering 54:428-433.
[18] Ren, N. Q., D. F. Xing, B. E. Rittmann, L. H. Zhao, T. H. Xie, and X. Zhao. (2007). Microbial community structure of ethanol type fermentation in bio-hydrogen production. Environmental Microbiology 9:1112-1125.
[19] Renou S., Givaudan J.G., Poulain S., Dirassouyan F., Moulin P. (2008). Landfill leachate treatment : Review and opportunity. Journal of Hazardous Materials, 150: 468-493.
[20] Temudo, M. F., R. Kleerebezem, and M. van Loosdrecht. (2007). Influence of the pH on (open) mixed culture fermentation of glucose: A chemostat study. Biotechnology and Bioengineering 98:69-79.
[21] Temudo, M. F., R. Poldermans, R. Kleerebezem, and M. C. M. van Loosdrecht. (2008). Glycerol fermentation by (open) mixed cultures: A chemostat study. Biotechnology and Bioengineering 100:1088-1098.
[22] Ujang, SZ; Salim, MR; Din, MFM; et al. (2007). Intracellular biopolymer productions using mixed microbial cultures from fermented POME. Water Science and Technology. 56, 8 : 179-185.

General Thematic Lectures



Ina Körner: Water, processed photograph

G-O_01 Dealing with future European manure management challenges

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1. Objectives

European livestock produce about 1,270 million ton manure/year. Manure disposal represents a multiple challenge and raises a real need for more appropriate treatment options. Concomitantly, manure is also an important resource of nutrients and organic matter and their recycling may support societal transformation to a circular economy [1].

Our objective here is to present a vision for what potential policy responses should be to ensure a sustainable future manure management.

2. Methodology

The analysis focuses on exploring three core areas intricately linked to sustainable future manure management: nutrients, energy and environment.

Each of these issues are reviewed in terms of what policy responses should be with regard to manure management and what the trade-offs might be.

For each of these three areas we will focus on what should be done

- from a scientific perspective;
- from an acceptability perspective, both market and social; and
- from a policy and regulatory perspective.

A major basis for the analysis is the conclusions and outcomes from the large EU project *ReUse-Waste* (see www.reusewaste.eu), an EU FP7 Marie Curie Initial Training Network which will be concluded in 2015, see Fig. 1 for an overview of the project activities.

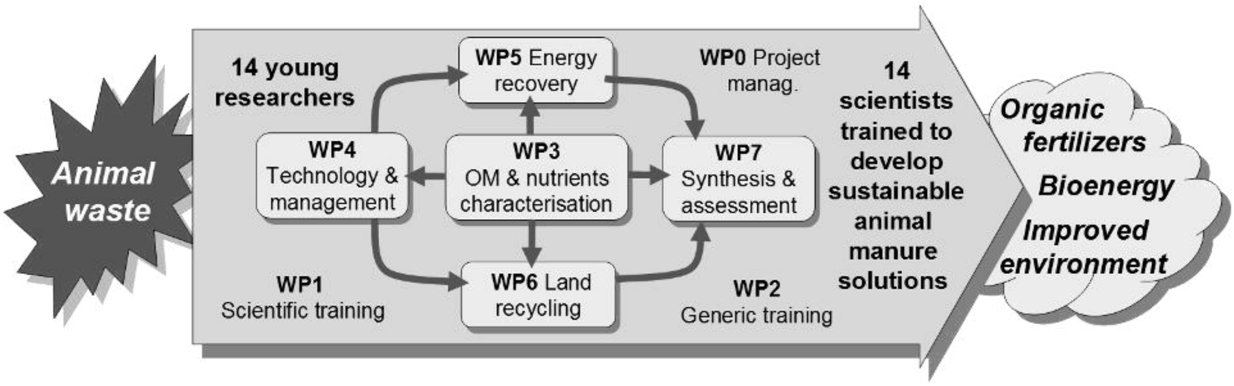

Figure 1 Overview of the ReUseWaste project

3. Results and discussion

Separation of animal manure or slurry into solid and liquid fractions may improve manure management as it makes handling more feasible, and enables fractions with a nutrient content in better balance with the plant/crop requirements. The solid fraction can serve as a co-substrate in biogas plants, and thus offers both an operational and economic advantage over digestion of non-separated manure alone [2]. Separation technology can be greatly improved, but these technologies have not been optimised economically or energetically or considering the end use of the products i.e. as plant nutrient and energy resources.

Combustion, gasification or pyrolysis of the solid fraction may enable utilisation of the energy content of the solid fraction; however, major obstacles exist in optimising thermal treatment processes for these relatively wet biomasses, and the fertilising properties of the residual products varies greatly.

Anaerobic digestion and combustion may also result in reduced C input to the soil, but technologies returning the most recalcitrant components of the manure as bio-based fertiliser could enhance soil C sequestration and quality and may reduce environmentally detrimental gas emissions from soil [3].

All these parameters will affect market and societal acceptability of such manure-based fertilisers. Technologies and policies to address the manure challenge should also account for (further) potential environmental effects of manure (mis)management.

4. Conclusion and outlook

We will present a vision for how to address the dual challenge posed by increased manure production.

We will focus on responses – scientific as well as policy oriented – that can reduce the environmental effect whilst optimizing the potential of manure as a bio-based fertiliser and energy resource.

References

[1] Kahiluoto et al. 2014 Taking planetary nutrient boundaries seriously: Can we feed the people? *Glob Food Sec* 3 16-21

[2] Hjorth et al 2009 Nutrient value, odour emission and energy production of manure as influenced by anaerobic digestion and separation. *Agron Sust Dev* 29, 329-338

[3] Diacomo & Montemurro 2010. Long-term effects of organic amendments on soil fertility. A review. *Agr Sust Dev* 30, 401–422

G-O_02 Marketing of biogas fermentation residues – The providers´ perspective

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1. Objectives

The number of biogas plants in Europe has increased significantly in the last decade. In Germany alone, almost 8.000 biogas plants were operating in 2014. The input materials (substrates) determine the nutrient and dry mass content of the fermentation residues. The dry mass content for example may vary from 1.5 to 45.7 % [1]. Due to restrictions in the timeframe and maximum amount of application, fermentation residues can often not be applied in the vicinity of the biogas plant. The situation is especially tense in regions with high nutrient surpluses. Figure 1 illustrates the regional nitrogen surpluses in German districts and reveals several hotspots.

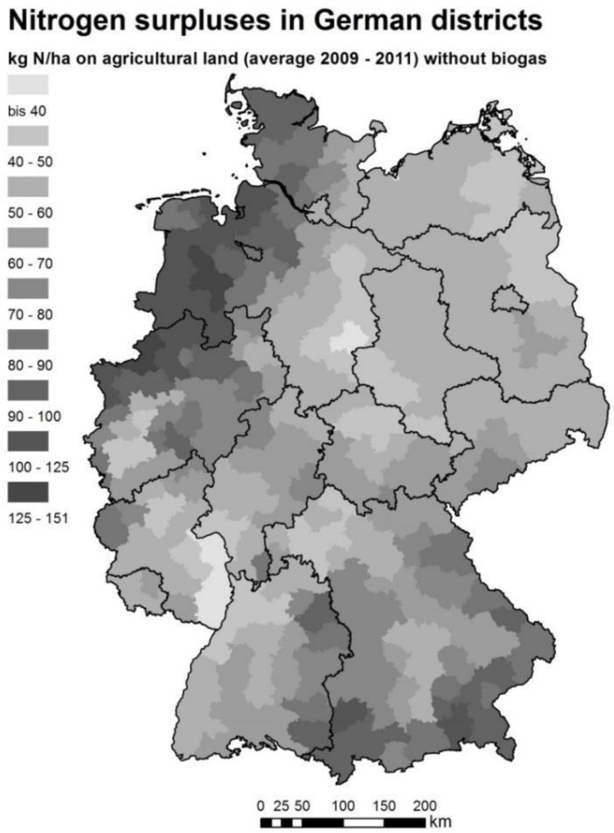

Figure 1: Nitrogen surplus regions in Germany - own figure based on data provided by [2]

Fermentation residues that cannot be applied as an organic fertilizer in the vicinity of the plant have to be marketed to other customers. There are two main approaches to cope with this issue:

- Market the products into regions with a nutrient demand
- Market the products into the non-agricultural sector

Upgrading technologies facilitate these approaches. There are several treatment options and technologies available. They range from the separation of the solid and the liquid phase of fermented residues to highly refined processes such as evaporation technologies. Due to a volume reduction and a value increase per cubic meter, the ratio of product value to transportation cost is increased and this facilitates the export of the final products to regions with a nutrient demand [3].

On the other hand the creation of novel products (e.g. pellets) also enables entry into new markets in the non-agricultural sector. The aim of this study was to investigate different approaches of current marketers. Supporting and inhibiting factors as well as different customer groups were analyzed from the providers` perspective. The valuable insights will be a helpful source for all actors who aim at marketing the fermentation residues of their biogas plant.

2. Methodology

Especially within new fields of research, such as the marketing of fermentation residues, an exploratory approach is a suitable starting point within qualitative research. An extensive literature review was conducted beforehand. Afterwards a semi-structured interview guide was designed that would enable the interviewees to respond to the questions very openly. We then conducted an internet search for marketers of biogas fermentation residues and contacted them by phone or e-mail. A total of 18 interviewees, familiar with the marketing of these products, agreed to be available for a face-to-face interview. The interviews were conducted in the second half of 2014 and took place in four different countries (Germany, Austria, Switzerland and the Netherlands). They were all recorded on tape and afterwards transcribed. The transcribed interviews then underwent a qualitative content analysis. We used the software MAXQDA to code and extract the most relevant information from the interviews.

3. Results and discussion

The results of the interviews are summarized in the following paragraphs and represent the viewpoints of the interviewees.

3.1 Market participants and motives

We have identified several actors that market the fermentation residues of biogas plants. These actors can be assigned to the following categories:

- Operators of biogas plants
- Agricultural contractors
- Potting soil and organic fertilizers manufacturers
- Others, such as nutrient exchange brokers

The motivation for marketing biogas fermentation residue based products is very diverse and mainly depends on the regional agricultural structure and the plant operators` background. Main drivers are:

- Regions with a nutrient surplus
- No application sites on the biogas operators` side (operators that do not operate any other agricultural businesses than the biogas plant)
- Legal incentives (heat incentive bonus under the German law)
- Legal restrictions (Biowaste Ordinance under the German law)
- Potential additional revenues

3.2 Target Groups and area of application

Fermentation residues are very heterogeneous and depending on their properties, suitable for different applications e.g. as a compost or organic fertilizer. Main target groups are farmers, landscaping companies and private gardeners. They use the products as a soil fertilizer and value their humus enhancing properties. In a few cases, the residues were stated to be used as a solid fuel. A very important but often resource intensive (time and money) aspect of reaching new customers is marketing. A lot of effort has to be put upfront into the promotion of a product. Thereby considering the customer demands is important in order to create a market-compliant and sought after product. In order to evaluate the different approaches we have applied the marketing-mix and looked at the product, price, place and promotion of biogas fermentation residues.

Product: Depending on the input materials and treatment method applied, there are different varieties available. For example raw fermentation residues, liquid ammonium sulfate, concentrate, granulate, pellets and beads among others. Dried products such as pellets can either solely originate from fermentation residues or mixed with other raw materials to reach the desired properties and nutrient contents. Dried fermentation residues may just be one ingredient among others in the recipe. Bulk weights of up to 600 kg per cubic meter can be reached by pellets and are favorable for transportation. In many cases, a consistent quality and uniformity is vital for products to be integrated with the existing technology, for example when applied through a fertilizer spreader by farmers. Products, descending from kitchen waste sometimes contain contaminants such as plastics, which is perceived by many customers as unfavorable.

Price: Prices for untreated fermentation residues range from 5 €/t to -15 €/t from the biogas plant. Untreated products, already applied on the field by agricultural contractors, reached prices of 60 % of the fertilizing value in spring and 40 % in autumn in an arable region. Upgraded products, such as pellets, achieve prices of up to 200 €/t from bulk buyers. Private gardeners are willing to pay up to 9 €/Liter for a garden fertilizer. The prices in the private garden sector seem to be very attractive. Nevertheless the increased marketing expenditure and margins of over 50% on the retailers` side have to be considered. The nutrient value of a product is often not reflected in the price – in fact, the high nutrient value is often unfavorable in nutrient surplus regions. Other factors that determine the price were stated to be season, input materials, agricultural structure and transportation cost.

Place: The distribution channels are very diverse and range from farm shops and internet sales to agricultural traders, fertilizer manufacturers and garden centers. The quantities sold through these distribution channels range from several hundred buckets of dried products (approximately 1 ton) to more than 200.000 tons of raw fermentation residues per year.

Promotion: The products where commonly being promoted through websites. Other promotion activities include workshops on the use of compost, field days for farmers and visitor days. Some marketers even offer the service of delivering composts to garden owners. Other activities include brochures, giveaways and sponsorships.

3.3 Customer perception

Farmers: From our respondents' perspective, many farmers using biogas fermentation residues value their fertilizing and soil enhancing properties. Farmers are in general very satisfied with the products. Especially organic farmers value the properties of the fermentation residues due to the waiving of mineral-based fertilizers. Products stemming from municipal and kitchen waste often received critical feedback. Unsatisfied customers were assigned to application errors by the interviewees.

Private gardeners and horticultural businesses: Most customers were stated to be very satisfied with products stemming from the biogas fermentation process. However many customers were not aware of using a biogas-based product. Due to the often negative perception of biogas in general, the origin of the product is often concealed by marketers. On the other hand private customers are often in favor of organic, regional and renewable products. A strong smell or dust-like quality was often perceived as negative. Other customers were stated to be just looking at the price tag. For most private customers, potting soil is just a brownish material therefore packaging is of high importance when targeting private customers.

3.4 Marketing barriers

We have observed several barriers that inhibit the marketing success of biogas fermentation residues. These are outlined below from the viewpoint of the interviewees:

Biogas plant operators: Most plant operators solely focus on the energy production and neglect the management of their fermentation residues. This can partly be ascribed to the fact that operators with smaller plants or with an agricultural background have usually a second pillar such as crop cultivation or animal husbandry and therefore limited time resources. The fixed costs for marketing expenditures are relatively high for smaller companies and on the other hand it is difficult for them to compete with large structures and companies. The disposal costs for unprocessed fermentation residues seem manageable and prevent many plant operators from pursuing further product enhancement and marketing approaches. The further processing of the fermented residues would implicate further financial investments into new technologies and an increased expenditure of time.

Manufacturers: Abundant amounts of homogeneous products are required by soil and organic fertilizer manufacturers. Fermentation residues from different input materials and varying plant sizes can only partially fulfill these requirements. Alternative substrates were stated to be more convenient to use for soil and organic fertilizer producers. In the case of peat for example, higher quantities with consistent and approved properties are available. Large-scale harvest is possible and economically supposed to be more profitable.

Garden centers: A wide product range is from a technical point of view not essential. However it is required by many retailers, garden centers and private customers. Product diversity as well as an established brand is valued by private customers. Therefore new brands often have a difficult time competing with established brands. Too many small suppliers are more time consuming for retailers and an impediment to get listed at corporations. Big corporations mostly prefer to collaborate with large suppliers.

3.5 Possible success factors from the providers` perspective

Biogas plants are very heterogeneous concerning the technology, size and the input materials used. It's therefore difficult to make general suggestions on success factors. However, the following aspects can be recommended. A specialization is always advisable, especially for smaller operators. When creating and marketing a novel product, they should focus on their core competence and if required, get assistance from external contractors. Cooperating with other partners is therefore advisable e.g. bulk buyers could take care of the product marketing. Another possibility mentioned was the merger of several biogas plants. This would enable a united effort, increase the product quantities and reduce the risk and time consumption for each member. A franchise model, which is already established in the composting sector, could also be implemented in the biogas sector. This approach would increase the awareness of the products and simplify the startup time.

4. Conclusion and outlook

The rapid expansion of the biogas sector, especially in Germany, is the result of strong subsidization. This development might have led to an increase of disposal problems for nutrients in some regions. Manure disposal prices from bordering Netherland were stated to be reaching prices of over 20 €/t in Northwest Germany. Biogas fermentation residues, commonly descending from a highly subsidized sector, now have to enter a free market. Although a great uptake of fermentation residues could be observed, there are still many well established alternatives for farmers that these products have to compete with. On the other hand the new fertilizer ordinance for Germany is under revision. Increased limitations on the use of biogas fermentation residues are expected. An important task for the biogas sector is first, to continue enhancing the treatment technologies and reducing the costs for the valorization of the fermentation residues and second, to create a market for the final products in the agricultural sector and beyond. This issue could be seen as an interesting business opportunity, especially for manufacturers of upgrading technologies or organic fertilizers.

Acknowledgements

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References

[1] Nkoa, R. (2014): Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. Agron. Sustain. Dev. 34 (2), 473–492.

[2] Bach, M. (2014): Berechnung von Stickstoff-Flächenbilanzen für die Landwirtschaft in Deutschland mit Regionalgliederung Kreise und kreisfreie Städte - Jahre 2003 bis 2011. Abschlussbericht, Univ. Gießen, 44 S.

[3] Golkowska, K., Vázquez-Rowe, I., Lebuf, V., Accoe, F., Koster, D. (2014): Assessing the treatment costs and the fertilizing value of the output products in digestate treatment systems. Water Sci. Technol. 69 (3), 656–662.

G-O_03 Implementing anaerobic digestion into the german agricultural emission inventory

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Objectives

Biogas production is a source of renewable energy and aims at reducing greenhouse gas emissions. Substrates for digestion often originate from agriculture and digestates are used as fertilizers. Therefore emissions from agriculture are affected by biogas production.
The IPCC (2006) guidelines for GHG inventories require to consider anaerobic digestion as method of manure treatment. For the German emission inventories a method was developed and implemented also considering the digestion of energy crops, as they have a great share in German biogas production and considerably affect the amount and properties of digestates. This methodology was used to calculate the emissions of GHG and NH_3 for the year 2013.

Methodology

Emissions are calculated along the chain of biogas production and digestate utilization. For methane the method described in IPCC (2006) was used as a basis. Due to errors in considering volatile solid contents it needed to be adapted. Also parameters needed to be chosen, that are available or can be deduced from research results.
For N_2O and NH_3 emissions no guidebook methodology is given. Therefore a methodology was developed to calculate emissions especially from pre- and post-digestion storage and the application of digestates. The methodology follows the calculation procedures for manure storage and application. Changes in NH_4^+ content during digestion are accounted for.
A procedure for the acquisition of activity data was also set up. It is based on the register of the transmission network operators and allows to identify each of the more than 8800 biogas plants. Data on substrate input and the distribution of gastight storage of digestates were available for a subset of biogas plants. By assuming analogy of substrate input for biogas plants of similar size and regional setting, overall input of the different substrates for biogas production was deduced.

Results

Integrating anaerobic digestion into the inventory calculations decreases emissions from manure management. Annual CH_4 emissions are reduced by 36.2 Gg (12.5%), N_2O by 0.43 Gg (4.3%) and NH_3 by 21.4 Gg. Emission reductions of all gases are mainly due to reduced emissions from storage of manures as a great share of digestate storages in Germany are gas tight.
If the method is also applied to the digestion of energy crops, additional emissions need to be accounted for. For German conditions, with a great amount of energy crops being used, these are 44.6 Gg for CH_4, 5.5 Gg for N_2O and 50.9 Gg for NH_3. This is mainly due to emissions from digestate storage (for CH_4) and the spreading of digestates (for NH_3). Additional N_2O emissions are mainly being derived from the N-input with digestates into soils.

Conclusion

The method of calculation allows to monitor the effect of changes in substrate use as well as digestion and digestate management on the emissions.
Assumptions on duration of pre-storage of manures and parameters of digestion still need to be verified. Resarch on leakages from biogas plants and emissions from digestate spreading is necessary. In order to acquire activity data on substrate input as well as storage and spreading techniques for digestates, surveys need to be adapted and improved to meet future needs.

References

[1] Rösemann C., Haenel H.-D., Rösemann C., Dämmgen U., Poddey E., Freibauer A., Wulf S., Eurich-Menden B., Döhler H., Schreiner C., Bauer B., Osterburg B. (20154): Calculations of gaseous and parti-culate emissions from German agriculture Berechnung von gas- und partikelförmigen Emissionen aus der deutschen Landwirtschaft 1990 – 2012. Report on methods and data (RMD). In: Thünen Report (in preparation)

G-O_04 Macroalgae as a resource for biobased industry

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1. Objectives

In the light of current environmental challenges a global switch for the production of high value products from biomass is highly demanded. Thus different concepts have been developed for the utilization of plant biomass based on starch or lignocellulose. Recently, degradation of marine biomass became more attractive. Especially macroalgae represent a promising alternative feedstock due to their lack of lignin, their abundance in nature and high content of carbohydrates [1-3]. Certainly their hydrolysis is a challenging task that requires efficient enzymes for the bioconversion of this highly complex substrate.

2. Methodology

For the identification of marine biomass-degrading enzymes both a sequence-based and an activity-based screening approach were applied. Metagenomic data sets were screened by the sequence-based approach using database comparison of annotated proteins. Moreover, enzyme activities were directly revealed in our activity-based approach by using azurine cross-linked (AZCL) polysaccharides. Promising candidates were produced heterologously in *Escherichia coli*. The recombinant proteins were purified to homogeneity by affinity and size-exclusion chromatography. Biochemical properties of the promising biocatalysts were determined. A photometric assay was applied for a quantitative determination of specific alginate lyase activities. For qualitative analysis we used cetylpyridinium agar plates. Characterizations of laminarinases were performed by DNS as well as BCA assay by determination of total amount of reducing sugars after enzymatic degradation of the preferred substrate. Furthermore, the performance of the enzymes regarding macroalgae hydrolysis was determined by HPLC.

3. Results and discussion

3.1 Identification of novel marine biomass-degrading enzymes

We report on the identification of various macroalgae-degrading enzymes by sequence-based screening approaches of metagenomic datasets from extreme environments with temperatures regarding from 4 ° to 80 ° C. Furthermore, we present biochemical characteristics of some of these promising biocatalysts that were heterologously produced in *Escherichia coli*.

3.1.1 Alginate lyases

One promising group of enzymes we identify during sequence-based screening approaches is alginate lyase that degrades the polysaccharide alginate by a ß-elimination reaction. Alginate is highly abundant in various species of brown algae and therefore represents a promising bioresource. For the identification of active alginate lyases alginate plates with the cationic surfactant cetylpyridinium chloride was used. Clear halos were observed indicating alginate lyase activity (Figure 1).

Following this screening approach on alginate plates and cetylpyridinium chloride we successfully identified several alginate lyases (Figure 1). Using this approach both thermophilic as well as psychrophilic microorganisms were identified. For a deeper look into the biochemical properties of these enzymes their activities were quantified by photometric measurements at A_{235}. Various parameters such as temperatures and pH values were tested. Furthermore, we determined the stability of these enzymes in presence of elevated temperatures, extreme pH values and detergents. For further characterization the enzymes were purified to homogeneity by affinity chromatography as well as size exclusion chromatography. The purification procedure is shown in Figure 2.

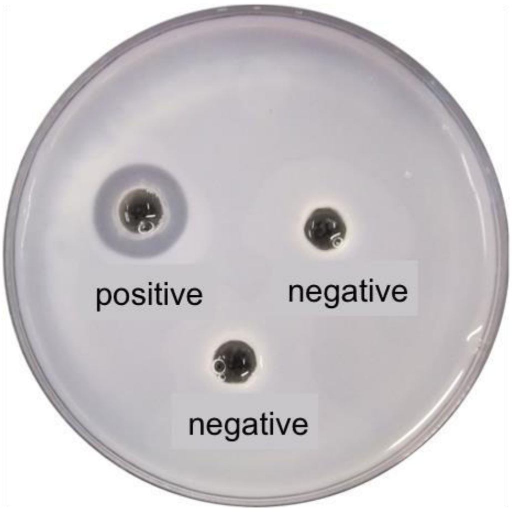

Figure 1: Screening approach for alginate lyase activity. For the detection of specific alginate lyase crude extract was used, that was obtained after cell lysis by French Pressure cell treatment or sonification. Incubation was performed for 2-3 hours at 37 °C on alginate plate (0.2 % (w/v)). Afterwards the plates were flooded with 10 % cetylpyridinium chloride to detect areas, where alginate was degraded enzymatically.

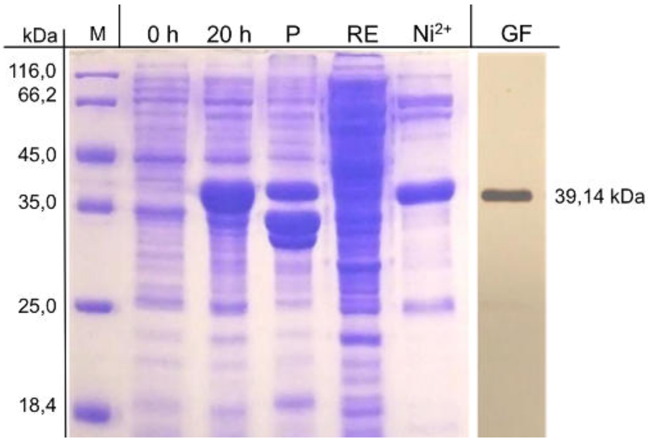

Figure 2: Purification of cold-active alginate lyase. For this SDS PAGE samples were taken prior induction (0 h), 20 hours after induction (20 h), from the pellet fraction, the soluble crude extract (RE) as well as pooled elution fractions from affinity chromatography (Ni^{2+}) and gel filtration (GF). The band of the corresponding fusion enzyme appears approximately at the predicted molecular weight of 39,14 kDa. To visualize the separated proteins the poly-acrylamide gel was dyed with Coomassie blue and silver staining agent.

The most efficient purification step was performed by Ni^{2+} affinity chromatography and size exclusion chromatography by gel filtration. The molecular mass of the enzymes was 39.14 kDa.

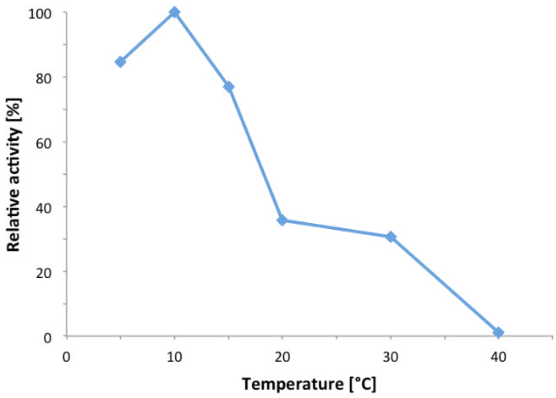

Figure 3: Temperature optimum of alginate lyase. Activity was measured photometrically at A_{235}. The temperature range for activity was between 5 °C and 40 °C. Reaction parameter are 0.0125 % (w/v) alginate from *Macrocystis pyrifera*, 20 mM Britton-Robinson buffer, pH 7 and 0.2064 U.

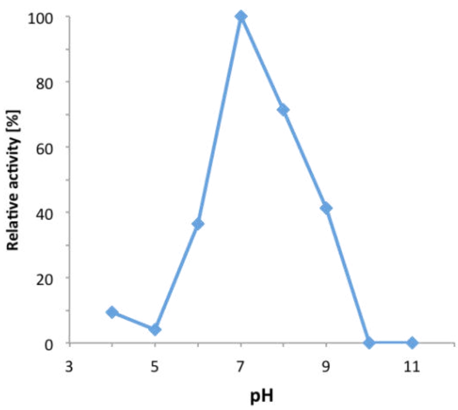

Figure 4: pH optimum of alginate lyase. Activity was measured photometrically at A235. The pH range for activity was between pH 6 and pH 9. Reaction parameter are 0.0125 % (w/v) alginate from *Macrocystis pyrifera*, 20 mM Britton-Robinson buffer, 10 °C and 0.2064 U.

As shown in Figure 3 the enzyme was highly active at 5 °C (>80 %) with an optimum at 10 °C. The optimal pH-value was pH 7.0 (Figure 4).

3.1.2 Laminarinases

Accordingly to the previously described alginate lyase we characterized a series of other promising enzymes capable of macroalgae degradation. Thus we also focused our efforts on identifying novel laminarinases. They belong to the enzyme class of endo-1,3(4)-β-glucanase (EC 3.2.1.6) and play an important role in the bioconversion of brown algae that contain high level of laminarin. The screening approach to identify novel laminarinases was accomplished by using AZCL polysaccharides. The preferred substrate used was pachyman, a 1,3-β-D-glucan. A representative screening plate with AZCL pachyman is shown in Figure 5. Due to laminarinase activity a blue halo appears around the corresponding enzyme.

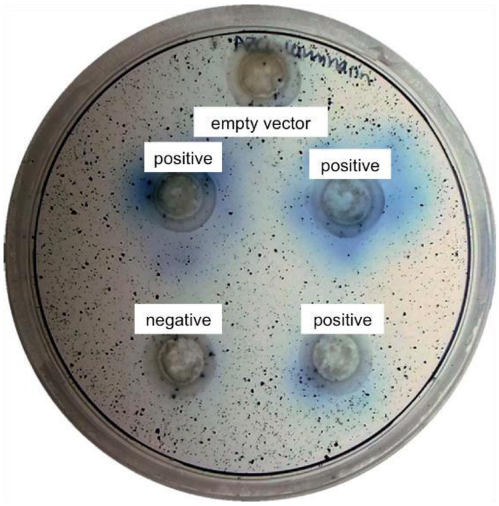

Figure 5: Screening approach for laminarinase activity on azurine cross-linked pachyman. For the detection of specific laminarinase activity crude extract was used, that was obtained after cell lysis by French Pressure cell treatment. Incubation was performed for 2 hours at 70 °C. Blue halos represent laminarinase activity.

Following this approach, we identified a series of putative laminarinases that were further examined regarding their specific biochemical properties. The corresponding enzymes were successfully purified to homogeneity by affinity and size exclusion chromatography. One of the purified laminarinase is heat-active and shows optimal activity at 80 °C with a high thermostability and a broad pH range (pH 5 – pH 8). These properties in combination with good performance on a degradation of the brown algae *Laminaria digitata* makes this enzyme a promising candidate for industrial applications at elevated temperatures.

4. Conclusion and outlook

We present results of the project LIPOMAR, "lipids and surface active molecules from marine biomass", that was established by a consortium of academic and industrial partners. Especially, the enzymatic bioconversion of beach-stranded macroalgae e.g. brown algae or seaweed is in focus. Various macroalgae-degrading enzymes, e.g. alginate lyases and laminarinases were identified by sequence-based or activity-based screening approaches of metagenomic datasets from extreme environments (4 – 80 °C, pH 3 – 10). Furthermore, we present biochemical characteristics of some of these biocatalysts that were heterologously produced in *Escherichia coli*. In more detail we describe the first cold-active alginate lyase with optimal activity at 10 °C and heat-active laminarinases that show optimal activity at elevated temperatures (>80 °C). The significance of these results on the bioconversion of macroalgae on industrial scale will be presented.

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References

[1] Kraan, S. (2012): Algal Polysaccharides, Novel Applications and Outlook, In: C.-F. Chang (ed) Carbohydrates - Comprehensive Studies on Glycobiology and Glycotechnology. InTech.

[2] Wargacki, A.J.; Leonard, E.; Win, M.N.; Regitsky, D.D.; Santos, C.N.; Kim, P.B.; Cooper, S.R.; Raisner, R.M.; Herman, A.; Sivitz, A.B.; Lakshmanaswamy, A.; Kashiyama, Y.; Baker, D., and Yoshikuni, Y. (2012): An engineered microbial platform for direct biofuel production from brown macroalgae, Science 335, 308-313.

[3] Wei, N.; Quarterman, J., and Jin, Y.S. (2013): Marine macroalgae: an untapped resource for producing fuels and chemicals, Trends Biotechnol 31, 70-77.

G-O_05 Towards nitrogen neutrality – Assessment of the tool on the example of conferences

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1. Objectives

Nitrogen (N) neutrality [1] is a new tool to increase awareness on environmental impacts related to the losses of reactive nitrogen (Nr) and the responsibility of an individual person or other entities. It includes measures to reduce and also to compensate losses of reactive nitrogen, in contrast to other available N footprint tools [2]. The objective of this paper is to assess the implementation of the N neutrality approach at the RAMIRAN2015 and compare it with the implementation of previous conferences (N2013 conference and 18th N workshop).

The current paper focuses on the quantification of the footprint factors of the food that will be offered at the RAMIRAN2015 conference. We quantified Nr footprint, C footprint, as well as the land use requirements of the meals. Other aspects of the N-neutrality concept are still in preparation and will be presented at the conference.

2. Methodology

To calculate achievements towards N neutrality for the conferences, three steps were carried out:

- Quantification of the N footprint of the food supplied at the conferences. For Europe, generic (food production) N footprint factors are available in [3]. We quantified a range of possible meals for which recipes were available, out of which a number of meals offered at the conference were selected;
- Choosing a project which may have a negative N footprint or where an additional reduction of the N footprint can be achieved;
- Linking the N footprint of the conference and the project via the financial support that is collected at the conference as a voluntary compensation fee and evaluating the achievements towards N neutrality.

For the quantification of the N footprint the following information was collected:

- Lunch meals will be provided by the cafeteria of TUHH. In cooperation with the responsible staff of the cafeteria, information on possible meals were collected for 28 lunch options [4], including a list of ingredients and their quantities used in each of the meals for the preparation of 100 portions. Thirteen options were offered for lunch of a previous 3-day conference. The meals from Tuesday and Wednesday, the same days RAMIRAN2015 is taking place, were used as baseline scenario. In addition, a number of 'Klimateller' meals were assessed which are usually served on Wednesdays. Furthermore the menu from the gala dinner and the coffee breaks was studied, but not considered here.
- The ingredients were grouped into about 200 ingredient types which were mapped with 144 foods available in the USDA National Nutrient Database [5]. The nutrient database served to quantify the amount of energy, proteins in the ingredients, as well as information on water content, density (to convert volumetric data into the unit of mass), and default values for the share of product refused (to convert amounts of ingredients to quantities supplied).
- The ingredients were further 'mapped' to one of 40 primary food groups (i.e. wheat instead of bread) that have been defined for the 'JRC Food Challenge Tool' [6] and for which default footprint factors (Nitrogen footprint, carbon footprint, and land use) were available. Most of the footprints factors were taken from the CAPRI data base [3,7], for other food groups (i.e. wild fish and fish from aquaculture, mushrooms, mussels, shrimps, yeast) new factors were developed [6] (See Table 1). A few ingredients used were 'ready-made' and default recipes were used to link those ingredients to the list of food groups

(pesto, ketchup, gnocchi and salad dressing) or obtained from the TUHH canteen. For the link between ingredient types and food groups, conversion factors were defined [6].

Table 1: Footprint factors used to quantify the impact of RAMIRAN menus with regard to emissions of reactive nitrogen (Nr), greenhouse gases (CO_2) and land.

	Food group	Nr [kg N/kg]	CO_2 [kg CO_{2eq}/kg]	Land [m^2/kg]	Source	Comment
Vegetable products						
35	SWHE	0.0112	1.2	2.7	[3,7]	Soft wheat
12	BARL	0.0087	1.2	2.9	[3,7]	Barley
40	MAIZ	0.0087	0.9	1.7	[3,7]	Maize
24	PARI	0.0200	2.1	2.3	[3,7]	Rice
18	OCER	0.0165	2.2	7.4	[3,7]	Other cereals
26	POTA	0.0024	0.2	0.5	[3,7]	Potatoes
28	PULS	0.0082	2.4	7.6	[3,7]	Pulses
31	SOYA	0.0103	2.0	5.9	[3,7]	Soya
29	RAPE	0.0058	1.1	2.6	[3,7]	Rape seed
34	SUNF	0.0223	2.5	6.8	[3,7]	Sunflower
38	TOMA	0.0012	0.1	0.2	[3,7]	Tomatoes
23	OVEG	0.0018	0.3	0.5	[3,7]	Other vegetables
11	APPL	0.0010	0.3	0.9	[3,7]	Apples
14	CITR	0.0032	1.0	2.0	[3,7]	Citrus fruits
36	TABO	0.0193	6.1	11.0	[3,7]	Table olives
37	TAGR	0.0018	0.4	0.9	[3,7]	Table grapes
20	OFRU	0.0012	0.7	1.4	[3,7]	Other fruits
21	OLIV	0.0097	1.4	3.9	[3,7]	Olive oil
22	OOIL	0.0058	1.1	2.6	[3,7]	Other vegetable oils
19	OCRO	0.0165	2.2	7.4	[3,7]	Other crops
			Animal products			
13	BEEF	0.2954	26.4	41.9	[3,7]	Beef
25	PORK	0.0594	9.1	8.2	[3,7]	Pork
27	POUM	0.0468	6.3	7.6	[3,7]	Poultry meat
30	SGMT	0.2300	25.3	37.5	[3,7]	Sheep and goats meat
15	COMI	0.0137	1.9	3.8	[3,7]	Cow milk
16	EGGS	0.0310	3.6	5.3	[3,7]	Eggs
3	LFIA	0.0070	2.2	2.0	[6]	Fish - aquaculture
4	LFIW	0.0029	2.1	0.0	[6]	Fish - inland water
8	SFIW	0.0005	0.3	0.0	[6]	Fish - wild fang
6	MUSH	0.0057	3.4	0.0	[6]	Mushrooms
7	MUSS	0.0001	0.1	0.0	[6]	Mussels
9	SHRI	0.0083	3.6	1.3	[6]	Shrimps
			Other			
32	STAR	0.0066	1.4	1.5	[3,7]	Starch
33	SUGA	0.0060	1.3	1.3	[3,7]	Sugar beet
17	MOLA	0.0066	1.4	1.5	[3,7]	Molasse (ingredient for yeast)
39	TWIN	0.0025	0.8	2.2	[3,7]	Table wine
10	SODA	0	0	0	[6]	Soda
2	ESSI	0	0	0	[6]	Vinegar
41	HONE	0	0	0	[6]	Honey
42	SALT	0	0	0	[6]	Salt

3. Results and discussion

The results for all menus are given in Table 2. Overall, there is a large range in energy intake (190 to 1680 kcal per portion) and protein intake (4 to 75 g proteins per portion). Accordingly, also the footprints cover a large range with 0.9-25 g Nr releases; 0.1 to 3.2 kg CO_2 equivalents emitted or 0.2 to 4 m^2 of land occupied for one year. This is explained by the fact that some of the dishes are not 'designed' to be full meals, but rather light (soups) or dessert meals which can be taken alone or in combination with other choices.

According to the data, meals included in the 2014 baseline menu (BIO) and meals identified as 'climate friendly' (KLI) are not significantly different with respect to nutrition value (BIO energy 670 ± 390; KLI energy 820 ± 400; BIO protein 26 ± 19; KLI protein 32 ± 19) and footprints (BIO Nr 6 ± 3.9; KLI Nr 6.8 ± 6.8; BIO GHG 0.8 ± 0.5; KLI GHG 1.0 ± 0.9) with higher average footprints and nutrition values for KLI than for BIO, however also with larger variance. The highest emissions of Nr in relation to the protein delivered is for the meal 'Cheese and leek soup with beef' (with 4 times as much Nr emissions compared to N intake with proteins) and the menu with the highest nitrogen efficiency is 'Gnocchi pan with soy strips' (less than one fifth of N emitted as Nr as compared to N intake with proteins). The same meals score also best with regard to greenhouse gas emissions, as compared to energy intake (KLIGPS (#22) 0.19 g CO_2-equivalent/kcal; BIOCLS (#5) 4.2 g CO_2-equivalent/kcal).

Table 2: Nutrient data and footprint values for the menus proposed by the TUHH cafeteria. All data calculated per portion served. qty: quantity in kg; Nr: losses of Nr in g; GHG: emissions of greenhouse gases in kg CO_2-equivalents; land: land use required in m^2/yr; water: average weighted water content of the ingredients in percent; energy: energy content of menu in kcal; protein: protein content in g.

ID	menuName	Acro-nym[$]	qty	Nr	GHG	Land	Wa-ter	En-erg	Prot
1	Apple fritters with vanilla sauce	BIOAFV	0.26	2.6	0.39	0.76	0.17	462	6
2	Cabbage rolls with cumin sauce	BIOCRC	0.60	4.0	0.57	0.73	0.39	577	13
3	Pollock with broccoli filling	BIOPBF	0.41	2.9	0.39	0.52	0.28	533	43
4	Penne pasta with oyster mushrooms	BIOPPO	0.66	6.3	1.11	2.20	0.40	1259	23
5	Cheese and leek soup with beef	BIOCLS[b]	0.46	14.1	1.37	2.31	0.32	330	20
6*	Tortellini with mince beef	BIOTMB[b]	0.23	4.7	0.45	0.78	0.14	401	11
7*	Escalope chasseur breaded pork	BIOECB[b]	0.65	11.3	1.86	1.71	0.39	809	33
8*	Cauliflower cheese patty	BIOCCP[b]	0.73	3.3	0.43	0.82	0.45	636	13
9*	Pollock filet natural fried	BIOPFN[b]	0.81	2.4	0.37	0.63	0.54	701	47
10*	Chicken soup with vegetables	BIOCSV[c]	0.51	4.6	0.64	0.90	0.40	191	11
11*	Chicken Tandoori with ginger carrots	BIOCTG[c]	0.55	10.6	1.39	1.93	0.30	711	30
12	Soy ragout with peppers and mushrooms	BIOSRP[c]	0.45	2.4	0.45	0.66	0.30	489	20
13	Turkey steak fried with broccoli, gnocchi	BIOTSF[c]	1.00	8.9	1.18	1.58	0.47	1621	75
14*	Carrot soup with chervil	KLICSC	0.59	0.9	0.11	0.21	0.46	253	4
15	Fish cakes tartar sauce chives potatoes	KLIFCT	0.65	2.4	0.84	0.43	0.47	619	58
16	Chicken strips al'arrabiata green tagliatel-le	KLICSA	0.48	7.2	0.94	1.31	0.25	714	28
17*	Asian vegetable stir-fry with tofu rice	KLIAVS	0.48	1.8	0.24	0.52	0.30	638	25
18*	Filet of duck breast cream sauce carrots	KLIFDB	0.70	13.5	1.76	2.70	0.32	1676	46
19	Apple fritters vanilla sauce	KLIAFV	0.26	2.6	0.39	0.76	0.17	462	6
20*	Caribbean lentil curry rice	KLICLC	0.60	3.1	0.51	1.14	0.32	964	40
21	Chicken leg pommes frites tomato salad	KLICLP	0.84	13.1	1.75	2.29	0.44	1229	32
22	Gnocchi pan with soy strips and Pak Choi	KLIGPS	1.09	1.9	0.27	0.58	0.67	1380	60
23	Turkey gyros rice vegetable salad yogurt	KLITGR	0.73	13.1	1.74	2.44	0.43	1022	45
24	Warm apple strudel vanilla sauce	KLIWAS	0.22	1.9	0.28	0.55	0.13	391	6
25	Penne pasta with mushrooms tomatoes	KLIPPM	0.46	2.3	0.64	0.37	0.31	482	21
26	Onion chicken Risi Bisi	KLIOCR	0.41	10.8	1.44	2.09	0.19	543	24
27	Vegetable stir fry with chickpeas	KLIVSF	0.63	3.2	0.47	1.29	0.38	961	31
28	Half a chicken BBQ sauce french fries	KLIHCB	1.09	24.3	3.24	3.98	0.52	1000	58

*Meals selected for the RAMIRAN2015 conference based on the N footprint evaluation and culinary issues; [$]Meals includ-ed in the BIOCAT 2014 baseline menu are identified by an acronym starting with BIO. Meals included in the Klimateller baseline menu are identified by an acronym starting with KLI. b) meals from the BIOCAT conference on Tuesday; c) BIO-CAT meals served on Wedneday were at the same time 'Klimateller'; b+c) represents the baseline as, being the same days as the RAMIRAN conference.

Based on this data, culinary aspects and other criteria, the following selection was made for RAMIRAN2015: BIOTMB, BIOECB, BIOCCP, BIOPFN, BIOCSV, BIOCTG, KLICSCC, KLIAVS,

KLIFDB, KLICLC (marked with a * in Table 2), with an average impact of 6.3 g Nr and 0.75 kg CO_2-equivalent per portion, an energy intake of 720 kcal and a protein intake of 26 g protein per portion. For the estimated 250 participants, total losses of Nr of about 1.5 kg Nr and GHG emissions of about 190 kg CO_2-equivalents will be caused by the lunches served at the conference requiring about 290 m^2 of land to produce the food.
Generally, the footprints of the meals proposed are in the low range. For example, at the N2013 conference (Kampala, December 2013), the average losses of Nr for a lunch was 29 g Nr, mainly due to high shares of meat (15% by weight) and about 50% larger quantities provided [1]. The N footprint per meal estimated for the 18th Nitrogen Workshop (Lisbon, July 2014) was 17 g N, with emissions of reactive nitrogen of about 9 g Nr (unpublished). Note that the N footprint includes also N losses as N_2 (inert molecular nitrogen) which is not accounted for the total losses of Nr. On the other hand, losses related to energy use in farming are included in the total losses of Nr, but not in the N footprint.

Likewise, Nr emissions from food production of the meals included in the JRC Food Challenge average 32 g Nr per meal of 900 kcal and total GHG emissions were 4.0 kg CO_2-equivalent per portion of 900 kcal [6].

Compared to the 'baseline' situation of meals offered by the TUHH canteen to the BIOCAT conference on the same work days, the selection for the RAMIRAN conference has 9% smaller Nr footprint and land use requirements and 17% smaller C-footprint. Protein intake is also smaller by 9% but with 26 g per portion about half the recommended daily intake, and thus suitable for a meal consumed at lunch.

4. Conclusion and outlook

We quantified so far the possible environmental impact of RAMIRAN2015 meals with regard to losses of Nr, greenhouse gas emissions, and land use. The data will be improved during the conference by taking into consideration 'real' data measured during the conference including the whereabouts of the residues. Participants of the conference were asked to contribute 30 € per participant in order to compensate the nitrogen losses for a project which helps reducing the N-footprint of food production. The project chosen for support is an ongoing Urban gardening project in Indonesia, especially in densely populated kampong areas in Jakarta. Next steps for achieving N-neutrality are the quantification of the overall reduction of N-losses that can be achieved with the voluntary compensation contributions.

The N neutrality approach has proven to be an efficient tool for increasing the sustainability of a conference. However, even though the concept and general methodology is well developed, the complexity of the food chain and N interactions in the environment leaves many options for its implementation. RAMIRAN2015 is an important milestone to formulate standards and improve future implementations.

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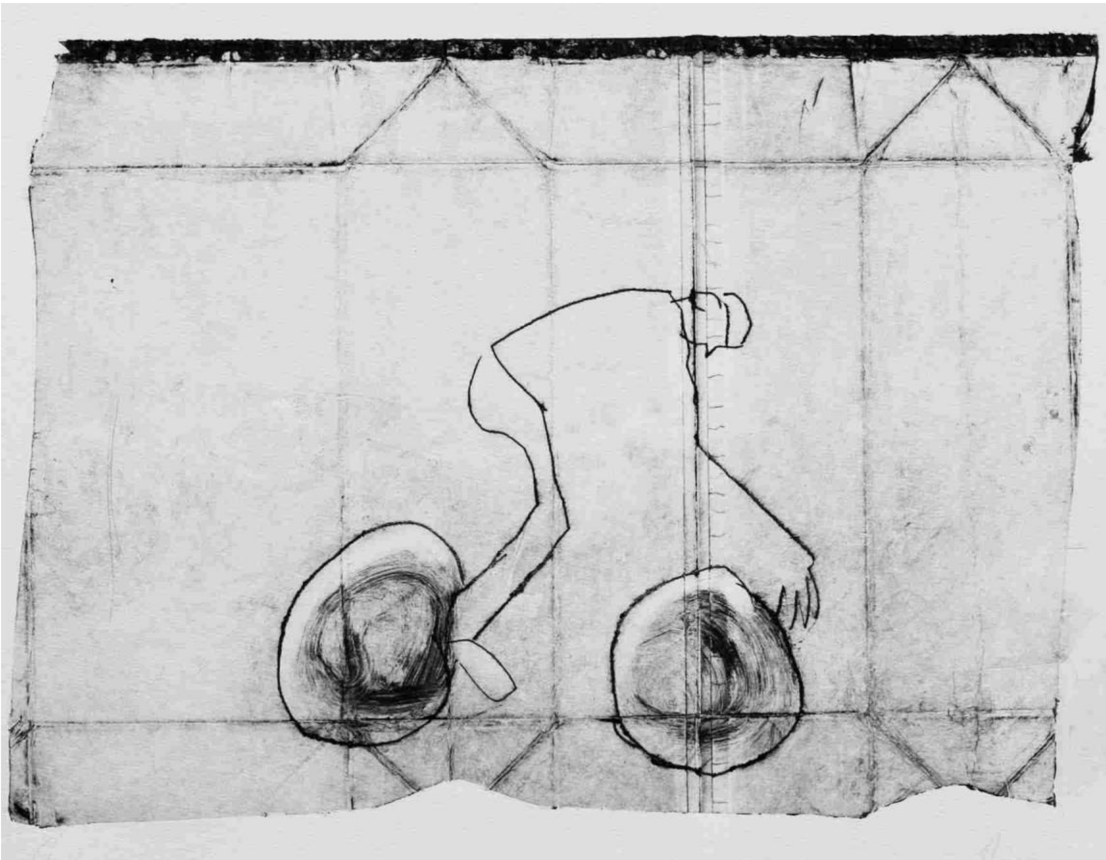
References

[1] A. Leip, A. Leach, P. Musinguzi, T. Tumwesigye, G. Olupot, J. Stephen Tenywa, J. Mudiope, O. Hutton, C.M. d S. Cordovil, M. Bekunda, J. Galloway, *Nitrogen-neutrality: a step towards sustainability*. Environ. Res. Lett. 9 (2014) 115001.
[2] J.N. Galloway, W. Winiwarter, A. Leip, A.M. Leach, A. Bleeker, J.W. Erisman, *Nitrogen Footprints: Past, Present and Future*. Environ. Res. Lett. 9 (2014) 115003.
[3] A. Leip, F. Weiss, J.P. Lesschen, H. Westhoek, *The nitrogen footprint of food products in the European Union*. J. Agric. Sci. 152 (2014) 20.
[4] C. Orozco García, Analysis of the Food Chain Regarding Nitrogen Footprint on the Example of an International Conference. 3rd Semester Project for the Joint European Master in Environmental Studies – Cities and Sustainability, Institute fuer Abwasserwirtschaft und Gewaesserschutz, Technische Universitaet Hamburg-Harburg, 2015.
[5] USDA, USDA National Nutrient Database for Standard Reference, Release 27 (revised). Version Current: May 2015, US Department of Agriculture, Agricultural Research Service, Nutrient Data Laboratory, n.d.
[6] JRC, An Interactive Game Assessing the Carbon, Nitrogen, and Land Use Footprint of European Dishes. Installed at Silvia's Lab, Ispra, EXPO2015, 2015.
[7] A. Leip, G. Billen, J. Garnier, B. Grizzetti, L. Lassaletta, F. Raes, S. Reis, M.A. Sutton, W. De Vries, F. Weiss, H. Westhoek, *The environmental Impact of European Livestock Production*. ERL submitted (n.d.).

Thematic area TA – Quality fertilizers from residues (Oral presentations)



Christiane Lüdtke: Art from Tetrapak

TA-K_01 Example of a country manure management profile – Switzerland

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1. Background and objectives

Manure and other organic fertilisers are a crucial aspect of the interactions between agriculture and the environment. On the one hand, the local recycling of manure saves non-renewable resources (mineral fertilisers) and reduces nutrient flows, on the other hand, manure is also a major source of emissions and other impacts on the environment. Especially in grassland areas, where livestock production is the major or only branch of agricultural production, manure recycling and good manure management practice is essential to limit the environmental impact of agriculture. To assess the importance, the potential benefits and impacts, as well as, generally, the strengths and weaknesses of the use of manure and other organic fertilisers in agriculture, a holistic view at the farm, regional and national level is important. This requires information on the structure of agriculture, the total flows of plant nutrients and organic substance, common management practice, the relevant policy and economic framework etc. In most countries no compiled source of such information is available. To bridge this gap, the idea of compiling "Country manure management profiles" (CMMP) in a more or less standardized form was discussed at the RAMIRAN conference in 2013.

RAMIRAN uses the instrument of "task groups" to compile transnational information and to make optimal use of the unique potential of the expertise of its wide membership. Using Switzerland as an example, this contribution is meant to illustrate the potential content and structure of such a CMMP as a basis for discussion in a task group how and how far a common template should be established for the CMMPs. As a full CMMP would certainly surpass the scope of this contribution, information on the Swiss situation will only be given "in a nutshell". A full CMMP will be available at the RAMIRAN conference and www.ramiran.net.

2. Methodology

This section should briefly summarize how the information on the current management practice, the relevant framework conditions (policy, structural conditions etc.), recommendations and management aids was compiled, analyzed and interpreted. For the Swiss profile the information is mainly based on the results of representative surveys on farm management technique (most recent for 2010, approx.. 3000 farms; Kupper et al. 2015), on the national nutrient balance, on the official "guidelines on fertilization of arable crops and grassland" and legislative requirements.

3. Manure management in Switzerland

3.1 Role of manure in fertilization

This section should give a general overview of the contribution of manure (and other organic residues) in the nutrient flows of agriculture.

Livestock manure is by far the most important fertiliser in Swiss agriculture. In 2010, the average amount of nutrients used per hectare of utilized agricultural area (UAA; excluding alpine pastures) was 179 kg N (total N), 26 kg P and 165 kg K (data from Spiess 2011, supplemented). Manure (livestock excreta with no account of losses) contributed 70% of N, 80% of P and 85% of K and other organic fertilisers 1.6% of N, 3.7% of P and 1.2% of K. If we assume that for N only 50% of the excretion is plant available (rest: losses from housing/storage/spreading plus share of spread N that is not plant available), manure still accounted for more than half of N fertilization.

The importance of manure is also evident if we look at the total nutrient cycle. According to Spiess (2011, supplemented), who calculated a farm-gate balance for total Swiss agriculture according to the OSPAR methodology, the internal nutrient flow is far greater than the flow in external inputs. For 2010 inputs of N/P/K, respectively, were 51/4/22 kg ha^{-1} in mineral fertiliser, 48/9/13 kg ha^{-1} in imported feedstuffs, 3/1/2 kg ha^{-1} in recycled non-agricultural residues, 35/0/0 kg in biological fixation and 24/0/0 kg in atmospheric deposition. Exports were 35/7/13 kg ha^{-1} in livestock products and 10/1/3 kg ha^{-1} in crop products. At the same time the flow in locally grown plant feedstuffs was 131/22/145 kg ha^{-1} and in livestock excreta 140/22/151 kg ha^{-1}.

3.2 Current manure management practice

This section should provide an overview of current management practice for manure and other organic residues recycled in agriculture. Depending on the availability, this information can be based on statistics, survey results or expert judgement. If not mentioned otherwise, the data shown for Switzerland is from the 2010 survey on management practice performed by Kupper et al. (2015).

Share of different livestock categories: Using N as indicator, cattle contribute 79% to total livestock excretion (dairy cows 49%, heifers 16%, suckling cows and their calves 9%, other beef and veal production 5%), pigs 11% (fattening pigs 7%), poultry 3.7% (hens 1.7%, broilers 1.9%), equids 2.3% and small ruminants 3.5% (sheep 2.8%, goats 0.7%) (calculated from official livestock number statistics and excretion guide values).

Housing systems and share of liquid and solid manure: In 2010, 52% of the dairy cows were still kept in tied housing systems (heifers about 30%). 53% of the dairy cows produced only slurry and 46% slurry plus solid manure.
Nearly 18% of livestock excretion is on pasture. Of the N excretion in the housing area 72% is collected in the form of liquid manure and 28% as solid manure.

Manure storage: For new slurry storage facilities Swiss farms should have a storage capacity of at least 5 months in the valley and hill zones and 6 months in the mountain zones. Taking into account the average dilution of slurry of about 1 : 1 (parts excreta : parts water), this would for example mean 19-23 m3 per dairy cow (7000 kg milk yield, housing system producing slurry only) or 67-80 m3 for 100 fattening pig places. Before 2011, the minimum storage capacity was defined in more steps ranging from 3.0 to 6.0 months. Most farms comply with these requirements. In 2010, 67% of the slurry was stored with solid cover, 16% with perforated cover, 6% with natural crust and 11% without cover. The average depth of the slurry stores was 3.0 m.
The storage capacity for solid manure should be sufficient for at least six months. Solid manure has to be stored on a concrete edged surface that is drained to the slurry store.

Manure use and application: Over 90% of the manure is used on the farm on which it is produced; even in the area with the highest livestock density (Canton Luzern) it is 83%. Over 80% of the slurry and around 60% of the solid manure is used on grassland. About 44% of the slurry and 24% of the solid manure is used during the summer months of June to August.
The average dose of slurry per application is about 25 m^3. With the average dilution of 1 : 1.2 (parts undiluted excreta : water added from washing, milking, drained surface etc.) this would be 20-25 kg/ha N_{total} or 9-15 kg/ha P. About three fourths of the slurry is spread with splash plate and one fourth with trailing hose. About two thirds is spread with tankers and one third with umbilical systems. Around 20% of the slurry is intentionally spread in the evening after 6 p.m. to reduce ammonia losses. About one fourth of the solid manure is incorporated within one day (about 9% within 8 h) and 45% is not incorporated (mostly on grassland).

Feeding practice: The feeding practice has a large influence on livestock excretion and must therefore be considered when nutrient flows in manure are assessed. In Switzerland, representative survey data on ration composition is available for dairy cows and pigs.
In 2010, the share of dairy cows receiving silage during winter feeding was 59% and during summer feeding 40% (Menzi, this conference), respectively (silage is prohibited for farms producing milk for cheese making). During winter feeding, 48% of the dairy cows received hay, grass silage and maize silage, for 29% the roughage diet consisted of hay only. During summer feeding, the roughage ration consisted of grass only for 20% of the dairy cows, while all the rest received some additional roughage with lower protein content. The average amount of concentrate used per cow was 740 kg per cow per year, but there is a high variability between farms.
Over 95% of the dairy cows are grazed, about 80% during the whole summer feeding period. 55% of the cows are grazed for 5 to 12 hours per day, about 20% <5 h and 20% >12 h.
Around 70% of the pig feed sold in Switzerland has a reduced protein and P content (rPP; Spring and Bracher 2014). For grower/finisher feed the crude protein (CP) content is 10% lower and the P content is 22% lower in rPP than in standard feed. For all-round sow diets the difference is -8% and -22%, for CP and P respectively. Phase feeding is still not common (only about 10% of the fattening pigs).

3.3 Relevant framework conditions

Structural conditions: The most important structural indicators of Swiss agriculture are the relative small average farm size of 18.6 ha (2012) and that livestock production is a major branch for over 80% of the farms. Consequently, nearly all farms have some animals and forage (71% of UAA) and in the lowlands crop production (26% of UAA; "mixed farms").

Policy requirements: As condition for receiving direct payments all Swiss farms have to fulfill a relatively broad list of criteria in favor of ecology and animal welfare. For manure management the most relevant is the soil surface nutrient balance ("Suisse-Bilanz"; input in form of manure and mineral fertiliser minus requirements according to recommendations) for N and P. This balance aims at preventing nutrient losses and restricting the livestock density to the local potential. 60% (base value; reduction for solid manure, arable crops, grazing) of the N applied in form of manure and 100% of the P is counted as plant available. The maximum surplus tolerated is 10%. Thanks to these policy requirements, the use of mineral fertiliser decreased between 1990 and 2010 by approximately 20% for N and by over 70% for P (Spiess 2011, supplemented), while the amount in agricultural products increased by nearly 25% for N and over 50% for P. Thus, the surplus was reduced by 15%, 75% and 55% for N, P, and K, respectively.
Farm facilities like animal houses and manure stores have to fulfill detailed and rather restrictive standards (e.g. new slurry store must be covered, solid manure stores must be drained to slurry store) and are controlled by the authorities. Furthermore, incentive programs have been initiated at the regional level (e.g. low NH_3 emission application techniques) and newly at the national level.

3.4 Recommendations and management aids

Fertilization guidelines: The Federal Agricultural Research Station Agroscope publishes the Fertilization Guidelines (FG), which should contain all the basic information needed for sustainable and good practice in nutrient management. This document consisting of extensive tables and some explanatory text is revised every few years (2002, 2009, 2016). It is the standard document on which more application oriented extension documents and tools as well as policy implementation tools like the "Suisse-Bilanz" for the N and P balance are based. For manure the FG (and other documents derived from it) contain standard values for excretions of different livestock categories (N/P/K/Mg/Ca), the amount of liquid (without dilution) and solid manure, the content of liquid and solid manure produced per animal place per year, typical amounts of additional water getting into the slurry from washing, milking, drained surface around the buildings etc. Apart from these standard values the document contains recommendations for good fertilization practice respecting the environment.

Nutrient balance "Suisse-Bilanz": For the compulsory N and P balance "Suisse-Bilanz" detailed rules for the calculation procedure have been developed and are yearly updated. While the balance was originally filled in by hand or in an Excel template, it is today mainly calculated with special software. Farmers often rely on the support of experts from the extension services.

Policy implementation aids: In 2012 and 2013, the Federal Offices for Agriculture and Environment published new "Guidelines for environmental protection in agriculture" (www.bafu.admin.ch/gewaesserschutz/01308/10890/index.html?lang=en). So far the following modules have been published: 1) Rural installations, 2) Fertiliser elements and the use of fertilisers 3) Agricultural biogas plants, 4) Plant protection products in agriculture, 5) Soil. These documents, which were prepared by groups of dedicated experts, have no juristic power, but should summarize all the relevant restrictions and recommendations.

3.5 Historical development

Production systems: The biggest change in production systems could be observed between 1950 and 1980 and since 1990 (see chapter " Relevant framework conditions: policy requirements"). From 1950 to 1980 there was a rapid increase in the production intensity. For example, the average milk yield per dairy cow increased from 3000 to 5000 kg per cow. The increase in production intensity had strong effects on animal performance, feed rations, mineral fertiliser use and the appreciation of the nutrient value of manure. An impressive illustration of this is the dramatic increase in mineral fertiliser use. Between 1950 and 1980 the use of mineral N, P and K fertilisers increased by a factor of 8.1, 1.8 and 3.4, respectively. However, even at the peak of mineral N fertiliser use around 1990, the amount in livestock excretion, which was all recycled as manure, was more than double that in mineral fertiliser for N and about 150% for P. In 2010, manure was 240%, 480% and 640% of that in mineral fertilisers, for N, P and K respectively.

Livestock production and N excretion: Cattle production (especially dairy) has quite a long tradition in Switzerland while pig and poultry passed the level of local consumption only after the Second World War. To take into account not only the change in animal numbers but also changes in management and productivity, we can look at the development of N in livestock excreta which we tried to reconstruct for a historical view on NH3 emissions. Between 1900 and 2010 total N excretions increased by a factor of about 1.4, those of dairy cows by 1.2, other cattle 1.7, pigs 3.5, poultry 4.5, horses 0.5 (50% decrease), small ruminants 0.95. Comparing 1900 with the peak around 1980 it was for total N excretion 1.5, dairy cows 1.45, other cattle 1.65, pigs 6.2, poultry 3.3, horses 0.2 (80% decrease), small ruminants 0.6. The most drastic growth occurred between 1950 and 1980.

Livestock and manure management: The further back we look, the bigger the uncertainty about farm and manure management. However, livestock excreta was always fully used as fertiliser (at least since 1800) and we can assume that changes in manure management practice were not very big until about 1960. Farmers mostly collected both solid and liquid manure. After about 1960, slurry increasingly gained in importance in cattle and pig production.
With the rapid increase of fertiliser use after 1950 (especially N and K) the value of manure was hardly recognised by many farmers, although with about 60 kg N and 15 kg P the maximum average fertiliser use per hectare UAA was lower than in many other countries. This changed quite drastically with the new agricultural policy introduced in the 1990s (see chapter "Relevant framework conditions"). With the nutrient balance requirements farmers had to fully take into account the N and P value of the manure and they also increasingly started to systematically reduce NH_3 losses and to reduce N and P excretions of pigs and poultry by using phytase and pure amino acids in the diet.
Since 1990 there were also considerable shifts in livestock management, which especially had a strong influence on NH_3 emissions (Kupper et al. 2015). The most important were 1) the shift from tied to loose housing systems for cattle (loose housing for dairy cows 1990 6%, 2010 48%) and the introduction of exercise yards for dairy cattle, 2) the increasing use of silage (especially maize) for cattle (less milk used for cheese making, abolishment of ban on silage use for dairy cattle during summer feeding period), 3) the increasing importance of grazing because of incentives for animal friendly systems, 4) the introduction of multi-pen housing systems with outside access for pigs because of label programs for animal friendly systems. Evidently, the continuous increase in productivity of livestock systems (e.g. milk yield per cow 2010 7000 kg), the introduction (1976) and abandonment (2009) of milk quotas and the effect of free-trade agreements also were important drivers.

Ammonia emissions: In 2010, agriculture contributed 92% to the Swiss NH_3 emissions, of which livestock and manure management were responsible for 90% (Kupper et al. 2015). Between 1870 and 1983 agricultural NH_3 emissions increased by a factor 2.4 and then decreased again by 16% until 2010. This reduction was mainly due to declining livestock numbers, since the emissions per animal became bigger for most livestock categories.
In 2010, cattle, pigs and poultry contributed 78%, 15% and 3%, respectively, of NH_3 emissions from livestock and manure management. Housing, storage, application and grazing contributed 34%, 17%, 46% and 3%, respectively.

4. Conclusions and outlook

Manure clearly always was and most probably will be the most important fertiliser in Swiss agriculture. During the past 20 years, the awareness of farmers about the value of manure and good manure management practice has grown considerably as a consequence of policy incentives.
Most countries will probably not have such detailed and representative information on manure management as Switzerland. The template for the country manure management profile therefore has to be flexible. Nevertheless, a common structure and approach will be helpful to make such profiles comparable. We hope that this example can stimulate similar activities and a coordinated approach to provide a good knowledge base about manure management.

References

[1] Kupper, T., Bonjour C., Menzi H., (2015). Evolution of farm and manure management and their influence on ammonia emissions from agriculture in Switzerland between 1990 and 2010. Atmospheric Environment 103, 215-221
[2] Spiess, E., (2011): Nitrogen, phosphorus and potassium balances and cycles of Swiss agriculture from 1975 to 2008. Nutrient Cycling in Agroecosystems 91, 351-365.
[3] Spring, P., Bracher, A., (2014): Survey of current Swiss pig feeding practices and potential for ammonia emission reduction. Journal of Applied Animal Nutrition, e9. doi: 10.1017/jan.2014.3

TA-K_02 Organic residue management options in a Chinese peri-urban region with intensive animal husbandry and high nutrient loads

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1. Objectives

The “N and P costs” of food production in China almost doubled between 1980 and 2005 [1]. In the peri-urban region of Beijing with very high livestock densities, industrialized land-less animal operations are facing a large number of small-scale crop farmers. Shunyi District in particular has a very intensive animal production [2] with very high livestock densities amounting to 10.6 livestock units (LU) ha^{-1} arable land in 2007 ([3], calculated from [4]; 1 LU approx. 500 kg). High amounts of organic residues are improperly disposed of, causing severe environmental pollution. At the same time, the National Development and Reform Commission (NDRC) of China is seeking to vastly expand energy production from biogas. The aims of the research were to:

- estimate N and P inputs, outputs and balances of major cropping systems in a typical peri-urban area of China at a farmer’s field scale and a cropping system scale
- compare different manure treatment systems under nutrient, soil fertility and economic aspects
- elaborate recommendations for improving the current soil nutrient status.

2. Methodology

A large pig farm in the Shunyi District of Beijing with attached biogas and composting plants served as pilot farm for investigations 2008-2011. On this farm pig production is carried out on a dual scale. Large-(industrial-)scale: The “Centralized pig plant” has an annual stock of 12,000 breeding boars and 20,000 fattening pigs yr^{-1}, with less than 10 ha cropland area. Small-(household-)scale: Meat production is also carried out in so-called “Ecological Feeding Gardens” as part of the same farm, consisting of 160 households with a total cropland area of 25.33 ha and producing a total of about 25,000 fattening pigs yr^{-1}. Each of the small-scale farm households sells on average 140 fattening pigs yr^{-1} and has approx. 0.2 ha of cropland. [5]. The so-called *gan qing fen* system [6] is used as manure system on both the large and the small-scale farms, with pigs housed on solid floors and the solids and liquids in manure separated manually. For this study, three major investigations were carried out on the farm and in the wider surrounding area.

- A soil screening investigation covering 26 plots under the five major cropping systems of the area (winter wheat-summer maize double-crop rotation, Chinese cabbage (Brassica rapa, ssp. pekinensis)-maize (glutinous corn, with a higher amylopectin content) double-crop rotation, open-field vegetables, orchards and poplar plantations) was carried out from 2009 to 2011 [7]. In the Shunyi and Huairou Districts of Beijing, 19 and 7 farmers’ fields were selected, respectively. These included plots belonging to the “Centralized pig plant”, to the “Ecological Feeding Gardens” as well as to crop farms in the area surrounding the large farm of the study. In March 2009, 2010 and 2011, as well as at the end of each growing season after summer crop harvest in September/October 2009, 2010 and 2011, soil samples were taken with an auger in 0-200 cm, in six depth increments.
- In another area of Shunyi District, soil surface N balances (68 households, two-year period, 2008/2009 - 2009/2010 [3]) and P balances (65 households, one year period, 2008/2009 [8]) were calculated for the cropland. Three specific villages with cereals (maize and wheat double-crop rotation), orchards and open-field vegetables as the predominant cropping systems were selected, and smallholder households with an arable land area ranging from 0.2 to 1.5 ha per farm were monitored during interviews performed twice over the two-year period. The N and P balances were calculated separately on an annual and per hectare basis as well as on a farmer’s field and a cropping system scale [3; 8].
- In a life cycle assessment (LCA), the required cropland area for a sound disposal of biogas effluent from animal wastes was calculated based on crop demand [9]. Annual input and out-

put of different types of pigs of the “Centralized pig plant” was calculated on the basis of LU. The then “current” system (status quo) regarding manure treatment, with 75% of the solid fraction (pig feces) used in a biogas plant and 25% in a composting plant, without, and with land application of biogas effluent (Option 1) was compared to two alternative management systems: In Option 2 all the feces were used in an improved composting plant, while in Option 3 100% of the feces were fed to the biogas plant, in both cases with land application of biogas effluent.

3. Results and discussion

3.1 Soil nutrient status

Typical soil types in Shunyi and Huairou Districts of Beijing are classified as Eutric Cambisols (WRB) with relictic hydromorphic characteristics and a silt loam texture [7]. The soils were heavily over-fertilized. Residual mineral nitrogen (N_{min}: NO_3^--N + NH_4^+-N) contents in 0-200 cm soil profiles sampled at five different dates during 2009-2011 ranged from 412 ± 281 to 1299 ± 287 kg N ha^{-1}, showing no significant differences between plots of the five cropping systems and seasons [7]. N_{min} contents in topsoils (0-20 cm) were equally very high, corresponding to 10-20% of the amounts in the 0-200 cm profiles [7].

The mean available P content (Olsen P) on all the 26 plots monitored was 73 ± 55 mg kg^{-1} in 0-20 cm topsoils, with half of the 26 investigated plots showing “very high” (>50 mg kg^{-1} available P contents in topsoils (Fig. 1) [7]; classification according to [10]. On the 19 investigated plots in Shunyi District, mean contents of available P had increased almost tenfold from 1981 to 2009, from 7.3 to 67 mg kg^{-1}, respectively, and this increase was very likely mainly due to high farmyard manure (FYM) inputs [7]. On average, the critical limit for increased risk of P loss of 30 mg kg^{-1} Olsen P in soil (cited in [11]) was reached or exceeded in all five cropping systems in the study [7].

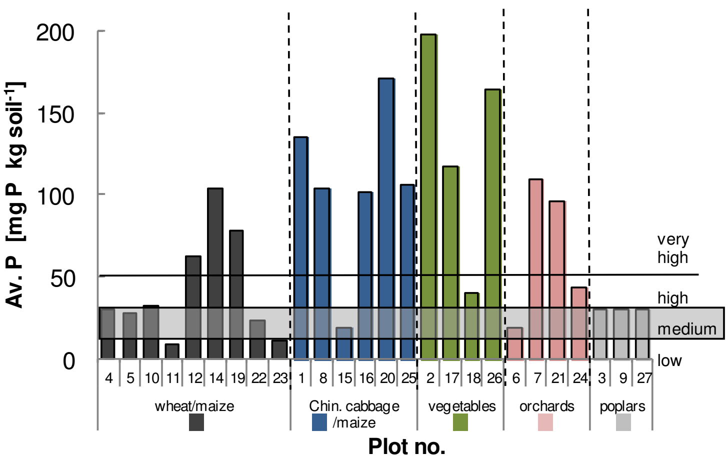

Figure 1: Available P contents in topsoils (0-20 cm) on 26 plots of the five major cropping systems in Shunyi and Huairou Districts of Beijing in March 2009 (classified according to [10]. Plot no. designates the number of each individual plot (from [7], modified).

3.2 Soil surface N and P balances

Extremely high amounts of FYM, ranging from 6-293 t DM ha^{-1} yr^{-1}, mean 45 ± 69 t DM ha^{-1} yr^{-1} were applied to most of the observed field plots receiving FYM (n=19) in 2008, resulting in the investigated soils being vastly over-supplied with nutrients from organic and inorganic sources [7].

The mean annual soil surface P balance surpluses calculated for the three cropping systems (Table 1) amounted to 83 kg P ha^{-1} yr^{-1} for cereal crops, 130 kg P ha^{-1} yr^{-1} for orchards and 492 kg P ha^{-1} yr^{-1} for the vegetable production system, the latter significantly higher (P<0.05) than the other two [8]. Based on these values, upscaling to the level of the whole of Shunyi District, the total annual surplus P amounts on cropland reached 4,300 tons (Table 1; Hou, 2009, unpublished). For the whole of Beijing Municipality, an annual P surplus of 15,689 t ha^{-1} was calculated for the year 2011 [2].

The soil surface N balances calculated for the cropping systems in the three specific villages of Shunyi District resulted in annual balance surpluses of 531, 519 and 1,548 kg N ha^{-1} yr^{-1} for the cereal crops, orchards and vegetable production systems, respectively [3]. Upscaling by the respective cropping areas of the whole of the District (see Table 1), the annual surplus N amounts reached 11,294 t N yr^{-1} for cereal, 634 t N yr^{-1} for orchard and 7,501 t N yr^{-1} for open field vegetable systems, totaling 19,429 t N yr^{-1} (Hou, 2009, unpublished). These high P and N surpluses pose a serious threat to the environment.

Fertilization in all three cropping systems, and to vegetables in particular, exceeded crop demand by far (Fig. 2). There was a great variation amongst the different farms in mineral fertilizer N and animal manure N input for the three cropping systems [3]. Farmyard manure requires higher labour for transport and handling and was therefore only applied to open-field vegetables and orchards from which higher economic benefit is achieved in this peri-urban area with high off-farm labour and income opportunities. However, the nutritive value of FYM was not properly taken into account since farmers also applied additional mineral N and P fertilizer (Fig. 2, Table 1).

Table 1: Soil surface P balance calculation for three major cropping systems based on a survey of 65 households in Shunyi District (2008/2009). Mean values and range (in brackets) (from [8]). Upscaled figures for Shunyi District: Hou (2009, unpublished). Sown area in Shunyi District from [4].

P balance items	**Cropping systems**		
	Cereals (n=21) kg P ha^{-1} yr^{-1}	**Orchards (n=23) kg P ha^{-1} yr^{-1}**	**Vegetables (n=21) kg P ha^{-1} yr^{-1}**
Inputs			
Mineral fertilizer	111.3 (28.5 - 271.1)	89.8 (0 - 350.4)	59.6 (0 – 98.2)
FYM	3.9 (0 - 35.6)	59.1 (0 – 304.5)	617.7 (177.1–1298.6)
Incorporated residues	13.1	2.6	0
Atmospheric P deposition	0.25	0.25	0.25
Total	128.6 (44.0 - 286.9)	151.8 (2.8 - 479.8)	677.6 (226.4–1362.7)
Outputs			
Crop product	45.9 (29.1 - 59.4)	22.3 (7.4 – 39.6)	185.7 (66.0 – 351.9)
P Balance			
Surplus/deficit	82.7 (-1.4 – 294.0)	129.5 (-10.7 – 464.8)	491.8 (111.3–1198.8)
Upscaling to Shunyi District			
Sown area Shunyi District (year 2009) [ha]	21262	1221	4847
Total annual P surplus Shunyi District [t yr^{-1}]	1758	158	2383

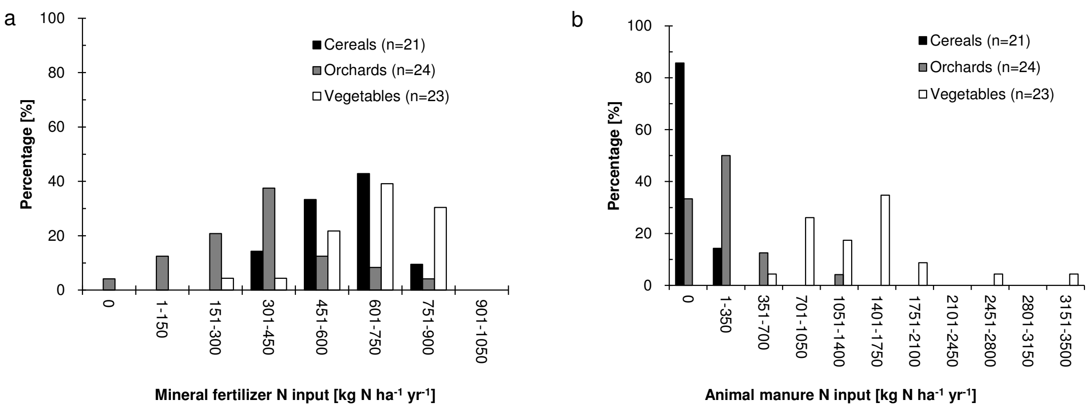

Figure 2: Frequency distribution (in percent) of number of surveyed farms based on different classifications of mineral fertilizer N input (a), animal manure N input (b) (mean value of the investigated two years) for the three cropping systems (after [3], modified).

3.3 Life Cycle Assessment (LCA)

In the year 2011, the centralized pig plant had a stock of 1,037 LU and produced 1,956 LU annually [9]. Only the P flows are presented here. The annual P input and output of the pig plant, the biogas plant and the composting plant for the "Current" manure management system without, and with (Option 1) land application of biogas effluent, as well as for Options 2 and 3 in 2011 is given in Table 2. Input and output flows of the pig plant, the biogas plant and the composting plant between the "Current" manure system and Option 1 were equal (Table 2), the difference consisting in the use of the biogas plant output. Under the "Current" manure management system, a large part of the liquid biogas plant effluent flowed into a nearby lagoon (with no insulation-liner) and was frequently discharged into a semi-dry riverbed, with very little proper land application [9]. Phosphorus losses during composting were assumed to be zero. The improved compost under Option 2 can be sold and thus offers the possibility of exporting surplus nutrients to other farms with nutrient demand. Thus in the LCA the cropland area required considered only biogas effluent.

Table 2: Annual P input and output of the centralized pig plant, biogas plant and composting plant of the Pilot Pig Farm (based on Life Cycle Inventory (LCI, year 2011) under the current manure management system without and with land application of effluent, as well as for two alternative options (adapted from [9 and] [12]). (Unit: tons P yr^{-1}).

Pig stables		Biogas plant		Composting plant	
Input	**Output**	**Input**	**Output**	**Input**	**Output**
Current manure management system without and with (=Option 1) land application of effluent (unit: t P yr^{-1})					
Feed: 23.64 t	Pigs to market: 1956 LU	Pig faeces: 9.89 t	Liquid effluent: 13.32 t	Pig faeces: 3.30 t	Compost: 3.99 t
	Dead pigs: 49 LU	Poultry manure: 1.16 t	Biogas sludge: 0.68 t	Biogas sludge: 0.68 t	
	Pig faeces: 13.19 t	Wastewater: 2.95 t		Sawdust: 0.02 t	
	Wastewater: 2.95 t				
Sum: 23.64 t		**14.0 t**	**14.0 t**	**4.00 t**	**3.99 t**
Option 2: All pig feces to improved composting plant (unit: t P yr^{-1}) with land application of effluent					
Feed: 23.64 t	Pigs to market: 1956 LU	Pig faeces: 0 t	Liquid effluent: 2.95 t	Pig faeces: 13.19 t	Compost: 19.57 t
	Dead pigs: 49 LU	Poultry manure: 0 t	Biogas sludge: 0 t	Corn stalks: 0.61 t	
	Pig faeces: 13.19 t	Wastewater: 2.95 t		T-Superphos. 5.76 t	
	Wastewater: 2.95 t				
Sum: 23.64 t		**2.95 t**	**2.95 t**	**19.56 t**	**19.57 t**
Option 3: All pig feces to biogas plant (unit: t P yr^{-1}) with land application of effluent					
Feed: 23.64 t	Pigs to market: 1956 LU	Pig faeces: 13.19 t	Liquid effluent: 16.14 t	Pig faeces: 0 t	Compost: 0 t
	Dead pigs: 49 LU	Poultry manure: 0 t	Biogas sludge: 0 t	Biogas sludge: 0 t	
	Pig faeces: 13.19 t	Wastewater: 2.95 t		Sawdust: 0 t	
	Wastewater: 2.95 t				
Sum: 23.64 t		**16.14 t**	**16.14 t**	**0.0 t**	**0.0 t**

[a] T-Superphos. = Triple-superphosphate

Biogas effluent should be applied onto the surrounding cropland area but strictly based on crop nutrient demand. A typical winter wheat-summer maize double-crop rotation in Shunyi District has an approximate nutrient demand per ha and year of 338 kg N, 56 kg P, 263 kg K and 50 kg Mg [7; 9] in a balanced system (optimum plant-available nutrient contents in soil). The resulting cropland area demand for a sustainable land application of biogas effluent for the "Current" system with land application (Option 1) would then be 238 ha yr^{-1} (with 14.3 L effluent m^{-2}, limited by P demand of crops), for Option 2 it would be 139 ha yr^{-1} (24.4 L m^{-2}, limited by N demand), while for Option 3, the demand would be 288 ha yr^{-1} (11.8 L effluent m^{-2}, limited by P demand) [9; 12]. However, as long as the cropland is over-supplied with nutrients (see 3.1), the land area necessary for a sustainable utilization of the wastewater nutrients would be 476 ha yr^{-1} at the "Current" manure management (Option 1), 168 ha yr^{-1} for Option 2, and 576 ha yr^{-1} for Option 3 (nutrient recommendations based on one half of the annual crop requirement during years 1-5) (Schuchardt, Luo, Heimann, 2012; unpublished). However, the centralized pig plant only cultivated about 10 ha. If all solids were used for composting (Option 2), it would be possible to transfer 87% of P, 29% of total N, 34% of K and 75% of magnesium (Mg) to compost for export out of the farm and region [9]. Outlying areas of the North China Plain have low soil organic matter (SOM), total N and low to medium available P and K contents. Transport of processed manure would lead to a better and faster build-up of SOM compared to returning crop straw alone in this sub-humid region.

4. Conclusion and outlook

- Differences in farming practices within and among cropping systems should be taken into account when calculating nutrient balances and designing strategies of integrated nutrient management on a regional scale [3].
- As an immediate measure, part of the excessive FYM should be processed (e.g., composted) and surpluses nutrients be transported out of the peri-urban region via marketable products [7].
- Governmental subsidies to compost production are in place in several Chinese provinces [7]. It is assumed that the market price for quality fertilizer products is likely to increase. Good composting procedures may become economically attractive without subsidies in the future.
- The question of whether to promote manure processing or whether to focus on energy production from biogas and subsequent nutrient removal from effluent via technological means can only be solved on a case by case basis in China.
- There is an urgent need for on-farm research on the use of biogas effluent in Chinese cropping systems, considering nutrient as well as emission (NH_3, N_2O, N leaching) aspects.
- In the medium term, a reduction of livestock densities, by moving part of the intensive, landless livestock operations to surrounding provinces closer to the cropland demand, is essential.

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References

[1] Ma, L., Velthof, G.L., Wang, F.H., Qin, W., Zhang, W.F., Liu, Z. Zhang, Y., Wei, J., Lesschen, J.P., Ma, W.Q., Oenema, O., Zhang, F.S. (2012): Nitrogen and phosphorus use efficiencies and losses in the food chain in China at regional scales in 1980 and 2005, Sci. Total Environ. 434, 51-61

[2] Jia, W., Yan, Z.J., Chadwick, D.R., Kang, L.Y., Duan, Z.Q., Bai, Z.H., Chen, Q. (2015): Integrating soil testing phosphorus into environmentally based manure management in peri-urban regions: A case study in the Beijing area, Agric. Ecosyst. Environ. 209 (Special Issue), 47-59.

[3] Hou, Y., Gao, Z.L., Heimann, L., Roelcke, M., Ma, W.Q., Nieder, R. (2012): Nitrogen balances of smallholder farms in major cropping systems in a peri-urban area of Beijing, China, Nutr. Cycl. Agroecosyst. 92, 347–36

[4] Shunyi District Statistical Yearbook (2008): Statistical Bureau of Shunyi District, Beijing *(in Chinese)*

[5] Hou, Y. (2011): Characteristics and regulatory approaches of nutrient flows in intensive „crop-livestock“ production systems of the peri-urban area of China. MSc Thesis. Agricultural University of Hebei, Baoding, P.R. China, 43 p. (*in Chinese with English abstract*)

[6] Ju, X.T., Zhang, F.S., Bao, X.M., Römheld, V., Roelcke, M. (2005): Utilization and management of organic wastes from agricultural origin in China: Past, present and perspectives, Science in China, Ser. C Life Sciences 48 Special issue: 965-979

[7] Heimann, L., Roelcke, M., Hou Y., Ostermann, A., Ma, W.Q., Nieder, R. (2015): Nutrients and pollutants in agricultural soils in the peri-urban region of Beijing: Status and recommendations, Agric. Ecosyst. Environ. 209 (Special Issue), 74-88.

[8] Hou, Y., Gao, Z.L., Ma, W.Q. (2011): Evaluation on the Phosphorus Balance in the Production Systems of Main Crops in the Suburb of Beijing — Case in Shunyi District, Journal of Anhui Agricultural Sciences 39, 14693-14695 (*in Chinese with English abstract*)

[9] Luo, Y.M., Stichnothe, H., Schuchardt, F., Li, G.X., Mendoza Huaitalla, R., Xu, W. (2014): Life cycle assessment of manure management and nutrient recycling from a Chinese pig farm, Waste Manage. Res. 32, 4-12

[10] Sun, Y.X., Guo, Y.S., Yu, S.Z., Jiang, Q.G., Cheng, L.L., Cui, Z.L., Chen, X.P., Jiang, R.F., Zhang, F.S. (2009): Establishing phosphorus and potassium fertilization recommendation index based on the '3414' field experiments, Plant Nutr. Fertil. Sci. 15, 197–203

[11] Xue, Q.Y., Shamsi, I.H., Sun, D.S., Ostermann, A., Zhang, Q.C., Zhang, Y.S., Lin, X.Y. (2013): Impact of manure application on forms and quantities of phosphorus in a Chinese Cambisol under different land use, J. Soils Sediments 13, 837-845

[12] Final report of BMBF “Recycling” Project (2012): Recycling organischer Reststoffe aus der Landwirtschaft und dem städtischen Bereich in China. FKZ 0330847. Schlussbericht (inhaltlicher Sachbericht) der Teilprojekte FKZ 0330847A-H. Birkenfeld and Braunschweig, Sept. 30, 2012, 372 pp. (*in English with German summaries*). http://edok01.tib.uni-hannover.de/edoks/e01fb15/823216497.pdf

TA-O_01 Development of a mobile app for manure management – 'The Farm Crap App'

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1. Objectives

To create a mobile application that helps farmers and growers put a nutritive and economic value on slurries and manures, and aid with manure management by calculating crop available nitrogen, phosphate and potash in different manures and slurries at different spreading rates based on the soil type, season and crop type. Allow farmers to visualise their applications in terms of nutrients via an image library. To aid record keeping for nutrient management plans.

2. Case study description

The Farm Crap App is a manure management mobile application designed to raise awareness amongst farmers and growers about the nutritive and economic values of slurries and manures, principally so they start to value it rather than see it as a waste product. The app can help with planning manure applications, avoiding over application and mitigating pollution risks.

The app has three functions: The first is a simple calculator (Figure 1) that determines the crop available Nitrogen (N), Phosphate (P_2O_5) and Potash (K_2O) in different slurries and manures at different spreading rates depending on the season, soil type and crop type. DEFRA's Fertiliser Manual, RB209 [1] was used to populate the calculator function of the app. RB209 provides industry derived estimates of mean nutrient contents of different manure types and the percentage of those nutrients that are crop available, depending on when and how the manure is applied. The app utilises the financial data from Manner NPK [2], assigning a monetary value to the manure applied.

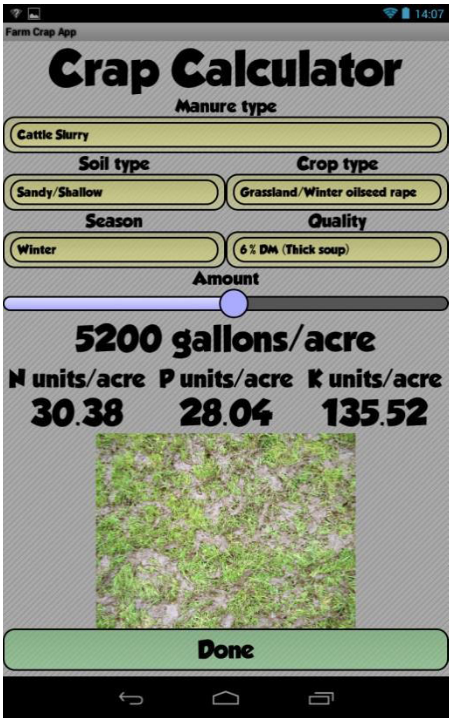

Figure 1. The apps calculator function and image library, in this screen shot the crop available NPK are expressed in units per acre, the user can select the measurement units at the field input page.

Table 1 represents a typical example of the data used to populate the calculator function of the app. In this example the total and readily available nutrient content on fresh weight basis for cattle slurry is shown.

Table 1: Total and readily available nutrient contents on a fresh weight basis for cattle slurry.

Dry matter (%ge)	Total N (kgN/m³)	Readily available N (kgN/m³)	Total P_2O_5 / m³	Availability	Total K_2O / m³	Availability
2	1.6	0.9	0.6	50	2.4	90
6	2.6	1.2	1.2	50	3.2	90
10	3.6	1.3	1.8	50	4.0	90

An example of the data used to determine available nitrogen on different soil types is presented in table 2. The data is seasonal and represents surface applied and soil incorporation six hours after application, figures in parenthesis are for winter grassland and oilseed rape cropping.

Table 2: Percentage of available nitrogen to next crop from cattle slurry surface application and incorporation by ploughing. (Figures in parenthesis are for winter grassland and oilseed rape.)

	Autumn (Aug –Oct, 450 mm rainfall to end March)		**Winter (Nov – Jan, 205mm rainfall to end March)**		**Spring (Feb – Apr)**	**Summer use on grassland**
	Sandy / shallow	Medium / heavy	Sandy / shallow	Medium / heavy	All soils	All soils
Cattle slurry – Surface applied						
2% DM	5 (10)	30 (35)	30	30	45	35
6% DM	5 (10)	25 (30)	25	25	35	25
10% DM	5 (10)	20 (25)	20	20	25	20
Soil incorporated 6 hours after application						
2% DM	5 (10)	35 (40)	25	35	50	N/A
6% DM	5 (10)	30 (35)	20	30	40	N/A
10% DM	5 (10)	25 (30)	15	25	30	N/A

The second unique function is the image library (Figure 1) which allows farmers to visually assess their application. This is thought to be a unique function that connects manure application rates to imagery and crop available nutrients, with the added practical benefit that it can be used in the field to make real time comparisons.

The third function is for record keeping (Figure 2). The user can set up their individual fields, recording name, size, soil and crop type and then add the details of the manure application. The user can select their desired units, and then export a csv (comma separated value) file containing all of the field records via email to the farm computer. These records can be kept, shared or incorporated into a nutrient management plan.

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P
1	Field name	Size	Units	Soil	Crop	Manure	Date	N	P	K	Amount	Amount	Total	Total Units	Quality	Season
2	Barn field	6	ha	Medium/Heavy	Grassland/Winter oilseed rape	Pig Slurry	20/9/2012	26.4	11	39.6	22	m3/ha	132	m3	2% DM (Thin soup)	Autumn
3	Barn field	6	ha	Medium/Heavy	Grassland/Winter oilseed rape	Pig Slurry	1/2/2013	15.84	15.6	30	12	m3/ha	72	m3	6% DM (Porridge)	Winter
4	Barn field	6	ha	Medium/Heavy	Grassland/Winter oilseed rape	Pig Slurry	7/4/2013	36	18	44	20	m3/ha	120	m3	4% DM (Thick soup)	Spring
5	Barn field	6	ha	Medium/Heavy	Grassland/Winter oilseed rape	Farmyard Manure	21/8/2013	10.2	32.3	122.4	17	tons/ha	102	tonnes	Surface applied or old	Summer
6	Barn field	6	ha	Medium/Heavy	Grassland/Winter oilseed rape	Poultry Litter	7/10/2013	72	120	129.6	8	tons/ha	48	tonnes	Broiler litter	Autumn
7																
8	Long meadow	5	ha	Sandy/Shallow	All crops	Farmyard Manure	24/11/2012	7.5	47.5	180	25	tons/ha	125	tonnes	Soil incorporated, fresh	Winter
9	Long meadow	5	ha	Sandy/Shallow	All crops	Cattle Slurry	4/2/2013	29.28	18.3	134.2	61	m3/ha	305	m3	2% DM (Thin soup)	Winter
10	Long meadow	5	ha	Sandy/Shallow	All crops	Cattle Slurry	7/4/2013	21.84	14.4	69.6	24	m3/ha	120	m3	6% DM (Thick soup)	Spring
11	Long meadow	5	ha	Sandy/Shallow	All crops	Pig Slurry	18/7/2013	46.2	14	50.4	28	m3/ha	140	m3	2% DM (Thin soup)	Summer
12	Long meadow	5	ha	Sandy/Shallow	All crops	Poultry Litter	24/9/2013	7.6	33.6	34.4	4	tons/ha	20	tonnes	Layer manure	Autumn
13																
14	Marshland field	11	ha	Medium/Heavy	Grassland/Winter oilseed rape	Cattle Slurry	24/7/2012	11.65	14.56	58.24	18	m3/ha	198	m3	10% DM (Porridge)	Summer
15	Marshland field	11	ha	Medium/Heavy	Grassland/Winter oilseed rape	Pig Slurry	24/2/2013	26.96	11.23	40.44	25	m3/ha	275	m3	2% DM (Thin soup)	Winter
16	Marshland field	11	ha	Medium/Heavy	Grassland/Winter oilseed rape	Poultry Litter	9/5/2013	53.37	88.96	96.07	3	tons/ha	33	tonnes	Broiler litter	Spring
17	Marshland field	11	ha	Medium/Heavy	Grassland/Winter oilseed rape	Cattle Slurry	31/8/2013	7.59	7.01	33.88	13	m3/ha	143	m3	6% DM (Thick soup)	Summer
18	Marshland field	11	ha	Medium/Heavy	Grassland/Winter oilseed rape	Poultry Litter	30/9/2013	90.14	132.84	136.01	8	tons/ha	88	tonnes	Layer manure	Autumn
19	Marshland field	11	ha	Medium/Heavy	Grassland/Winter oilseed rape	Farmyard Manure	24/10/2013	40.33	127.7	483.93	34	tons/ha	374	tonnes	Soil incorporated, fresh	Autumn
20																
21	Top field	7	ha	Sandy/Shallow	All crops	Poultry Litter	4/2/2013	14.25	25.2	25.8	3	tons/ha	21.9	tonnes	Layer manure	Winter
22	Top field	7	ha	Sandy/Shallow	All crops	Farmyard Manure	5/3/2013	21	66.5	252	35	tons/ha	256	tonnes	Surface applied or old	Spring
23	Top field	7	ha	Sandy/Shallow	All crops	Pig Slurry	28/7/2013	14.85	4.5	16.2	9	m3/ha	65.7	m3	2% DM (Thin soup)	Summer
24	Top field	7	ha	Sandy/Shallow	All crops	Cattle Slurry	14/10/2013	6.3	31.5	126	35	m3/ha	256	m3	10% DM (Porridge)	Autumn

Figure 2: Record keeping, in this screenshot the csv file of the field records opened in excel. The records can be emailed directly to the farm computer, or to any chosen email address.

2.1 The development

Engaging with the end user was an important consideration from the inception of the app. It was important that the app was genuinely useful to farmers and simple to use. A beta version of the farm crap app on the android platform was developed; it was presented to young farmer clubs in the south west, to farmer discussion groups, agricultural students, farm advisors, industry partners and the scientists behind RB209 and MANNER-NPK.
The feedback was overwhelmingly positive and constructive, resulting in modifications to the layout of the app, and the addition of more images. Importantly this feedback led to the incorporation of equivalent prices of nutrients based on current bagged fertiliser prices, i.e. the incorporating the relative monetary value of manure application. It was evident that to capture our target audience the app needed to be developed on iOS for Apple devices.

3. Observations

3.1 Benefits

The practical benefits to the farmer that are related to the safe recycling of organic residues are: increasing soil organic matter and enriching soil biology, leading to a healthier soil. In addition soil organic matter affects the physical and chemical properties of soils, improving structure, water holding capacity, root penetration and ultimately crop productivity.
As an agronomic tool the app has the potential to demonstrate to farmers the affect that their management in terms of application timings and amount has on the amount of nutrients available to their crop, thus allowing them to get the most benefit from their organic residues.
The app has the potential to reduce costs to farmers in terms of fertiliser applications and savings on fuel, with targeted manure applications. It also saves the farmer time in respect of nutrient management planning by calculating the crop available nutrients instantly and can be performed anywhere even in the field!
Interestingly, the app offers a novel vehicle for communicating changes in legislation. For example should there be a change to the close period in Nitrate Vulnerable Zones (NVZ), the app can be updated with this information by way of an informative ‘pop-up’, which will be notified to those devices on which the app has been installed.

4. Conclusion and future developments

For livestock farmers, good manure management planning can be a key component of farm profitability. Applying nutrients at recommended rates can potentially double yields of most crops, and getting this wrong risks yields, profits and breaching environmental and compliance regulations. The Farm Crap App allows farmers to access science in a very practical way allowing them to see direct benefits to their farm business from manure management.
Future developments include the input of the farmer’s own slurry and manure analysis data for N P_20_5 and K_20 values to allow a further level of farmer confidence. The ability to include the method of application to the calculation so that slurry injection and band spread slurry applications are factored into the available nutrients for the crop. The app has potential as an education tool, it is currently utilised in nutrient management planning lessons at Duchy College. The app has potential to be integrated into the teaching of agronomy across higher education establishments in the UK. A more ambitious development is to explore the ability of the app to import field and manure application data directly into whole farm management software programs.

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References

[1] Anon. (2010): Fertiliser Manual (RB209) access at: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/69469/rb209-fertiliser-manual-110412.pdf
[2] Anon. (2013): MANNER-NPK access at: http://www.planet4farmers.co.uk/Manner

TA-O_02 Farmers' reasons to accept bio-based fertilizers – A choice experiment in Flanders

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1. Objectives

Different regions in Western Europe are large importers of chemical fertilizer, while at the same time livestock intensive regions face disposal problems of the nutrients in animal manure. If processing transforms farm waste streams into biobased fertilizers, what are the reasons for farmers to accept these products? What is the willingness to pay for the different attributes of this fertilizer?

2. Methodology

A discrete choice experiment was conducted in Belgium to reveal the importance of key characteristics for new bio-based fertilizers to replace their current mineral fertilizer use. The method is in accordance with Lancaster's [1] attribute theory of value, which states that a good can be described as consisting of a bundle of characteristics at certain levels where utility is not derived from the good as such, rather from the specific attributes. The convenience of choice experiments is that respondents choose between analogous products depending on their attributes. In this case, respondents were asked to choose between two bio-based fertilizers (A or B) depending on their described characteristics, or none of them. The key attributes for a new alternative organic fertilizer were identified by experts, at stakeholder meetings and interviews. Six attributes were included in the choice experiment. The attributes considered as key in determining the preferences of farmers are presented in table 1.

Table 1: Attributes and attribute levels.

Attributes	Attribute levels
Price reduction	0 -20% -40% -60%
Form	Liquid Pasty Granulate Combination of liquid and solid
Volume	Same as current ×2 ×4 ×6
Uncertainty of N	0 25% 50% 75%
Organic carbon	Yes No
Hygienic	Yes No
Rate of nutrient release	Fast slow

A conditional logit was used to analyze the key attributes. The logit analysis is the simplest model for the analysis of choice experiments, assuming: the preference of individuals can be determined based on the selected attributes and homogeneous preference among respondents [2].

3. Results and discussion

Farm characteristics of the respondents are shown in Table 2. 43% of the respondents had an age above 50 years. Looking to the main economic activity of the farm, livestock farming was most represented in our sample (53,25 %), followed by the production of crops 32% and horticulture 14,8%. 92.5% of respondents used manure as a fertilizer.

Table 2: Farm characteristics.

Characteristics		
Surface		48.98 ha
Age	20-30	9.5%
	30-40	15.56%
	40-50	31.13%
	>50	43.71%
Main activity	Crop production	31.95%
	Livestock	53.25%
	Horticulture	14.79%
Farms using manure as fertilizer		92.46 %
Main responsible for the farm?		92.81%

Our results show that Flemish farmers have significant preferences for granular form of the fertilizer over pasty or liquid forms, a concentrated product, certainty in the nitrogen content, presence of organic carbon, hygienization of the fertilizer and fast release of the nutrients at lower price than the conventional ones. The coefficients for the attributes of the organic fertilizer are presented in Table 3.

Table 3: Flemish farmers' reference for attributes or bio based fertilizer.

Attributes	Coefficient		Standard Error	P-value
Price	0.014	***	0.003	0
Solid form	0.253	***	0.077	0.001
Liquid form	-0.127		0.13	0.329
Pasty form	0.086		0.097	0.381
Volume	-0.118	***	0.031	0.000
Uncertainty of N	-0.017	***	0.002	0.000
Organic carbon	0.385	***	0.075	0.000
Hygienic	0.285	***	0.067	0.000
Rate of nutrient release	0.155	**	0.069	0.025
Protest	-0.799	***	0.169	0.000

Note: ***, **, * ==> Significance at 1%, 5%, 10% level.

The uncertainty in nitrogen content is a well-known problem of the use of manure, because manure is not a homogenous product and nutrient contents can differ as a function of the feeding efficiency as well as storage and spreading conditions [3]. Nitrogen content is a critical component for crop yield, thus certainty in nitrogen content is key for crop fertilization. At the same time, an accurate reading of the nitrogen content and its available fraction would help prevent environmental problems such as the losses of nutrients to surface or ground water [4,5].

Due to the comparatively lower nutrient content, the amounts of manure required to fertilize the fields are comparatively larger than those of mineral fertilizers. Larger volumes of manure have also an effect on the spreading conditions. In some cases, spreading more might influence soil compaction, soil structure or crop damage. This will also increase the costs, which explains why our sample of farmers dislike larger volumes.

Preferences for more solid and concentrated fertilizer forms are also related to the reduction in transport costs and simplified ways to spread the product. This is indeed one of the main advantages often associated with mineral fertilizers. Additionally, solid forms of organic fertilizer have higher C/N ratios. This is in line with the expressed preference for organic carbon. The depletion of organic carbon is a common problem nowadays in agricultural land that has been cultivated intensively. An increase in the organic content contributes to a better soil structure and soil physical structure [6]. In this context, farmers usually have to apply manure or other bio-based products with organic carbon content, particularly in intensive crop rotations unable to replenish the organic matter requirements via crop residues.

A preference for the hygienization of the product was also found significant. Hygienization is applied to avoid the presence of pathogens and pests or herbs [7]. At the same time, it is a legal requirement for the transport of organic fertilizers across borders [8]. However, due to financial reasons only few of the available preventive treatment alternatives, such as composting or anaerobic treatment, are routinely applied [9].

The attribute rapid release of nutrients was found significant expressing a preference of farmers for the fast release of nutrients.

At the time of the analysis an extra “Protest” variable was added. In this study, the negative sign indicates that respondents had a tendency not to choose the Opt-out. This tendency may not be related to the attributes of the designed alternatives, but perhaps because they perceive that there is a current issue regarding the question of disposal and recycling nutrients.

The most commonly used chemical fertiliser in the EU is ammonium nitrate or calcium ammonium nitrate. These are fertilisers with readily available nutrients, with guaranteed nitrogen content in hygienic conditions, typically applied in concentrated granular form and without present organic carbon. It is interesting to see that farmers prefer a bio-based product that is quite similar to their current chemical fertilizer. A distinctive difference is that farmers prefer a product with organic carbon presence, mostly present in bio-based fertilizers.

Table 4 presents the willingness-to-pay (WTP) for the attributes in Flanders. The WTP measures are calculated as the ratio between the monetary attribute and the other significant attributes, keeping everything else constant. In this case, WTP measures are expressed in terms of a percentage of the price of mineral fertilizer.

Table 4: Estimated willingness to pay for different attributes.

	WTP	Standard Error
Solid form	17.86	0.165
Volume to spread (x2)	-8.57	5.088
Nitrogen Uncertainty (1%)	-1.21	2.035
Organic Carbon Presence	27.14	0.148
Hygienization	20	4.999
Fast release of Nutrients	10.71	-

According to the results, uncertainty about the nitrogen content or having to spread more organic fertilizer results in a reduction in WTP. Conversely, if the organic fertilizer is in granular solid form or in a hygienic condition or has organic carbon or a fast release of nutrients, farmers are willing to pay more for the organic fertilizer than for a bio-based fertilizer lacking these characteristics. The highest WTP is associated with the presence of organic carbon. In the Flemish context, the monetary values associated with granular solid forms in Belgium might be also understood in the light of the intensive livestock farming region in need for transport of manure (products). The WTP for hygienization of the products is easily explained with the pathogen constraints associated with the exchange of manure.

In this study, farmers were asked in a questionnaire whether they would be willing to replace their mineral fertilizer use by an alternative organic fertilizer with enhanced characteristics through processing. Nowadays, the availability of mineral fertilizers at relatively low prices has changed farmers’ view of manure. Despite its nutrient content, manure is often considered as a waste product. Martinez et al.[5] point out the need for new methods of waste management to protect the environment and, at the same time, reintroduce the recycling for manure management. Thus, processing of waste and manure streams into bio-based fertilizers with improved characteristics will increase the overall sustainability of the farming systems. At the same time, opportunities for nutrient exchange between different European regions would be created. Despite the existence of alternative management technologies, the adoption of these technologies has been low, stressing the importance of research on the topic. Previous research has focused on presenting new technologies [10–13], but there is a lack of studies examining the attitude and role of farmers in the adoption of these new technologies [14]. For policy makers it is of special interest to understand the factors explaining the adoption and diffusion of these technologies, as well as the acceptance of new products[15]. There is also a clear lack of knowledge in the literature about the preferences and WTP of farmers for the products of manure processing technologies. Due to the importance of this information for decision making and management in the agribusiness sector, retrieving these data is becoming increasingly important [16]. The main contribution of this research is that it gives insights in the preferences of farmers towards biobased fertilizers, and that it quantifies to what extend deviations from a preferred product should be compensated by lower prices for that product.

4. Conclusion and outlook

This paper elicited Flemish farmers' preferences for different attributes of a bio-based fertilizer, and quantifies to what extent deviations from a preferred product should be compensated by lower prices for that product. The results indicate that the demand for new bio-based fertilizer products could increase if the processing industry takes into account the preferences of farmers. Our results show that farmers Flanders have preferences for granular form of the fertilizer over pasty or liquid forms, a concentrated product, certainty in the nitrogen content, presence of organic carbon, hygienization of the fertilizer and fast release of the nutrients at lower price than the conventional ones. However, these preferences of the farmers are not always straightforward to apply for processers and markets due to technological constraints, legal and logistic issues. It is interesting to see that farmers prefer a biobased product that is quite similar to their current chemical fertilizer. One difference is that farmers would prefer a product with organic carbon presence.

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References

[1] Lancaster, K.J. (1966) A New Approach to Consumer Theory. *Journal of Political Economy*, 74, 132–157.
[2] Hensher, D.A., Rose, J.M. and Greene, W.H. (2005) Applied Choice Analysis. A Primer. Cambridge University Press, Cambridge, UK.
[3] Oenema, O., Oudendag, D. and Velthof, G.L. (2007) Nutrient losses from manure management in the European Union. *Livestock Science*, 112, 261–72.
[4] Schröder, J. (2005) Revisiting the agronomic benefits of manure: A correct assessment and exploitation of its fertilizer value spares the environment. *Bioresource Technology*, 96, 253–261.
[5] Martinez, J., Dabert, P., Barrington, S. and Burton, C. (2009) Livestock waste treatment systems for environmental quality, food safety, and sustainability. *Bioresource Technology*, Elsevier Ltd. 100, 5527–36.
[6] Wadman, W.P., Sluijsmans, C.M.J. and De La Lande Cremer, L.C.N. (1987) Value of animal manures: changes in perception. In: Van Der Meer HG, Unwin RJ, Van Dijk TA, and Ennik GC, editors. *Animal Manure on Grassland and Fodder Crops Fertilizer or Waste? SE - 1*, Springer Netherlands. p. 1–16.
[7] Alburquerque, J.A., de la Fuente, C., Ferrer-Costa, A., Carrasco, L., Cegarra, J., Abad, M. et al. (2012) Assessment of the fertiliser potential of digestates from farm and agroindustrial residues. *Biomass and Bioenergy*, 40, 181–9.
[8] European Parliament and Council (2002). Regulation (EC) no. 1774/2002 of the European Parliament and of the Council of 3 October 2002 laying down health rules concerning animal by-products not intended for human consumption. *Off J Eur Communities L*, 273, 1–95.
[9] Martens, W. and Böhm, R. (2009) Overview of the ability of different treatment methods for liquid and solid manure to inactivate pathogens. *Bioresource Technology*, Elsevier Ltd. 100, 5374–5378.
[10] Bernet, N. and Béline, F. (2009) Challenges and innovations on biological treatment of livestock effluents. *Bioresource Technology*, Elsevier Ltd. 100, 5431–5436.
[11] Bortone, G. (2009) Integrated anaerobic/aerobic biological treatment for intensive swine production. *Bioresource Technology*, Elsevier Ltd. 100, 5424–5430.
[12] Burton, C.H. (2007) The potential contribution of separation technologies to the management of livestock manure. *Livestock Science*, 112, 208–16.
[13] Flotats, X., Bonmatí, A., Fernández, B. and Magrí, A. (2009) Manure treatment technologies: On-farm versus centralized strategies. NE Spain as case study. *Bioresource Technology*, Elsevier Ltd. 100, 5519–26.
[14] Gebrezgabher, S. a., Meuwissen, M.P.M., Kruseman, G., Lakner, D. and Oude Lansink, A.G.J.M. (2015) Factors influencing adoption of manure separation technology in the Netherlands. *Journal of Environmental Management*, Elsevier Ltd. 150, 1–8.
[15] Vermeire, B. (2009) Absorptive capacity and innovation in the agrifood sector: role of regional networking and uncertainty.
[16] Lusk, J.L. and Hudson, D. (2004) Willingness-to-pay estimates and their relevance to agribusiness decision making. *Review of Agricultural Economics*, 26, 152–69.

TA-O_03 Agricultural reuse of the digestate from microalgae anaerobic digestion and co-digestion with sewage sludge

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1. Objectives

Microalge-based wastewater treatment systems have drawn attention to combine wastewater treatment and bioenergy production. The most straightforward process is anaerobic digestion, which produces biogas along with a digestate that may be reused as an organic fertilizer in agriculture in order to recover its mineral and organic constituents. However, the suitability of this microalgae-derived digestate for agricultural reuse has yet to be determined. The aim of this study was to characterize the digestate from microalgae anaerobic digestion and co-digestion with primary sludge in terms of nutrients, pathogens, emerging contaminants, heavy metals, dewaterability and potential phytoxicity.

2. Methodology

Microalgal biomass consisted of microalgae-bacteria consortia grown in a pilot raceway pond treating wastewater from a municipal sewer in Barcelona (Spain), while thickened primary sludge was collected in a municipal WWTP near Barcelona.

In order to improve microalgae biodegradability, algal biomass was thermally pretreated (75ºC, 10h) [1]. Besides the anaerobic co-digestion of pre-treated microalgal biomass with primary sludge (25%-75% VS, respectively) was also performed to enhance biogas production [2].

Anaerobic digestion was carried out in three lab-scale mesophilic reactors (1.5 L) using the following feedstock:

- Digester 1 (D1): Microalgal biomass;
- Digester 2 (D2): Co-digestion of pre-treated microalgal biomass with primary sludge;
- Digester 3 (D3): Thermal pre-treated microalgal biomass.

Influent and effluent (digestate) were characterized weekly (integrated samples) over a period of 11 weeks of stable operation. Samples of digestates were analyzed for pH, electric conductivity (EC), total solids (TS), volatile solids (VS), volatile fatty acids (VFA), chemical oxygen demand (COD), total organic carbon (TOC), total nitrogen (TN) total Kjeldhal nitrogen (TKN), ammonium nitrogen ($N\text{-}NH_4^+$), P, Ca^{2+}, Mg^{2+}, Na^+, K^+, pathogens, emerging contaminants (galaxolide, tonalide, triclosan, caffeine, triphenyl phosphate, methyl dihydrojasmonate, ibuprofen and naproxen) and heavy metals. Dewaterability was evaluated by means of the capillary suction time (CST) and potential phytoxicity by the Germination Index (GI) [3].

3. Results and discussion

3.1 Physical-chemical characterization and macronutrients

According to the results shown in Table 1, digestates presented appropriate pH and EC values for agricultural use [3]. Moreover, microalgae-derived digestates showed a high and fairly similar organic matter content (54%, 47% and 53% VS/TS for D1, D2 and D3 respectively). C/N ratios were also similar (ranging from 2.98 to 3.27), showing again that digestates could be considered as organic fertilizer containing high quantity of TN [4]. VFA concentrations at the end of the anaerobic digestion resulted to be far below the level supposed to cause phytotoxicity [5].

An important aspect for digestate management and final disposal is its dewaterbility, a parameter that describes the ability of biomass to release water. While microalgae digestates presented poor dewaterability (23 and 26 $s \cdot gTS^{-1} \cdot L$ for D1 and D3 respectively), after co-digestion the results were consistently improved (8 $s \cdot gTS^{-1} \cdot L$).

Table 1: Physical-chemical characterization of microalgae-derived digestates (mean ± SD, n=11).

		D1 (algae)	D2 (co-digestion)	D3 (preteated algae)
pH	pH unit	7.66 ± 0.07	7.30 ± 0.15	7.55 ± 0.08
EC	$mS \cdot cm^{-1}$	7.0 ± 0.7	6.0 ± 0.4	8.3 ± 0.3
TS	$g \cdot g^{-1}$,%	3.0 ± 0.6	2.9 ± 0.2	2.9 ± 0.4
VS	$g \cdot g^{-1}$,%	1.6 ± 0.3	1.4 ± 0.1	1.5 ± 0.2
VS/TS	%	54 ± 3	47 ± 1	53 ± 4
COD	$g \cdot L^{-1}$	45 ± 10	48 ± 6	42 ± 9
TOC	$g \cdot L^{-1}$	7.55	6.08	6.43
TN	$g \cdot L^{-1}$	2.38	1.86	2.16
C/N	-	3.17	3.27	2.98
VFA	$mgCOD\text{-}eq \cdot L^{-1}$	70 ± 31	92 ± 50	53 ± 24
CST	$s \cdot gTS^{-1} \cdot L$	23 ± 2	8 ± 1	26 ± 1

Macronutrients characterization results are shown in Figure 1. High contents of TKN were found in all digestates (from 1.7 to 2.4 $g \cdot kg^{-1}$); while the highest concentrations of $N\text{-}NH_4^+$ were found in the pretreated microalgae digestate, because of the thermal pretreatment that lead to the release of $N\text{-}NH_4^+$ from proteins [1]. Interesting quantities of P and K^+ were found in all the digestates; moreover, comparing these digestates with those derived from farm-byproducts [6,7], microalgae-derived digestates appeared to be richer in terms of phosphorus and potassium (from 1.6 to 1.9 g $P \cdot kg^{-1}$ and from 1.1 to 2.2 g $K^+ \cdot kg^{-1}$, respectively). Other macronutrients (Ca^{2+}, Mg^{2+} and Na^+) presented similar high concentrations in all the cases because of the composition of the wastewater treated. The content of Na^+ should be carefully considered while applying the digestates to the soils to avoid their salinization.

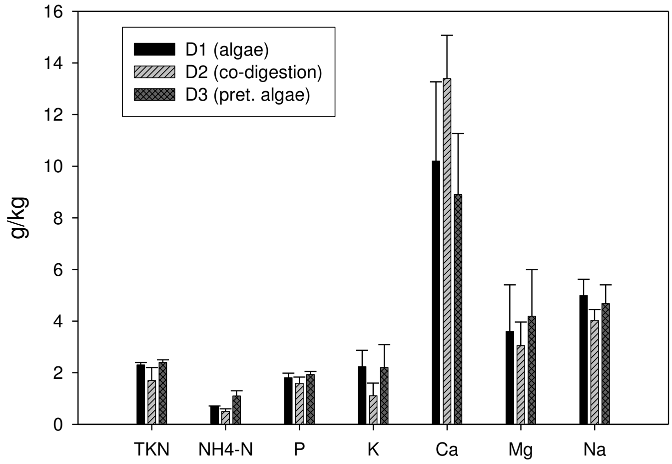

Figure 1: Concentrations of macronutrients in microalgae-derived digestates (mean+SD, n=11).

3.2 Heavy metals

With respect to heavy metals (Fig.2), the concentrations on a dry mass basis in the three tested digestates of all heavy metals were lower than the limit values established by the current European Directive[8], and also by EU draft published in 2003 [9]. Considering the common issues of Zn and Cu presented by farm-derived digestates, microalgae-derived digestates appear to be suitable for soil application. Attention should be paid when applying the co-digestion digestate because of its higher content in Zn originated from the primary sludge.

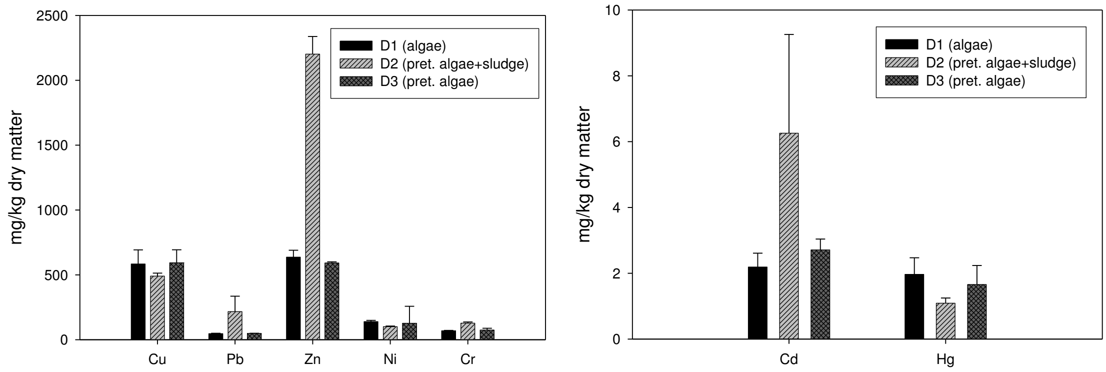

Figure 2: Concentration of heavy metals in microalgae-derived digestates (mean+SD, n=3).

3.3 Hygienisation

Regarding digestate hygienisation, low *Escherichia coli (E.coli)* presence was found in the digestates (Table 2), below the limit values proposed by the EU draft [9]., Moreover, the thermal pretreatment improved the hygenisation obtaining almost *E.coli* absence in the digestate.

Table 2: *E. coli* content in in microalgae-derived digestates (mean ± SD*;* n=8).

		D1 (algae)	D2 (co-digestion)	D3 (pret. algae)
E. Coli	ulog CFU·g^{-1}	1.6 ± 0.4	1.4 ± 1.3	0.1 ± 0.3

3.4 Emerging organic contaminants

The presence of some common emerging organic contaminants, mainly personal care products and pharmaceuticals, was evaluated for microalgae and co-digestion digestates. As can be observed in Table 3, the highest presence of these contaminants was detected in the co-digestion digestate, due to the primary sludge contribution. However, only five of the eight contaminants analyzed were found in the digestates. A correlation between their presence and the hydrophobicity behavior of the compound (kow) was observed. In spite of the low values obtained, these compounds may exert ecotoxicological effects at relatively low concentrations [10], so the presence of these compounds should be taken into account to avoid the problems associated.

Table 3: Emerging organic contaminants concentration in microalgae-derived digestates (mean values; n=4). Note: LOD=limit of detection

		D1 (algae)	D2 (co-digestion)
Galaxolide	ng g^{-1} dw	200	3500
Triclosan	ng g^{-1} dw	50	600
Tonalide	ng g^{-1} dw	400	1750
Triphenyl phosphate	ng g^{-1} dw	10	40
Methyl dihydrojasmonate	ng g^{-1} dw	25	25
Caffeine	ng g^{-1} dw	< LOD	< LOD
IIbuprofen	ng g^{-1} dw	< LOD	< LOD
Naproxen	ng g^{-1} dw	< LOD	< LOD

3.5 Potential phytotoxicity

Effects of the digestates tested in different concentrations (100%, 10%, 1% and 0.1%) on the germination of cress seeds are shown in Figure 3. Germination values are expressed as % of a control prepared with deionized water.

No germination was detected for all the pure digestates, probably due to the poor dewaterability that had not allowed for a good imbibition of the seeds or to some residual phytotoxicity effect. Conversely, positive results were found testing the lower concentration dilution. D1 and D3 gave similar trend of GI, showing the highest germination indexes for the 0.1% dilution (GI respectively of 109.9% and 97.3%). Concerning D2, there were no significant differences between dilutions of

10%, 1% and 0.1% (GI of 97.8%, 109.5% and 101.9% respectively), meaning that with co-digestion phytotoxicity effect was reduced. Nevertheless, all the digestates, conveniently diluted, can be considered to have plant nutrient or plant growth stimulant properties as reported in literature for manure digestates [6].

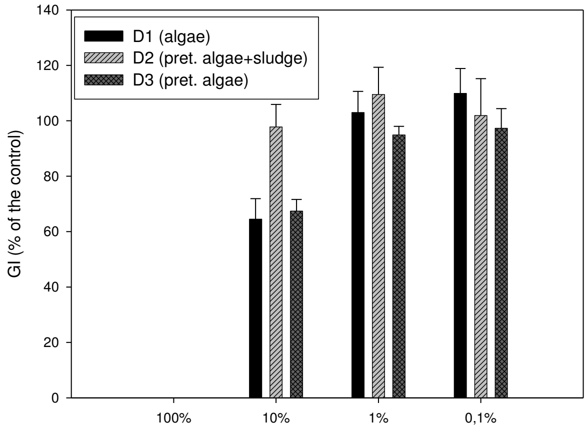

Figure 3: Effects of microalgae-derived digestates and dilutions on the germination index (GI) of cress (mean+SD).

4. Conclusion and outlook

Microalgae, pre-treated microalgae and in co-digestion with primary sludge digestates presented high organic matter and macronutrients content, especially organic and ammonia nitrogen, readily available for agricultural reuse as organic fertilizer. Dewatering proprieties were significantly improved after co-digestion. With the thermal pre-treatment, almost absence of *E.coli* was found in the digestate, while low *E.coli* content was observed in the other digestates. Only some emerging contaminants were detected in the digestates, mainly in the case of the co-digestion. When properly diluted, microalgae-derived digestates didn't show potential phytotoxicity; on the contrary, they showed interesting stimulant properties for plants. On the whole, co-digestion of thermal-pretreated microalgae and primary sludge appears as a promising alternative to improve microalgae digestion and move towards 0 waste generation in microalgae-based treatment systems. To further assess the agricultural value of the microalgae-derived digestates, in-depth assays will be conducted involving plant growth bioassays and the study of their biological stabilization.

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References

[1] Passos, F.; Ferrer I. (2014): Microalgae conversion to biogas: thermal pretreatment contribution on net energy production, Environmental Science & Technology, 48, 7171-7178
[2] Solé, M.; Passos, F.; Garfí, M.; Ferrer I. (2014): Co-digestion of microalgae and primary sludge as by-products from algal-based wastewater treatment systems. Proceeding of Biogas Science Conference, Vienna; October 2014.
[3] Zucconi F.; Pera A.; Forte, M.: Bertoldi, M. de. (1981): Evaluating toxicity of immature compost, Biocycle, 22(2), 3140-3142
[4] Smith K.A.; Metcalfe P.; Grylls J.; Jeffrey W.; Sinclair A. (2007): Nutrient value of digestate from farm-based biogas plants in Scotland, ADAS UK Ltd and SAC Commercial Ltd, 44p. Report for Scottish Executive Environment and Rural Affairs Department – ADA/009/06
[5] Chakwick D.R. (2007): Digestate as a fertilizer and environmental concerns, in: Anaerobic digestion stakeholder workshop, Session IV: building the market for digestate, UK: Exeter University; September 3-4
[6] Zucconi, F.; Monaco A.; Forte, M.; Bertoldi, M. de. (1985): Phytotoxins during the stabilization of organic matter, Composting of Agricoltural and other wastes, 73-86
[7] Alburquerque J.A.; Fuente, C. de la; Ferrer-Costa, A.; Carrasco, L.; Cegarra, J.; Abad, M.; Bernal, M.P. (2012): Assessment of the fertiliser potential of digestates from farm and agroindustrial residues, Biomass and Bioenergy, 40, 181-189
[8] Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture - Directive on spreading of sludge on land (2003)
[9] Proposal for an EU Directive on spreading of sludge on land (2003)
[10] Muñoz, I.; López-Doval, J.C.; Ricart, M.; Villagrasa, M.; Brix, R.; Geiszinger, A.; Ginebreda, A.; Guasch, H. ; de Alda, M.J.L.; Romaní, A.M.; Sabater, S.; Barceló, D. Bridging levels of pharmaceuticals in river water with biological community structure in the llobregat river basin (northeast Spain), Environ. Toxicol. Chem. 28 (2009) 2706–2714

TA-O_04 Strategies in sustainable utilization of manure in China – A review

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1. Intensive livestock industry development and environmental problem

Over the past 30 years the livestock industry in China has developed rapidly, and the percentage of livestock numbers reared in confined animal feeding operations (CAFOs) has significantly increased, with the value of 51%, 23%, 46%, 22%, 78% and 85% for pig, beef cattle, dairy, sheep, laying hen and broilers, respectively, in 2010[1]. We estimated that, the amount of livestock manure was 554 Tg DM, of which 277 Tg DM was generated from CAFOs, and 278 million Mg DM from traditional household systems in 2012. The livestock manure contained 14.4 Tg N, 5.0 Tg P and 7.7 Tg K (Jia Wei et al., unpublished). The development of a sustainable livestock industry in China is a huge challenge.

About 79% of manure N and 50% of manure P were lost to the environment in the management chain (barn-storage-treatment-application) in China in 2010, in which 33% of N and 95% P losses via discharge of manure to water bodies or landfilling (Bai Zhaohai et al., unpublished). The first national pollution census in China showed 57% of N and 67% of P were discharged from agriculture, in which 38% of N and 56% P discharge were associated with animal production source[2]. The NH_3, denitrification and N_2O emission generated by the livestock production were 6.4, 1.7 and 0.3 Tg in China in 2010, respectively (Bai Zhaohai et al., unpublished).

Recently, urbanization has resulted in livestock farms being located close to large cities. In these areas with high densities of intensive livestock rearing farms, manure loading rate to arable land are very high due to the lack of available land. In 2009, the average loading rates of manure, manure-N and -P to Chinese cultivated land were 6.3 Mg ha^{-1},105.9 kg ha^{-1} and 25.2 kg ha^{-1}, respectively[3]. In Netherlands, the application limit of manure P is 35 kg ha^{-1} [4]. Horticultural crops receive large quantities of manure in China[5]. For example, vegetable crops, which account for ca. 12% of the cropping area in China, receive 55% of manure P[6].

2. Separation technique and manure utilization in CAFOs

Separation technique of solid-liquid of animal manure in livestock farm is the key to select the method to recycle animal manure to farmland in China. Three systems are commonly used in China, including Shui Chong Fen ("use water to wash the manure"), Shui Pao Fen ("manure dunk in water") and Gan Qing Fen ("cleaning the manure dryly")[7-8]. The Shui Chong Fen system is often used in cattle farms, and solid-liquid separation is based on extrusion. The Gan Qing Fen method is wildly accepted by small pig farm. However, because of labor resource constraints, large pig farms usually adopt the Shui Chong Fen system.

Different techniques in solid-liquid separation mean different N and P contents in different animal waste fractions. Using the Gan Qing Fen method results in ca. 70% less total P content in the liquid fraction compared to that from the Shui Chong Fen system. Furthermore, the average pig daily water consumption is only 10-15 L $head^{-1}$, which is only 30-50% of that in the Shui Chong Fen and Shui Pao Fen systems. The Gan Qing Fen method not only reduce the content of various pollutants in wastewater (Table 1), but also obtain the solid waste with a high nutrient content. However, ca. 38%, 38% and 43% of N, P and K from manure is still lost during the storage stage of the manure management chain[3].

Table 1: Nutrient concentrations in different wastewater from livestock farms that adopt different in-house manure collection strategies[9]

Type	Manure collection strategies	NH_4-N (mg L^{-1})	Total N (mg L^{-1})	Total P (mg L^{-1})	pH
Pig	Shui Chong Fen	590	805	127	6.30-7.50
	Gan Qing Fen	261	370	43.5	
Beef Cattle	Gan Qing Fen	22.1	41.1	5.33	
Dairy Cow	Gan Qing Fen	51	67.8	18.6	7.10-7.51
Laying Hens	Shui Chong Fen	261	342	31.4	6.53-8.49
Duck	Gan Qing Fen	1.85	4.7	0.14	7.39

3. Manure and slurry processing techniques

In China, most solid manure (or solid fraction of manure) are used on farmland, but the utilization of liquid manure (slurry, digestate and liquid fraction of separated manure) are limited due to small household farms with little mechanisation. In most developed countries, slurry or solid manure are generally applied directly to farmland without composting or anaerobic fermentation[3,10-11].

3.1 Aerobic composting

Aerobic composting is the main form of solid manure treatment in China, including traditional and commercial aerobic composting[12]. Windrow, in-vessel, reactor and trough types composting strategies are widely used in China, especially groove type composting due to land limitation[3]. Manure heaps are turned and combined with additional aeration to accelerate the composting process, although only 38% of compost enterprises currently use forced aeration. It is estimated that ca. 31% of N, 11% of P and 19% of K in manure is lost during composting. NH_3 volatilisation represents the main form of N loss (Table 2).

Several methods have been commonly used to reduce N loss during composting. These include: (1) Choosing the most appropriate process condition; (2) Addition of Conditioner during composting; (3) Controlling the moisture of feedstock; (4) Covering in a shed, and so on. Recently we have found that addition of woody peat (10% w/w) to composting manure can reduce NH_3 emissions by 80%.

Commercial added-value products, i.e. ordinary organic fertilizers and bio-fortified organic fertilizers have been promoted by the composting industry in China. Commercial composting production is subsidized, for example subsidies of 200 yuan Mg^{-1} (23.5 Euro Mg^{-1}) were used for composting in 2012. Composting is the most appropriate way to reduce and redistribute manure N and P. After composting the mass of the composting feedstock has been reduced by 50%-80%[13], thus reducing transport costs and increasing the value of the product [14].

Table 2: Nitrogen losses, NH_3 and N_2O emissions during manure composting [15-29]

Type	Region	TN Losses (%)				NH_3 Losses (%)				N_2O Losses (%)			
		n	Mean	Max	Min	n	Mean	Max	Min	n	Mean	Max	Min
Pig Manure	Others	27	50.0	69.2	1.0	26	57.7	96.9	18.5	21	9.7	74.6	0.68
	China	100	24.2	78.0	6.3	100	66.5	64.2	2.1	66	6.9	78.0	6.4
Cow Manure	Others	36	54.6	79.0	22.3	5	38.5	55.4	21.6	22	1.8	6.1	0.001
	China	1	42.0	42.0	42.0	18	52.6	16.4	7.5	1	5.0	42.0	-
Chicken Manure	Others	12	8.7	24.6	7.60	18	12.64	16.4	7.6	-	-	-	-
	China	6	31.5	60.2	37.7	5	28.3	60.2	37.7	-	-	-	-

n: number of samples.

3.2 Anaerobic digestion

Anaerobic fermentation technology is generally more feasible for large-scale livestock farms in China. In 2010, there were ca. 5000 large and medium-sized biogas plants in China[30], and annual production of anaerobic digestate was >1.3 x 10^9 Mg[31]. However, discharge of untreated anaerobic digestate still occurs with severe effects on the environment. Digestate solid material is typically composted or applied direct to farmland[32]. Digestate liquid material is often used as a liquid fertilizer on some livestock farms, and is applied by drip irrigation, furrow irrigation and surface broadcasting. Improper application results in high losses of NH_3 volatilization[33-34], greenhouse gas emission sand N leaching[35-36], and greatly reduces the nitrogen use efficiency of digestate. In landless livestock farms, digestate needs to be treated to reach the standard (GB8978-1996) before discharge into watercourses.

3.3 Slurry and Wastewater treatment

At present, the utilization of the slurry and other liquid effluents are challenging for “landless” livestock systems, because of the high costs of processing to minimise pollution of watercourses. Most farmers have little enthusiasm to apply biogas digestate to land because of difficulties in transportation[37].
Chinese scientists are developing several engineering and ecological solutions to process the slurry and digestate, including physicochemical methods, biological-chemical methods, use of natural filtration beds and combined anaerobic-aerobic treatment strategies. Natural pretreatment can reduce the concentrations of pollutants with high organic matter contents and increase the biochemical degradability of wastewater with microorganism[38-39]. Then physicochemical deep treatment can further remove or recover the N and P nutrient in wastewater [40-41].

4. Land application of manure

There is no limitation to control manure and slurry use on farmland in China. In the USA and Europe, manure applications are commonly based on the manure P and/or N content, as there is a recognition of the different N:P ratios of manures and crops. A P-based manure application strategy would limit manure application rates significantly, and possibly require additional N fertilizer applications to satisfy crop demand.
However, the availability of N, P and K in manure includes fast-phase and slow-phase release. About 40-60% of N, 80-90% of P is available to crops in short-term, which reflects the manure application on the equivalent of chemical fertilizer in the same year. It is possible to predict the short-term availability through total N, ammonium N and the C:N ratio of the manure [42]. Long-term (slow-phase) availability reflects organic N release via mineralization following manure successive manure applications. This increases the long-term fertilizer equivalent value of manure to 40%-70%. It is necessary to understand mineralization rates of organic N to determine N fertilizer equivalent of manure so that using optimum manure rate to replace chemical N.

5. Perspective

Large-scale farming systems promoted by land consolidation and pressures to improve resource use efficiencies will require new strategies for manure management and land application. Farmers with large-scale planting areas could pay more attention to input-output efficiencies, and opt to utilize manure nutrients instead of chemical fertilizers to improve production efficiencies. Manure management in China should utilize technology with low cost according to local conditions. Production of the value-added commercial organic fertilizer product is a successful model for recycling nutrients from regions of intensive livestock production to other areas where organic matter and nutrients are required. This results in a recycling of nutrients at the broader/regional scale and reduces manure nutrient loads in areas of intensive livestock farms. Aerobic composting will remain an important treatment of manure in China for many years. However, technical innovation is urgently needed to a) reduce N loss from large composting factories and centralized anaerobic digestion plants, and b) sustainable utilize liquid manure (slurry, digestate and the liquid fraction of separated manure). Recently, the Chinese government has enhanced the environmental legal enforcement for livestock waste management, such as the Regulation on the Prevention and Treatment of Livestock Farm Pollution of 2014, New Environmental Protection Law of 2015, and the Water Pollution Control Action Plan of 2015. Legislation, guidance and process innovation area all needed to improve the sustainable use of manures in China.

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References

[1] Ministry of Agriculture of the People's Republic of China (MOA). (2011): The 12th five-year plan of national animal husbandry development of people's republic of China (2011–2015) , Beijing.

[2] Ministry of Environmental Protection of China (MEP), National Bureau of Statistics of China (NBS), Ministry of Agriculture of China (MOA), 2010. The first national survey of pollution sources Bulletin. Available at: http://www.zhb.gov.cn/gzfw/ xzzx/wdxz/201002/P020100225536025177826pdf (accessed on 03.06.15).

[3] Jia, W. (2014): Studies on the evaluation of nutrient resources derived from manure and optimized utilization in arable land of china, PhD Thesis: China Agricultural University, Beijing.

[4] Henkens, P., Van Keulen, H. (2001): Mineral policy in the Netherlands and nitrate policy within the European Community, NJAS-Wageningen Journal of Life Sciences, 49 (2), 117-134.

[5] Yan, Z.J., Liu, P.P., Li Y.H. *et al.* (2013): Phosphorus in China's intensive vegetable production systems: overfertilization, soil Enrichment, and environmental Implications. Journal of Environmental Quality, 42 (4), 982-989.

[6] Yan, Z.J. (2015): Effects of manure application on the form and mobility of soil phosphorus in vegetable greenhouse, PhD Thesis: China Agricultural University, Beijing.

[7] Schuchardt, F., Jiang, T., Li, G.X. *et al.* (2011): Pig Manure Systems in Germany and China and the Impact on Nutrient Flow. Journal of Agricultural Science and Technology,1, 858-865.

[8] Zhang, D.M. (2013): MAP precipitation recycling of N and P from swine wastewater and its impacts on traditional anaerobic and aerobic processe, PhD Thesis: Zhejiang University, Hangzhou.

[9] Minstry of Environmental Protection of the People's Republic of China (MEPC). (2011): Technical guidelines for controling livestock farm pollution (HJ-BAT-10). Beijing: China Standard Press.

[10] Lemaire, G., Franzluebbers, A., de Faccio Carvalho, P.C. *et al.* (2013): Integrated crop-livestock systems: Strategies to achieve synergy between agricultural production and environmental quality. Agriculture Ecosystems and Environment, 190, 4-8.

[11] Nicholson, F.A., Bhogal, A.D., Chadwick, D. *et al.* (2013): An enhanced software tool to support better use of manure nutrients: MANNER-NPK. Soil Use and Management, 29(4), 473-484.

[12] Tian, N.N., Wang, K.J. (2012): Livestock farm manure treatment technology, China Water & Wastewater, 18(2), 71-73.

[13] Bernal, M., Alburquerque, J., Moral, R. (2009): Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresource Technology, 100 (22), 5444-5453.

[14] Zhan, H.L., Yan, C.Y., Yang, Z. (2010): Research on the development of agricultural biomass energy industry. Chinese Agricultural Science Bulletin, 26(23): 397-402.

[15] Zhao, Q., Zhang, M.Q. (2008): Study on the nitrogen transforming regulation during the pig dung compost, Heilong jiang Agricultural Sciences, (2), 58-60

[16] He, Q. (2004): Studies on the nitrogen transform and reducing nitorgen loss during composting, Master Thesis: China Agricultural University, Beijing.

[17] Wei, Z.Q., Li, J.J. (2011): Nitrogen loss in chicken manure compost through runoff and leaching, Journal of Soil and Water Conservation, 25 (2), 44-47.

[18] Jiang, T. (2011): Composting process mechanism and emissions of greenhouse gases Research, PhD Thesis: China Agricultural University, Beijing.

[19] Shi, C.M., Wang, J.H., Li, G.X. *et al.* (2011): Control of different chemical additives on nitrogen loss during composting of pig manure, Journal of Agro-Environment Science, 30(5),1001-1006

[20] Hu, M.H, Liu, Q., Chen, X.Y. *et al.* (2009): Effects of two calcium compounds on deodorization and nitrogen retention during pig manure and rice straw composting, Hunan Agricultural Sciences, (7), 51-54.

[21] Hu, M.H, Liu, Q., Rong, X.M. *et al.* (2008): Effect of physical conditioner on deodorization and nitrogen retention during pig manure composting, Hunan Agricultural Sciences, (3), 85-89.

[22] Zheng, J.X., Wei, Y.S. (2011): Nutrients conservation of N & P and greenhouse gas reduction of N_2O emission during swine manure composting. Environmental Science, 32(7), 2047-2055.

[23] Amon, B., Amon, T., Boxberger, J. *et al.* (2001): Emissions of NH_3, N_2O and CH_4 from dairy cows housed in a farmyard manure tying stall (housing, manure storage, manure spreading). Nutrient Cycling in Agroecosystems, 60,103-113.

[24] Fukumoto, Y., Suzuki, K., Kuroda, K. *et al.* (2011): Effects of struvite formation and nitratation promotion on nitrogenous emissions such as NH_3, N_2O and NO during swine manure composting. Bioresoure Technology, 102,1468-1474.

[25] Liang,Y., Leonard, J.J., Feddes, J.J.R. *et al.* (2006): Influence of carbon and buffer amendment on ammonia volatilization in composting. Bioresource Technology, 97 (2006), 748–761

[26] Tran, M.T., VU, T.K.V., Sommer, S.G. *et al.* (2011): Nitrogen turnover and loss during storage of slurry and composting of solid manure under typical Vietnamese farming conditions. The Journal of Agricultural Science, 149(3), 285-296.

[27] Fukumoto, Y., Osada, T., Hanajima, D. *et al.* (2003): Patterns and quantities of NH_3, N_2O and CH_4 emissions during swine manure composting without forced aeration-effect of compost pile scale, Bioresource Technology, 89,109-114.

[28] Mahimairaja, S., Bolan, N.S., Hedley, M.J. *et al.* (1994): Losses and transformation of nitrogen during composting of poultry manure with different amendments: an incubation experiment, Bioresource Tethnology, 47, 265-273.

[29] Michel Jr, F.C., Pecchia, J.A., Rigot, J. *et al.* (2004): Mass and nutrient losses during the composting of dairy manure amended with sawdust or straw, Compost Science and Utilization, 12 (4), 323-334.[30] Lu, J.B, Zhu, L., Hu, G.L. *et al.* (2010): Integrating animal manure-based bioenergy production with invasive species control: A case study at Tongren Pig Farm in China. Biomass and Bioenergy, 34(6): 821-827.

[31] Marcato, C.E., Pinelli, E. Cecchi, M. *et al.* (2009): Bioavailability of Cu and Zn in raw and anaerobically digested pig slurry[J]. Ecotoxicology and Environmental Safety, 72(5): 1538-1544.

[32] Jin, H.M., Chang, Z.Z., Ye, X.M. *et al.* (2011): Physical and chemical characteristics of anaerobically digested slurry from large-scale biogas project in Jiangsu Province. Transactions of the Chinese Society of Agricultural Engineering, 27(1), 291- 296.

[33] Martines, A .M., Nogueira, M.A., Santos, C.A. *et al.* (2010): Ammonia volatilization in soil treated with tannery sludge. Bioresource Technology, 101(12), 4690-4696.
[34] Jin, H.M., Chang, Z.Z., Guo, D.J. *et al.* (2012): Effect of topdressing with digested pig slurry on ammonia volatilization in vegetable fields, Acta Pedologica Sinica, 49(1), 86-95.
[35] Meijide, A., Díez, J.A., Sanchez-Martin, L. *et al.* (2007): Nitrogen oxide emissions from an irrigated maize crop amended with treated pig slurries and composts in a Mediterranean climate. Agriculture, Ecosystems and Environment, 121(4), 383-394.
[36] Wang, Y.X., Su, X., Tang, Y.F. *et al.* (2008): Status analysis and countermeasures of large and medium scale biogas plants in Beijing rural areas. Transactions of the Chinese Society of Agricultural Engineering, 24(10), 291-295.
[37] Qu, M.S., Chen, B.H. *et al.* (2012): Beijing comprehensive agricultural biogas fertilizer technology. China Agricultural Technology Extension, 28(5), 43-45.
[38] Li, X.M. (2009): Researeh on the teehnology of natural biological purity and lime floceulation treatment with sewage on farms. Master Thesis: Si Chuan Agriculture University, Sichuan.
[39] Chen, P.Z. (2013): Research on Infiltration filter and the artificial fast filtration system coupling treatment the domestic sewage. PhD Thesis: China Agriculture University, Beijing.
[40] Song, Y.H., Qian, F., Xiang, L.C. (2011): Research on phosphorus removal from swine wastewater by precipitation of calcium phosphate. Environmental Science, 32(6),1679-1686.
[41] Wang, G.W., Qiu, L.P., Zhang, S.B. (2010): Development on Phosphorus Removal and Recovery from Wastewater. Technology of Water Treatment, 3,17-22.
[42] Gutser, R., Ebertseder, T., Weber, A. *et al.* (2005): Short-term and residual availability of nitrogen after long-term application of organic fertilizers on arable land. Journal of Plant Nutrition and Soil Science, 68(4), 439-446.

TA-O_05 Volunteers to translate waste prevention policy to citizens

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1. Objectives

Flanders (northern part of Belgium) is one of the best students in the field of separate collection of waste. This also counts for the organic fraction. A large amount of organic residues are processed by the people, at home or in their gardens. The Flemish government wants to further increase the number of home processors and the quality of home processing of these organic materials. To put this into practice, Flanders follows a very specific and unique approach.

2. Methodology

In Flanders, Vlaco npo supports and implements the policy of biowaste. Vlaco is a membership organisation with representation of both the Flemish government (OVAM and intermunicipal waste associations) and the private sector (private waste treatment companies). All the Vlaco activities support the sustainable material cycle of biowaste.
The 'Biocycling at home' unit of Vlaco focuses on raising an environmental awareness concerning bio-organic materials. All individuals are to be sensitized and convinced in a different way. On the longer term 'Biocycling' (or 'Kringlopen', a verb in dutch) must evolve in a 'way of life'. A 2-fold awareness approach has proven its success. The first approach - the most specific one - focusses on volunteering, enthusing and engaging. The second focusses on the content of the message and educating.

First approach
Around the year 2000, home composting was hot; it entirely fitted within the former Waste Decree. The main issues we distinguish in that 'Home Composting' scheme were composting itself, and the use of that self-made compost.
As the Waste Decree was interpreted more broadly, through the years, the 'Home Composting' scheme evolved to a 'Closed Loop Gardening' scheme (± 2003). From that moment on, besides composting, also other methods were recommended to process organic residues at home: f.i. lawn maintenance, processing of branches, perennials and how you can reduce organic waste by using these perennials instead of grass, chicken keeping ….
As the Waste Decree was replaced by the Materials decree (2012) solving the waste problem was no longer the central theme, but the using of waste as a basic material within a closed loop (or biocycle) became the main message from that moment on. Also Vlaco as initiator, facilitator and supervisor of the policy on organic biological waste evolves in this direction. In 2012 the 'BioCycle at Home' scheme was initiated. A new theme that was added to the responsibilities of Vlaco was the communication about food losses and how to prevent them. From that moment on also a lot of secondary, more specific themes came into the picture.

Currently, the Vlaco-unit 'Biocycling at Home' has trained several thousands of volunteers. These volunteers (so called Master Composters, or Biocycle Volunteers as we call them now) assist the Municipality by promoting the 'Biocycle at Home'. They communicate about seven different techniques to achieve the biocycle of:

- Food waste
- Lawn
- Prunings
- Home composting
- Compost use
- Chicken keeping
- Perennials

Vlaco trains the Master Composters. About 40 teachers are available to regularly train these volunteers and to update them. In total 4000 Master Composters have been trained the last 20

years. For the moment 2700 of those Master Composters are still active (which is about 1 per 2000 inhabitants).
They are volunteers that assist municipalities and the so called Intermunicipal Waste Associations in promoting the BioCycle at Home, in their own municipality or city.
Master Composters (or biocycle volunteers as we call them now) are volunteers in a team supported by the environmental, sustainability or green officer at the municipality.
The reason why Master Composters are being 'used' in communication (and not just folders or online information), and the cause of their ascertainable success, is because Master Composters have a better credibility compared to the 'official message', they stand with both feet in practice (that's why people believe them), they use their own words and work in their own village, and they can demonstrate things.

Second approach
Vlaco also approaches the public directly. This includes:

- organizing courses (about the preventing and processing of organic remains),
- (co-)organizing campaigns and events (Closed Loop Weekend, Closed Loop Festival, Floralies 2016 …),
- distributing leaflets, brochures, posters (leaflets as some sort of teasers, and booklets for those who want to know more about a specific theme),
- communicating by several types of (social, internet or paper) media, and through inter-municipal waste associations and the local environmental services,
- using other educational materials (demonstration tools about processing organic remains f.i. compost boxes and –bins, wormeries, insect hotels, mulch mowers, wood chipper, school games, compost information box…)

3. Results and discussion

Research shows that this approach makes sense and is gradually evolving the way we wanted. A combination of instruments undoubtedly lies at the basis of the success (financial, policy and public awareness).
In the table below you see the percentage of citizens that are practising the closed loop technique named 'Home Composting'.

- In 1991 5% of the people in Flanders were composting at home.
- In 21 years time this amount evolves to 52%.

Note that this percentage includes all inhabitants of the Flemish region, not just those who have a garden. Currently, pproximately 106.000 tons of organic waste is processed at home, by composting. The exact amount is very difficult to estimate.

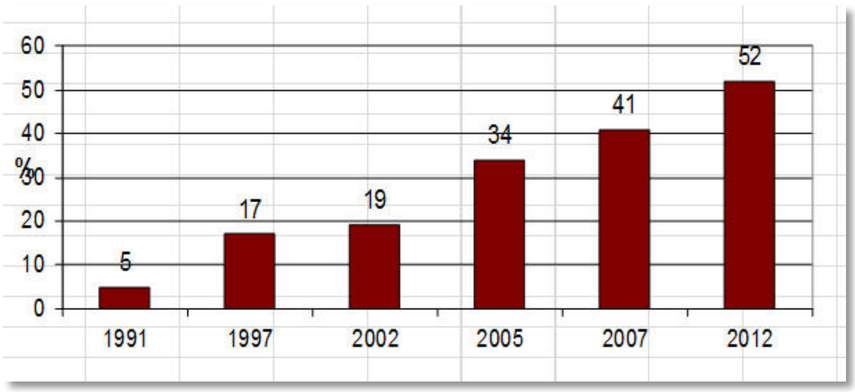

Figure 1: The percentage of inhabitants that are composting (all or part of) their organic waste at home (by using wormery, compost box, compost bin or compost heap), throughout the past 21 years.

In the table mentioned below, one can see some other methods of processing home produced organic residues and the percentage of people that are practising these methods at home.

- About 17% of the people practices one or another residue reducing method to maintain their lawn.

- Over 22% of the people applies one of the pruning processing methods I mentioned earli-er.
- Around 58% applies perennials in the garden to reduce the garden residues and garden work.
- And more than 1 on 4 families in Flanders keeps chickens to reduce the organic garden and kitchen residues.

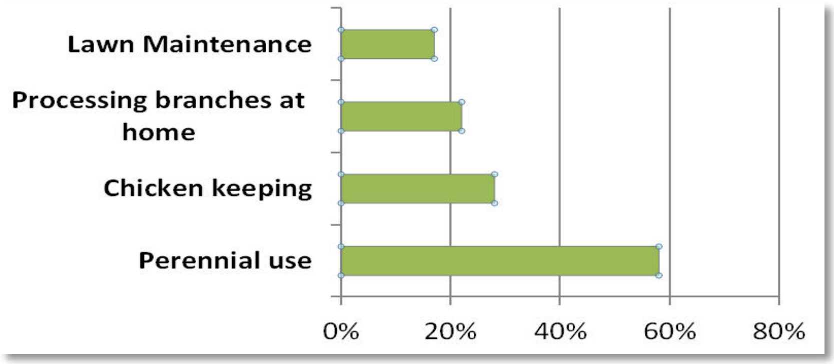

Figure 2: The percentage of inhabitants that are applying other closed loop techniques to reduce their organic waste.

The amount of people that practises (our) specific food loss reducing methods (= the most recent Biocycle at Home-theme) we do not know yet.

Almost 75% applies one or another biocycling technique to process organic waste at home, and more than 75% is planning to do so.

The unique co-operation model with authorities on different levels, a network of well trained teachers and trained volunteers (Master Composters / Biocycle Volunteers) who sensitize citizens was and still is successful.
The last years, we see that citizens still want to engage themselves and like to co-operate in or-ganisations, but they want to do it in a more noncommittal and trend-sensitive way. Social media and direct action are playing an important role in this. The lifelong commitment that we saw in former days is slowly disappearing.

4. Conclusion and outlook

The 2-fold policy followed, clearly has a positive result, but integrating the experience and knowledge of our original volunteers in the new ways of communicating and community creating, is the challenge for the future, ànd at the same time it is the assurance that the biocycle message is alive ànd meaningful!

What will bring the future for the ‘BioCycle at Home’ in Flanders?

The next years we will focus on some potentials, namely:

- Home Composting: research showed that 59% of the people are composting at home or are planning to do that on the short term. Getting to that 59% home composters is one target.
- Idem ditto what concerns Closed Loop Gardening. 75% is planning to implement one or another BioCycle theme in his own garden. So 75% or more is our goal.
- A gradual change of tasks and communication content requires a change of name. That’s why we will gradually evolve to the a new terminology (from Master Composters to BioCycle Volunteers).
- Trying to introduce the principles of the BioCycle into other (garden-, environment- and nature related) npo’s
- Taking into account and flexibly respond on a constantly changing legislation
- Further expanding our communication and sensitizing items

- We will focus on the future, but remembering and taking into account some preconditions, namely:
- That all individuals must / want to be sensitized, convinced in a different way (leaflet, education, google-discovering …)
- That Home composting is a valuable first step towards an improved relationship with environment
- That BioCycling at Home is part of a sensitizing process in order to change consumption behaviour (f.i. food loss).
- The roadmap on circular economy http://ec.europa.eu/smart-regulation/roadmaps/index_en.htm.

Acknowledgements

This 2-fold ‘Biocycle at home’-scheme that is the basis for the communication on treating organic waste in Flanders would not have been possible without the (financial) support of the Flemish government, the intermunicipal waste organisations, the municipalities and all the volunteers. Vlaco wishes to thank the biocycle volunteers / master composters for their ‘ever lasting’ engagement and enthousiasm.

References

[1] http://www.ovam.be/achtergrond-van-het-materialendecreet
[2] OVAM, Vlaco, 2003. - De kwaliteit van ‘thuis’compost. 136p.
[3] OVAM, 2012. - Evaluatieonderzoek materialenkringloop gft- en groenafval. 123p.

TA-O_06 Band application of acidified slurry as an alternative to slurry injection – An integrated evaluation

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1. Objectives

Slurry injection is currently the recommended method for slurry application to soil but, in some specific conditions (small field plots, stony fields), it might be difficult to use slurry injection and farmers are asking for some alternatives.
Slurry acidification has proved to be efficient to minimize ammonia (NH_3) emissions during soil application and band application of acidified slurry could be a good alternative to slurry injection. However, most of studies dealing with acidified slurry application to soil were performed in countries from North Europe with soils and climate conditions very different from those existing in Mediterranean countries. Furthermore, these studies are generally considering only NH_3 emissions and biomass production and there is a lack of knowledge relative to the impact of acidified slurry application to soil in terms of greenhouse gases emissions, nitrate leaching or long term effect on soil quality. Our main hypothesis is that slurry injection could be substituted by band application of acidified slurry and that soil incorporation of acidified slurry could be avoided.

The objective of the present study was to assess the efficiency of band application of acidified cattle slurry relative to non-acidified slurry injection, to supply plant nutrients with low nitrogen and carbon losses to air and water in two Portuguese soils.

2. Methodology

A medium scale experiment (1 m^2 plots) was performed at the campus of ISA in Lisbon (Portugal) with two different soils: a sandy soil (Haplic Arenosol) and a sandy loam soil (Haplic Cambisol). The sandy soil was collected in the Pegões area (near Lisbon) and the sandy loam soil in the center of Portugal (Castelo Branco). The former had a pH of 7.0 and its textural composition was 700 mg kg^{-1} of coarse sand, 170 mg kg^{-1} of fine sand, 97 mg kg^{-1} of silt and 26 mg kg^{-1} of clay. Sandy loam soil had a pH of 6.1 and its textural composition was 271 mg kg^{-1} of coarse sand, 558 mg kg^{-1} of fine sand 72 mg kg^{-1} of slit and 99 mg kg^{-1} of clay. The slurry used was taken from a dairy farm located near Lisbon. Slurry acidification was performed by addition of concentrated sulphuric acid (5ml to 1L slurry) to reach a final pH of 5.5.
Five treatments were considered: 1) control; 2) slurry injection (SI); 3) Band application of raw slurry followed by soil incorporation (SS); 4) Band application of acidified slurry (AS); 3) Band application of acidified slurry followed by soil incorporation (ASS). Tree replicates of each treatment and control were considered.
The traditional double cropping system maize (spring)/oat (autumn) was established between September 2012 and September 2014 in these two different soils. Untreated and acidified slurry were applied at a rate of 90 kg N ha^{-1} before oat and 170 kg N ha^{-1}, before maize sowing.
During the 2 years experiment, the following information was collected in all plots: NH_3, CO_2, CH_4 and N_2O emissions, nitrate leaching during oat growth, plant yield, slurry nutrients uptake by plants and finally soil quality with special emphasis on soil enzymatic activity.
Ammonia emissions were measured during the 8 days following slurry application using the dynamic flow chamber system. The nitrate-N leached from the crop root zone was estimated by combining the drainage flux with the nitrate-N concentration in the soil water collected from the suction cups installed at the depth of 70 cm. The soil water module of the Root Zone Water Quality Model (RZWQM) [1] was used to compute the drainage fluxes at the bottom of the root zone. The daily soil water balance was performed using precipitation data as a surface boundary condition. The bottom boundary condition was a variable drainage flux, calculated by the Darcy equation, as described in detail in [2]. Methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) fluxes were measured regularly using the closed chamber technique and the concentrations of the

gas samples stored in vials were measured by gas chromatography using a GC-2014 (Shimadzu, Japan). Plant harvest was performed manually; fresh material was weighed and subsamples taken for determination of dry matter and total N and P concentration, after pre-drying at 80 ºC for 24 h. Nitrification potential [3] and β-glucosidase, acid and alkaline phosphatases [4] and arginine deaminase activities [3] were determined in soil samples (0-20 cm) at the end of the experiment.

3. Results

Application of acidified slurry followed or not by soil incorporation, led to residual NH3 emissions similar to those observed in SI treatments while significant NH3 losses were observed in SS treatment (Figure 1). At spring application, before maize sowing, residual NH3 emissions were observed from SI and ASS treatments but a small amount of NH3 was also released from AS treatment, even if at a significantly (P<0.05) lower rate than from SS treatment. This release of NH3 in AS treatment occurred only 3-4 days after application and might be due to the pH increase of acidified slurry to values close to the untreated slurry. Our results showed that band application of acidified slurry is a good alternative to slurry injection to minimize NH3 emissions. Furthermore, incorporation of acidified slurry might not be necessary since it contributed to a very low decrease of NH_3 emissions.

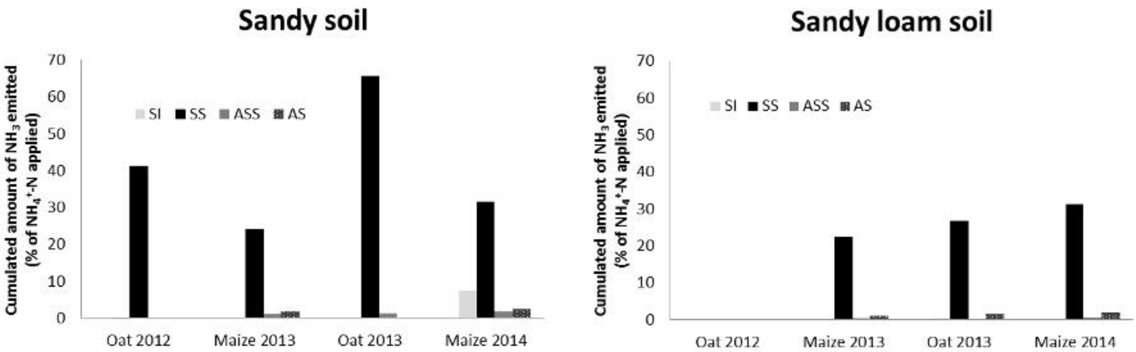

Figure 1: Cumulated amount of ammonia released after each slurry application before oat and maize growth in the two so and five treatments considered (*N*=3)

When considering greenhouse gases emissions, SI led to the highest (*P*<0.05) N_2O emissions during the summer period in sandy soil but to the lowest (*P*<0.05) during oat production where similar N_2O emission rates were observed in SS, AS and ASS. Slurry acidification has no negative impact on N_2O emissions since similar values were observed in SS, AS and ASS. Methane emissions were observed only during the first hours following application in all amended treatments with no significant (*P*>0.05) differences between treatments. Band application of acidified slurry might lead to lower GHG emissions than slurry injection even if such decrease relies strongly on the soil and climate conditions.

During the first oat production (2012-2013), slurry injection led to the highest (*P*<0.05) losses of N through nitrate leaching in both soils (Figure 2). Similar values of nitrate leaching were observed following acidified and raw slurry application when soil incorporation was performed immediately after application (SS and ASS). However, a significantly (*P*<0.05) higher nitrate leaching occurred in AS than in SS or ASS in both soils. During the second oat production (2013-2014), a significantly lower amount of nitrate was leached in all treatments and different trends were observed in each soil: the highest (*P*<0.05) nitrate losses were observed in AS and SI in the sandy soil and in SS and ASS in the sandy loam soil. Rainfalls were significantly lower in 2013-2014 relative to the previous years and rain events occurred mainly one month after slurry application what explained the lower N losses observed during this second year of experiment. Our results indicated that it might be of interest to incorporate the acidified slurry in soil to minimize nitrate leaching even if, such treatment led to the highest losses in sandy loam soils in 2013-2014. It is still to refer that the amounts of nitrate leached in SS and ASS were very close.

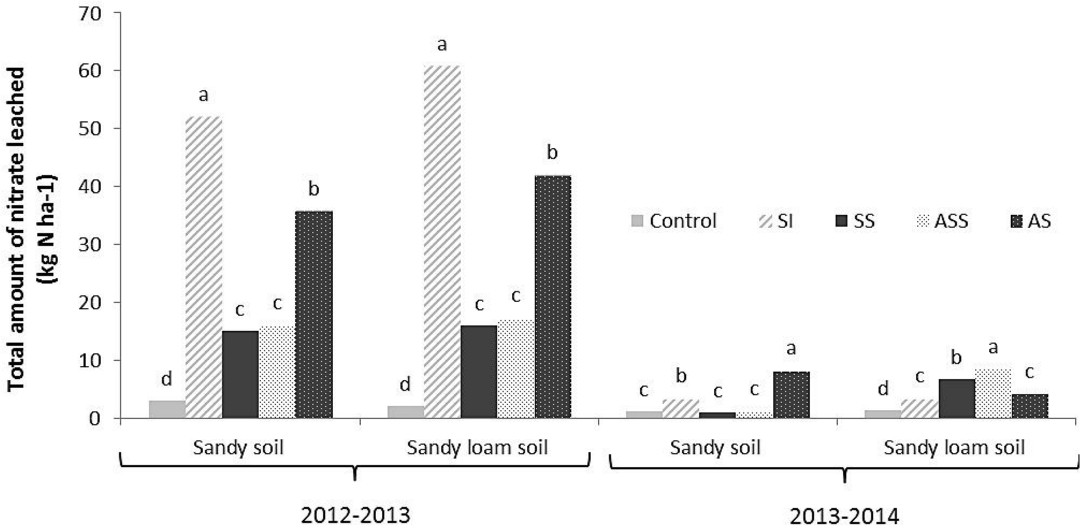

Figure 2: Cumulative amount of nitrate leached in each treatment in the two soils considered during oat production (N=3) – values quoted with similar letter are not statistically different at (P<0.05).

To assess the impact of slurry injection and band application of acidified slurry on crop yield, we calculated the percentage of yield increase obtained with each treatment relative to SS, considered here as the traditional (and still most used) techniques for slurry application to soil. All over the experiment, ASS, AS and SI treatments led to an increase of biomass production relative to SS except during oat production in 2013 where AS application to sandy soil led to a lower dry yield than SS (Figure 3). Nevertheless, the higher total yield increase relative to SS was obtained with ASS treatment in the sandy soil while in the sandy loam soil, ASS treatment led to the highest yield increase relative to SS. It is still to refer that an higher net N uptake was observed in ASS and AS treatments in the sandy and sandy loam soil, respectively, relative to SI and SS. It can then be concluded that band application of acidified slurry led to higher or similar yields relative to slurry injection. It is still to note that in the sandy loam soil, higher yields were obtained in AS relative to ASS indicating that incorporation should be avoided.

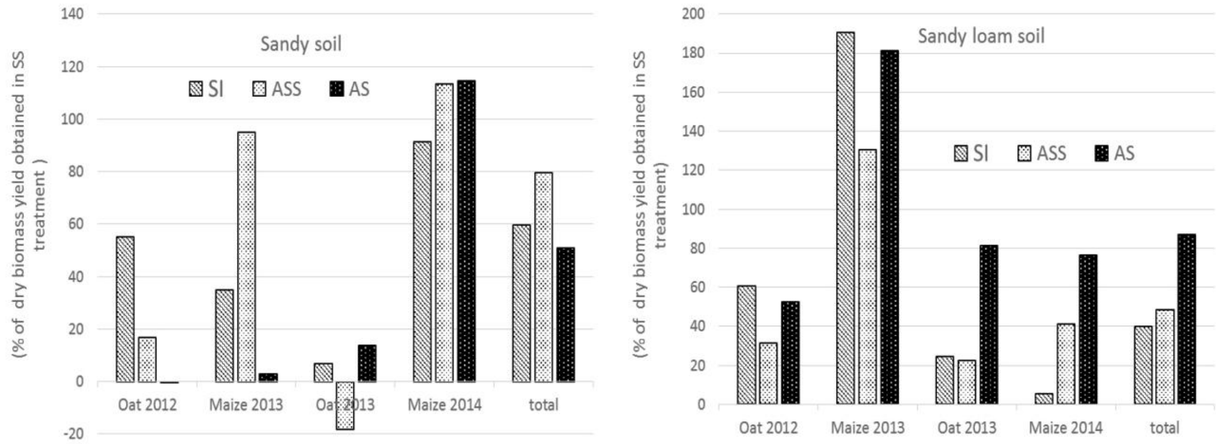

Figure 3: Variation of biomass production in SI, AS and ASS treatments expressed as % of the dry biomass yield obtained in SS treatment over the 2 years experiment (N=3).

After 4 consecutive slurry applications, a significantly higher activity of β-glucosidase was observed in AS and ASS relative to SS and SI in sandy soil whereas in the sandy loam soil, such effect was observed only in AS treatment (Table 1). Application of acidified slurry stimulated the activity of Acid phosphatase but depressed the activity of alkaline phosphatase relative to SS and SI. The impact of acidified slurry application on the activity of Arginine deaminase depends on the soils considered and is also influenced by soil incorporation. Finally, a significant (P<0.05) decrease of the Nitrification potential was observed in soils amended with acidified slurry relative to SI or SS, namely in the sandy loam soil where values in AS and ASS are similar to values observed in control.

Table 1: Nitrification potential and enzymatic activity in soils at the end of the experiment (*N*=3)

	β-glucosidase (µg *p*-nitrof.g^{-1} h^{-1})	Acid phosphatase (µg *p*-nitrof.g^{-1} h^{-1})	alkaline phosphatase (µg *p*-nitrof.g^{-1} h^{-1})	Arginine deaminase (µg N-NH_4^+ g^{-1} 3 h^{-1})	Nitrification potential (µg N-NO_2^- g^{-1} 5 h^{-1})
			Sandy soil		
control	$^{\alpha}$42.6^{c}	71.1^{e}	61.3^{d}	2.8^{c}	1.4^{e}
SS	70.1^{b}	116.5^{d}	129.1^{a}	3.8^{b}	3.6^{a}
SI	65.6^{b}	147.5^{b}	111.4^{b}	3.8^{b}	2.8^{b}
AS	83.2^{a}	136.7^{c}	106.4^{b}	4.2^{a}	2.1^{d}
ASS	87.7^{a}	176.0^{a}	97.5^{c}	3.7^{b}	2.4^{c}
			Sandy loam soil		
control	65.6^{d}	221.2^{e}	81.1^{d}	1.4^{c}	1.0^{cd}
SS	107.1^{b}	299.5^{d}	206.7^{a}	2.4^{a}	2.9^{a}
SI	97.6^{c}	325.9^{c}	159.3^{b}	2.2^{ab}	2.2^{b}
AS	130.5^{a}	341.2^{b}	134.0^{c}	2.1^{b}	0.9^{d}
ASS	96.9^{c}	378.4^{a}	88.0^{d}	1.6^{c}	1.1^{c}

$^{\alpha}$ values in a same raw followed by similar letter are not statistically different at ($P<0.05$)

4. Conclusion

Our results showed that band application of acidified slurry is a good alternative to slurry injection and that soil incorporation of acidified slurry is recommended to minimize nitrate leaching.

No negative impact on enzymatic activity was observed after 4 consecutive application of acidified slurry and in some instances, it can be said that application of acidified slurry is beneficial since it decrease the nitrification potential and increase enzymatic activity.

It is also important to highlight the fact that acidified slurry application induced, in most cases, an increase of crop yield relative to the traditional slurry broadcast followed by incorporation (SS). This increase of crop production and the lower energy required for soil application relative to injection might be sufficient to balance the cost associated with slurry acidification. Nevertheless more studies are still needed to improve safety during the acidification process, namely the utilization of other additives rather than concentrated sulfuric acid.

Acknowledgements

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References

[1] Ahuja, L.R., Rojas, K.W., Hanson, J.D., Shaffer, M.J., Ma, L. (2000): The Root Zone Water Quality Model. Water Resources Publ., LLC, Highlands Ranch, CO, p. 372.

[2] Cameira, M.R., Fernando, R.M., Ahuja, L., Pereira, L.S., (2005): Simulating the fate of water in field soil–crop environment. Journal of Hydrology 315, 1–24..

[3] Schinner, F., Kandeler, E., Öhlinger, R. and Margesin, R. (1996): Methods in Soil Biology. Springer-Verlag, New York. 425 p

[4] Tabatai, M.A. (1994) Soil enzymes. in Weaver, R.W et al. (eds.) Methods of Soil Analysis, part 2: Microbiologial and Biochemical Properties SSSA, Madison, Wisconsin, USA.

TA-O_07 The recycling potential of phosphorus in Norwegian secondary resources in a systems's context

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1. Objective

Substituting mineral phosphorus (P) fertilizer with secondary P fertilizers is a crucial step towards increased P sustainability. Secondary P fertilizers are based on P in waste products in comparison to mineral P fertilizer that is produced with rock phosphate, a limited resource. Substance flow analysis is a useful tool to estimate the amount of P in waste products and a country's recycling potential. Many studies, however, only consider total P amounts e.g. [1], [2] and [3], even though in secondary P resources the fraction of plant-available P is often considerably lower than the total P amount. This is because P in secondary resources is commonly found in the form of various complex P compounds of different solubility [4], in comparison to mineral fertilizer, where P is present as simple compounds (e.g. monocalcium phosphate). Mineral fertilizer equivalents (MFE, %) represent the fractions of P in secondary resources that are plant-available and that can substitute mineral fertilizer. The objective of the present study was to estimate the P recycling potential in Norway by comparing the total and plant-available amount of P in secondary resources. In order to do this, we combined the results from a P flow analysis [2] with the MFE of secondary P resources, as studied by a bioassay and literature.

2. Methodology

- P flow analysis (PFA):
  PFA was applied to the Norwegian food system to estimate the P recycling potential of Norwegian biomass. The system included plant agriculture, aquaculture/fisheries, human consumption and waste management. Averaged data from 2009-2011 were used in order to avoid annual variations. Data were collected from government statistics, reports, company publications, expert interviews and scientific publications. For a detailed description of the P flow analysis see [2].

- Mineral fertilizer equivalents (MFE)
  MFE were determined for those secondary resources that represented the largest amounts of total P. MFE were calibrated assuming that plant uptake of P in mineral fertilizer = 100%. MFE were mainly derived from a 1-year bioassay and supported by values from literature describing experiments on the P fertilization effect of secondary resources the first year after application. In the bioassay, the P fertilization effects of 7 Norwegian secondary resources (dairy manure, chicken manure, meat bone meal, fish sludge, 2 types of digestate based on source-separated household waste and reactor-composted catering waste) were compared with that of mineral fertilizer ($Ca(H_2PO_4)$). Ryegrass (*Lolium multiflorum* var. Macho) was used as experimental crop and a nutrient-deficient sand-peat mixture at two pH levels as experimental soil. Confidence intervals (95%) of MFE were calculated based on the available studies and assuming a normal distribution. The lower and the upper values were used to calculate the minimum and maximum recycling potential.

- Recycling potential
  The Norwegian recycling potential was defined as the sum of the recycling potentials of the secondary P resources and determined by multiplying total P amounts in secondary resources with their associated MFE. The total demand for fertilizer P in Norway was defined as equal to the P uptake by agricultural crops. With this definition we assumed a long-term situation where all agricultural soils have optimal P levels and losses are negligible, thus only the amount of P removed with the crop has to be applied as fertilizer to achieve optimal yields [5]. Data on plant-available soil P (measured as P-AL, mg P/100g

soil, extracted by 0.1 M ammonium lactate and 0.4 M acetic acid adjusted to pH 3.75) from the Norwegian soil database for the period 2001-2011 shows that 89% of the soil samples fall into the categories optimal, moderately high, high, and very high levels of P-AL [6]. Therefore, in the short run, our definition leads to an overestimation of the total demand for fertilizer P.

Figure 1 is an illustration of how our model distinguished between directly plant-available P versus not directly available P in returned secondary resources. We assumed that crops utilize only the directly plant-available fraction (equal to 100% MFE) the first year after application, while the not directly available fraction is added to the P-stock in the soil. P demand is defined here as demand for plant-available P. In our model, there is no flux between the not directly available and directly available P pools. We assumed that these two compartments remained isolated from one another. In reality, this is not the case, as there are several interactions and flows in both directions between the two P pools. These mechanisms, however, are dependent on local factors such as soil type and pH, which were not included in this model.

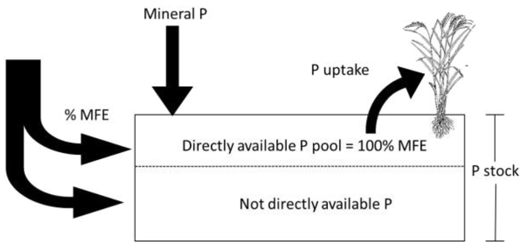

Figure 1. Illustration of secondary P partitioning

3. Results and discussion

3.1 PFA

In the period 2009-2011, on average agricultural crops took up 11000 Mg P/yr, in our model representing the demand for plant-available P. 8400 Mg P/yr was applied to Norwegian soil as mineral fertilizer, all of which was produced by imported rock phosphate. In comparison, 28000 Mg P/yr was estimated to be in secondary resources, of which the most important were animal manure (11400 Mg P/yr with 60% as cattle manure and the residual as sheep, pig, poultry, fox and horse manure), losses in aquaculture in the form of fish excrements and feed residues (fish sludge, 9000 Mg P/yr), municipal solid waste (2600 Mg P/yr), meat bone meal (2100 Mg P/yr), sewage sludge (1900 Mg P/yr) and dumped fish scrap (1100 Mg P/yr). In 2009 – 2011, all P in manure in addition to 20% of meat bone meal, 50% of P in sewage sludge and 3% of municipal solid waste was returned to agricultural soil.

3.2 MFE

MFE used in the study can be found in table 1.

3.3 Recycling potential

By combining the total amount of P in secondary resources with their MFE, we found the total P recycling potential in Norway to be 12600 – 26200 Mg P/yr. Despite large uncertainty, based on the minimum recycling potential we conclude that the total demand for fertilizer P in Norway could theoretically be covered by directly plant-available P in secondary resources and that all mineral fertilizer could therefore be substituted.

3.4 Data quality

Large deviation between the minimum and the maximum recycling potential can be explained by MFE of secondary resources being dependent on a range of parameters. Examples are the characteristics of the soil to be fertilized, the experimental crop used as well as the experimental design.

Table 1: Confidence intervals (95%) of mineral fertilizer equivalents (MFE, %) used in the study, calculated assuming normal distribution. If n = 1, we assumed the uncertainty to be 50%.

Secondary P resource	Category/treatment	MFE (%)	n	Reference
Manure	Cattle manure	[85, 115]	14	Bioassay; [7]; [8]; [9]; [10]; [11]; [12]
	Sheep and goat manure	[75, 125]	1	[13]
	Pig manure	[77, 123]	6	[10]; [12]; [14]
	Poultry manure	[65, 74]	3	Bioassay; [12]
	Fox manure	[29, 79]	1	[15]
	Horse manure [a]	[75, 125]	1	Own assumption
Fish sludge	Reactor-composted	[21, 115]	3	Bioassay, own unpublished data
Municipal solid waste	Compost	[39, 65]	10	Bioassay; [19]
	Digestate	[55, 86]	4	Bioassay
Meat bone meal	Treated with heat and pressure	[5, 64]	5	Bioassay; [15]; [16]
Sewage sludge	Chemical or chemical-biological treatment	[20, 29]	9	[17]
	Biological treatment	[75, 125]	1	[18]
	Mechanical treatment	[75, 125]	1	Own assumption
Dumped fish scrap [b]		[10, 60]	1	Own assumption

[a] Assumed to be equal to cattle manure, [b] assumed to be equal to meat bone meal

In addition to the MFE uncertainty, the data quality of the PFA greatly influences the reliability of this study. While the estimation of P uptake by agricultural crops is estimated based on relatively robust data, data gaps and mass balance inconsistencies within the PFA were most prevalent in the waste sectors. For example, poor data availability for food processing and post-consumer wastes required crude assumptions with high uncertainties. Additionally, the methods used to individually calculate each flow could have masked hidden waste flows, contributing to large mass balance inconsistencies. The sections of the model that were estimated due to large data gaps, however, were crosschecked with secondary data and were shown to be in the correct order of magnitude. For more information related to the PFA uncertainty, refer to [2].

3.5 Theory versus feasibility

Currently, P recycling is far from optimized, as shown by the great over-application of plant-available P in secondary resources and mineral fertilizer to agricultural soil compared to the actual crop P requirements [6]. This contributed to an average net accumulation of 12000 Mg P/yr in Norwegian soil stocks in the years 2009-2011. The reasons for this include a combination of technological, economic and social parameters as well as a lack of political incentives. Some aspects are discussed below:

One of the main reasons for insufficient utilisation of plant-available P in Norwegian animal manure is the uneven geographical distribution of animal husbandry and crop farming [6]. Transportation of P in animal manure to P-deficient areas would require treatment of the bulky material to reduce its water content without reducing the high P fertilisation effect.

The main challenge regarding the recycling of P in fish sludge from aquaculture is its collection from open cages in Norwegian fjords. While technologies exist, they are in their infancy and require further development before this can be realised. Moreover, collected fish sludge would require dewatering and desalting before it could be used as secondary fertiliser. Even though the P fertilisation effect of fish sludge has been shown to be promising in the bioassay, its performance under field conditions is still to be tested.

Today, fish scrap, meaning fish bones and guts from fisheries, is dumped off-shore. Bringing this secondary resource to land is costly, and therefore utilising P in fish scrap would require political incentives. Little is known about the P fertilisation effect of fish scrap. P in fish scrap is assumed to be mainly present as stable calcium phosphates in fish bone. Therefore, for the present study we assumed MFE to be equal to meat bone meal.

Meat bone meal and sewage sludge are both characterised by low MFE. In meat bone meal P is mainly present as hydroxyapatite [20], which is only partially soluble in soils, also dependent on pH in the soil to be fertilised. Only if P in meat bone meal is solubilized e.g. by treatment with sulphuric acid, could it be turned into a valuable secondary P fertiliser. In Norwegian sewage sludge, P is mainly present as phosphate bound to aluminium (Al) or iron (Fe) with low plant-availability. This is due to the common use of Al- and Fe-precipitation agents to remove P during wastewater treatment [18].

4. Conclusion and outlook

The presented study is an attempt to improve the quality of phosphorus flow analysis (PFA) and the estimation of a country's recycling potential, using Norway as a case. Combining PFA with mineral fertilizer equivalents as a parameter for the plant-availability of P in waste products, we demonstrate that the total demand for fertilizer P in Norway could theoretically be covered by plant-available P in secondary fertilizers. In this study we did not take into account other parameters that could limit the P recycling potential such as distribution (e.g. manure) of secondary resources and collection (e.g. fish sludge), or waste quality (e.g. heavy metals, pathogens, antibiotics, salt concentration), regulations and attitudes.

Acknowledgements

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References

[1] Antikainen, R.; Lemola, R.; Nousiainen, J.I.; Sokka, L.; Esala, M.; Huhtanen, P.; Rekolainen, S. (2005): Stocks and flows of nitrogen and phosphorus in the Finnish food production and consumption system, Agriculture, Ecosystems & Environment 107, 287–305

[2] Hamilton, H.; Brod, E.; Hanserud, O.; Gracey, E.; Vestrum, M.; Steinhoff, F.; Mueller, D.; Brattebø, H.: Investigating cross-sectoral synergies through integrated aquaculture, fisheries and agricultural phosphorus assessments: Norway as a case, Journal of Industrial Ecology. In press

[3] Senthilkumar, K.; Nesme, T.; Mollier, A.; Pellerin, S. (2012): Conceptual design and quantification of phosphorus flows and balances at the country scale: A case of France. Global Biogeochemical Cycles. doi: 10.1029/2011GB004102

[4] Hedley, M.; McLaughlin, M. (2005): Reactions of phosphate fertilizers and by-products in soils. In: Sims TJ, Sharpley AN (eds.) Phosphorus: Agriculture and the Environment. 2nd edn. Agronomy monograph no. 46, Madison, Wisconsin, pp 181-252

[5] Krogstad, T; Øgaard, AF; Kristoffersen, AØ (2008): New P recommendations for grass and cereals in Norwegian agriculture. Nordic association for agricultural scientists (NJF) report 4:42-46

[6] Hanserud, OS; Brod, E.; Øgaard, AF; Müller, D; Brattebøe, H.: A mulitregional soil-phosphorus balance for exploring secondary fertilizer potentials – the case of Norway. Accepted for publication in Nutrient Cycling in Agroecosystems

[7] During, C.; Weeda, W.C. (1973): Some effects of cattle dung on soil properties, pasture production, and nutrient uptake. New Zealand Journal of Agricultural Research 16, 423–430

[8] Goss, D.W.; Stewart, B.A. (1978): Efficiency of phosphorus utilisation by alfalfa from manure and superphosphate. Soil Science Society of America Journal 43, 523–528

[9] Larsen, K.E. (1981): Phosphorus effect of animal manure and sewage sludge. In: Hucker, T. W.G.; Catroux G. (eds.) Phosphorus in sewage sludge and animal waste slurries, pp 207–228. Dordrecht, The Netherlands: Reidel

[10] Leytem, A.B.; Westermann, D.T. (2005): Phosphorus availability to barley from manures and fertilizers on a calcareous soil. Soil Science 170, 401–412

[11] Motavalli, P.P.; Kelling, K.A., Converse, J.C. (1989): First-year nutrient availability from injected dairy manure. Journal of Environmental Quality 18, 180–185

[12] Smith, K.A.; van Dijk, T.A. (1987): Utilisation of phosphorus and potassium from animal manures on grassland and forage crops. In: v.d. Meer, H.G. (ed) Animal manure on grassland and fodder crops, pp 87–102

[13] McAuliffe, C.; Peech, M.; Bradfield, R. (1949): Utilization by plants of phosphorus in farm manure. II. Availability to plants of organic and inorganic forms of phosphorus in sheep manure. Soil Science 68:185–195

[14] Tunney, H.; Pommel, B. (1987): Phosphorus uptake by ryegrass from monocalcium phosphate and pig manure on two soils in pots. Irish Journal of Agricultural Research 26, 189–198

[15] Ylivainio, K.; Uusitalo, R.; Turtola, E. (2008): Meat bone meal and fox manure as P sources for ryegrass (Lolium multiflorum) grown on a limed soil. Nutrient Cycling in Agroecosystems 81,267–278

[16] Jeng, A.S.; Haraldsen, T.K.; Grønlund, A.; Pedersen, P.A. (2006): Meat and bone meal as nitrogen and phosphorus fertilizer to cereals and rye grass. Nutrient Cycling in Agroecosystems 76, 183–191

[17] Øgaard, A.F. (2013): Plantetilgjengelig fosfor i avløpsslam: Testing av analysemetodikk for tilgjengelig fosfor, Bioforsk Rapport Vol. 8 Nr. 34

[18] Krogstad, T.; Sogn, T.A.; Asdal, Å.; Sæbø, A. (2005): Influence of chemically and biologically stabilized sewage sludge on plant-available phosphorus in soil. Ecological Engineering 25, 51–60

[19] Sinaj, S.; Traore, O.; Frossard, E. (2002): Effect of compost and soil properties on the availability of compost phosphate for white clover (Trifolium repens L.). Nutrient Cycling in Agroecosystems 62, 89–102

[20] Brod, E.; Øgaard, A.F.; Hansen, E.; Wragg, D.; Haraldsen, T.K.; Krogstad, T. Norwegian waste products as alternative phosphorus fertiliser. Part I: Inorganic P species affect fertilisation effects dependent on soil pH. Submitted to Nutrient Cycling and Agroecosystems

TA-O_08 Nitrogen recovery from digested slurry with simplified ammonia stripping technique

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1. Objectives

Among the treatments to reduce nitrogen (N) surplus from livestock slurry, ammonia stripping is an effective technology [1], but currently expensive and difficult to manage. In this work we evaluated a simplified, slow-release ammonia stripping system to examine if nitrogen could be removed from slurry and collected in an acid solution, thus turning it into a mineral fertilizer that reduced its potential environmental impact. Here we report only results of the ammonia removal experiments.

2. Methodology

A pilot plant consisting of 3, 50-L reactors was used (Figure 1). Reactors were thermally controlled at 30, 40, and 50 °C and continuously mixed. Ammonia volatilization was obtained by pumping an air stream at 10 L min^{-1} through the headspace of the reactors. Two tests lasting 10 days each were conducted on a digested slurry (digestate). The first test was conducted at the natural pH of digestate (about 8) and the second at an adjusted pH of 9. The pH and Total Ammoniacal Nitrogen (TAN) content were determined daily.

Figure 1: Reactors to assess removal of nitrogen from slurry with a slow-stripping technique.

The digestate was collected from a biogas plant (1 MW) fed with animal manure (pig and cattle slurry, cattle farmyard manure and poultry manure) and a limited amount (5%) of maize silage. The digestate was collected following mechanical (screw press) separation.

At the start of each experiment, each reactor was filled with around 40 L of digestate drawn from a 150-L batch previously mixed manually. The digestate in the second test was corrected to pH 9 by adding sodium hydroxide. The Total Kjeldahl Nitrogen (TKN), TAN and dry matter content of the digested slurries were determined at the beginning and the end of each experiment using standard procedures (APHA, 1998). Results are reported in Table 1.

Table 1: Characteristics of digestate at the beginning and end of the two experiments.

Parameter	Experiment 1 (natural pH, ca. 8)	Experiment 2 (pH 9)
Total Kjeldahl Nitrogen (g/L)	4.50	2.96
Ammoniacal Nitrogen (g/L)	2.38	2.09
Dry matter (%)	3.14	2.89

The TAN content of each reactor was determined daily. As some water evaporation occurred during the experiment, the volume in each reactor was noted and the results of the analyses were corrected to obtain a mass balance of the nitrogen losses.

3. Results and discussion

3.1 Removal efficiencies at natural pH

The results of experiment 1 (natural pH) are reported in Figure 2 and show the expected effect of temperature. At 30 °C the stripping process was slower than at the higher temperatures, and after 10 days only 44% of the initial ammoniacal N was lost. At this temperature ammoniacal N in the reactor occasionally increased, albeit slightly, probably due to the mineralization of organic N together with the reduced ammonia emissions. At higher temperatures, nitrogen removal was more significant. Initial TAN was reduced by 67% and 81% at 40 °C and 50 °C, respectively.

These results can be explained also by the pH trend. Figure 1 shows that the pH increased to 9 (at the end of testing at 30 °C). As ammonia volatilization should decrease pH, this trend might be unexpected. However, it is well known that digested slurry contains considerable CO_2 that can be easily released in conditions such as those of the experiment (heating and mixing). Thus, the trend of pH can be interpreted as the result of two opposite phenomena: CO_2 stripping that increases the pH and ammonia stripping that reduces pH. This is confirmed by the lower pH measured in the reactor kept at 50 °C, in which ammonia removal was more substantial.

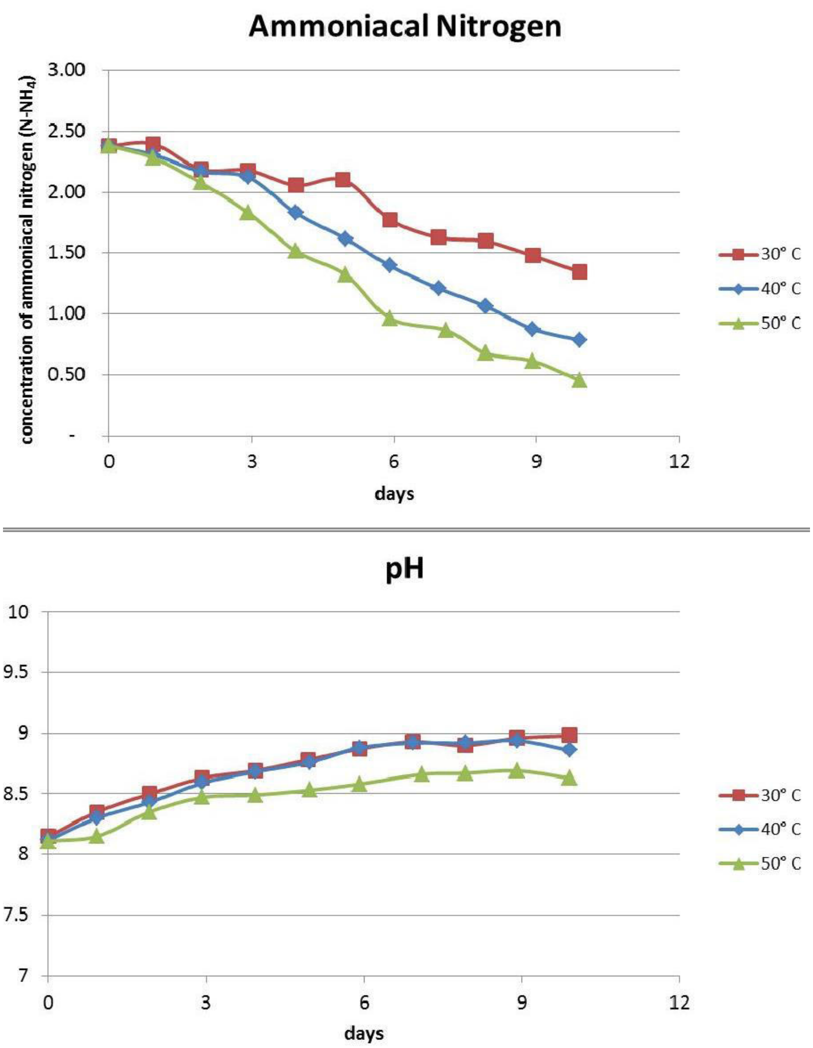

Figure 2: Trends in concentrations of ammoniacal nitrogen (above) and pH (below) during the experiment carried out with digested slurry at natural pH.

3.2 Removal efficiencies at modified pH

As expected, ammonia emission at pH 9 was significantly different from that at the lower, natural pH because the elevated pH moved the chemical equilibrium towards (non-dissociated) ammonia. Compared with Figure 2, Figure 3 shows clearly the quicker decrease of TAN at the higher pH, reaching reductions of 66%, 86% and 96% at 30 °C, 40 °C and 50 °C, respectively. However, the ammonia volatilization rate decreased noticeably after the first 3-4 days. This result can be explained by examining pH which, after the initial modification, remained relatively stable. Furthermore, the reduced concentration of ammonia in the liquid after the first days reduced the driving force, and thus the rate, of further emissions. As in experiment 1, there is a clear increase in TAN concentration after the first day in the reactor maintained at 30 °C.

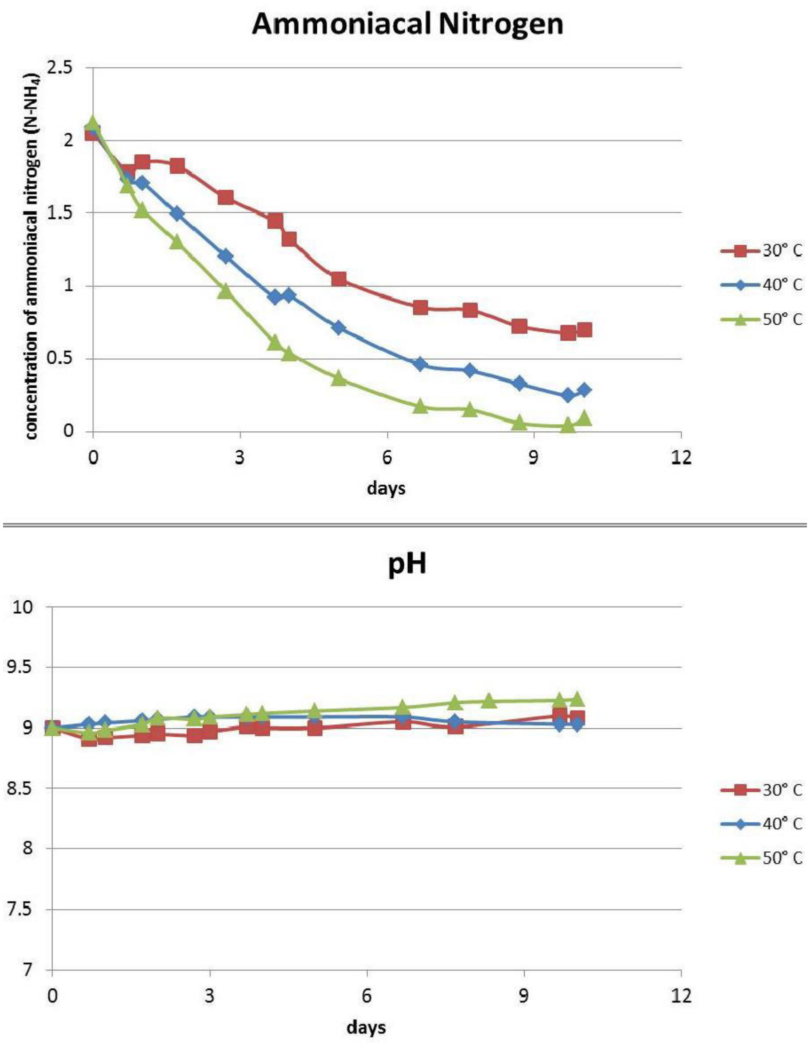

Figure 3: Trends in concentrations of ammoniacal nitrogen (above) and pH (below) during the experiment carried out with digested slurry with an initial pH value corrected to 9 by addition of NAOH.

3.3 Considerations on pH adjustment and process temperature

These experiments highlighted the potential of stripping technology based on a long retention time, and of limited pH adjustment, to obtain ammonia reduction in digested livestock slurry of up to 90%. These results are encouraging as this technique is rather simple and does not require dedicated preprocessing of slurry. In fact, considering that a separation step is already a common treatment in many biogas plants, the nitrogen removal process tested can be implemented easily. The relatively high dry matter content of digestate at the beginning of treatment does not seem to adversely affect process efficiency.

Comparing the results obtained with and without pH adjustment, it is obvious that better ammonia removal can be achieved at increased pH. However, Figure 4 shows that it is possible to have trade-offs among operating conditions. For example, the same ammonia removal efficiencies in 10 days of retention time can be obtained at 30 °C with pH adjusted to 9, and at 40 °C without pH adjustment. Moreover, the same efficiencies can be obtained by modifying the retention time. For example, if the target ammonia removal is 60%, this can be achieved with almost all the experimental conditions tested (with the exception of 30 °C without pH adjustment) by varying the retention time from 3 to 9 days.

In principle, high ammonia removal efficiencies can be obtained with an initial, natural pH around 8 and at ambient temperature, provided that the retention time is long enough. Further, considering that this treatment can be accomplished as a batch process, it might be integrated into slurry storage facilities, especially those that are covered. The subsequent storage and handling of the processed slurry will not require special attention as further ammonia emissions will be very limited.

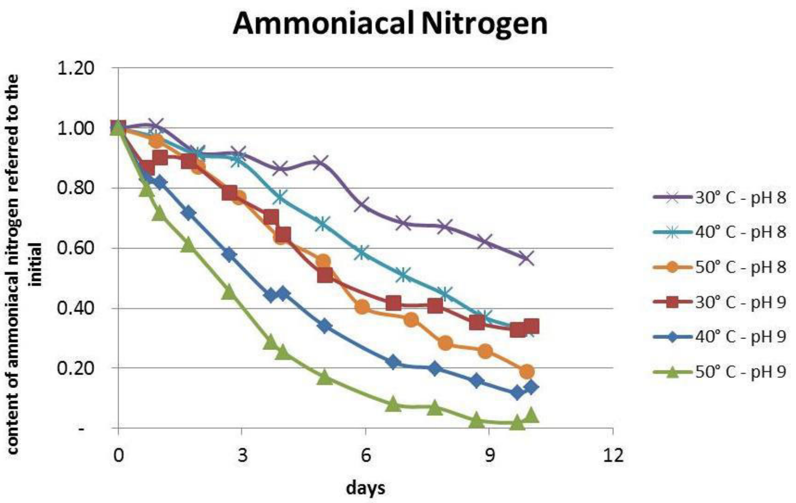

Figure 4: Ammoniacal nitrogen reduction in digested slurries under different conditions.

4. Conclusions and outlook

The results of this work show that air stripping at pH 9 and 50 °C promotes the greatest reduction of ammonia from digested animal slurry. Nevertheless, ammonia reduction at pH 8 and 40 °C is interesting because these are common conditions at typical biogas plants. As most of such plants are mesophilic, the stripping process does not need supplementary heat beyond what is necessary to maintain the liquid at the required temperature. The results demonstrate that the nitrogen removal efficiency of this slow-release system is close to that of faster treatments that require higher pH and temperature [2].

In order to develop a widely applicable technique however, further research is necessary. The process should be tested on different slurries to define the performances of the system when the initial waste material has characteristics different from those in this study. A second investigation should optimize the process in terms of airflow and mixing requirements, and an assessment of the energetic and economic sustainability of the process is needed. Lastly, research should evaluate if it would be advantageous to raise the reactor pH rapidly, accelerating the volatilization of CO_2, by optimizing the aeration system [3].

Acknowledgements

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References

[1] Bonmatì A.,Flotats X. (2003): Air stripping of ammonia from pig slurry: characterization and feasibility as pre- or post- treatment to mesophilic anaerobic digestion – Waste Management 23,261-272

[2] Gustin S., Marinsek-Logar R. (2011): Effect of pH, temperature and air flow rate on continuous ammonia stripping of anaerobic digestion effluent – Process Safety and Environmental Protection 89, 61-66

[3] Zhu, J., Luo, A., Ndegwa P. (2001): Raising pH by low–level aeration for soluble phosphorus removal of swine manure. Transactions of the ASAE 44, 391–396.

TA-O_09 Struvite recovery from methanogenic landfill leachate by chemical precipitation

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1. Objectives

The work is a part of the project for the treatment of leachate from the methanogenic landfill site at Ihlenberg, Germany. Removal and/or recovery of ammonium from the leachate as magnesium ammonium phosphate (MAP), also known as struvite was investigated. The work focuses on enhancing the quality of precipitate to facilitate agricultural use. The research targeted at:

- Assessing the influence of wastewater matrix on product purity,
- Investigating the combination of nanofiltration with struvite precipitation
- Determining the optimal pH for the process.

2. Methodology

A schematic of the membrane system used in this work (also existing at Ihlenberg) is shown in Figure 1. The reverse osmosis retentate (R1) obtained from the landfill site was further handled using a nanofiltration unit (DOW NF270 membrane).

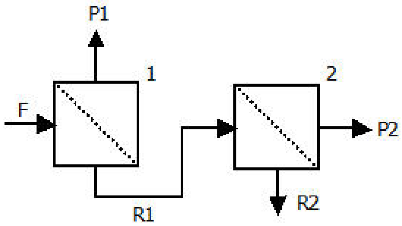

Figure 1: Membrane process train utilised

1 - Reverse osmosis (2 stage)
2 - Nanofiltration
F - Raw leachate
R - Retentate
P - Permeate

All precipitation experiments were performed with 1l batch volume in an Imhoff cone equipped with a propeller mixer (set at 130 rpm) and a pH meter. 1M solutions of $MgCl_2$ and H_3PO_4 were dosed in small volumes in 1:1 molar ratio at equal intervals. 2M NaOH solution was utilized to regulate the pH during the experiments.

- To establish the effect of wastewater matrix, experiments were conducted with the wastewater from the streams (see Figure 1) P2 and F (five times dilution of R1) at pH 8.75,
- To understand the effect of pH, experiments were conducted with the permeate P2 at pH values ranging from 8.0-9.5.

In all experiments, small aliquots were withdrawn from the supernatant at regular intervals and analysed for NH_4^+-N concentration using a Seal AutoAnalyser 3, following DIN 38406-E5-1. Samples from two experiments were analysed for Total Organic Carbon (TOC) by difference method and Total Nitrogen (TN) using a Multi N/C 3000 analyser from Analytik Jena, following DIN EN 1484-1997 and DIN EN 12260:2003-12 respectively. After the experiment, the precipitate was dried at 50°C for about 48h and analysed by wet chemical techniques i.e. by re-dissolving in deionized water acidified with HCl and analysed for NH_4^+-N, o-PO_4^{3-}-P (using Hach Lange LCK 348) and TOC for determining the product purity. Furthermore, the dried product was analysed using a Siemens D500 X-ray diffractometer (copper source, 30 mA, 40 kV).

3. Results and discussion

Table 1: Characteristics of different streams

Parameter	Unit	Method	F	R1	P2
NH_4^+-N	mg/l	DIN 38406-E5-1	594	3455	1966
TOC	mg/l	DIN EN 1484-1997	691	4328	320
TN	mg/l	DIN EN 12260:2003-12	660	3995	2432
Measured by the landfill operator in 2013					
Mg^{2+}	mg/l	DIN EN ISO 14911 - E 34	81	350	-
o-PO_4^{3-}	mg/l	DIN EN ISO 6878 - D 11	4.50	15.6	-
NO_3^--N	mg/l	DIN EN ISO 10304-2 - D 20	< 2	< 2	-
NO_2^--N	mg/l	DIN EN 26777 - D 10	< 0.3	1.15	-

The values of relevant parameters for different wastewater streams are presented in Table 1. It becomes clear that the leachate also contains organic bound nitrogen and that the measured TN values can be considered as total Kjeldahl nitrogen (TKN) since the concentrations of other forms is negligible.

3.1 Effect of wastewater matrix

To establish the effect of matrix and to analyse the combination of membrane techniques for enhancing product quality has been the main focus of this work. Figure 3a shows the measured NH_4^+-N and TN concentration in the supernatant with the addition of $MgCl_2$ and H_3PO_4 during the experiments with raw leachate (F) and nanofiltration permeate (P2) performed at a pH of 8.75.

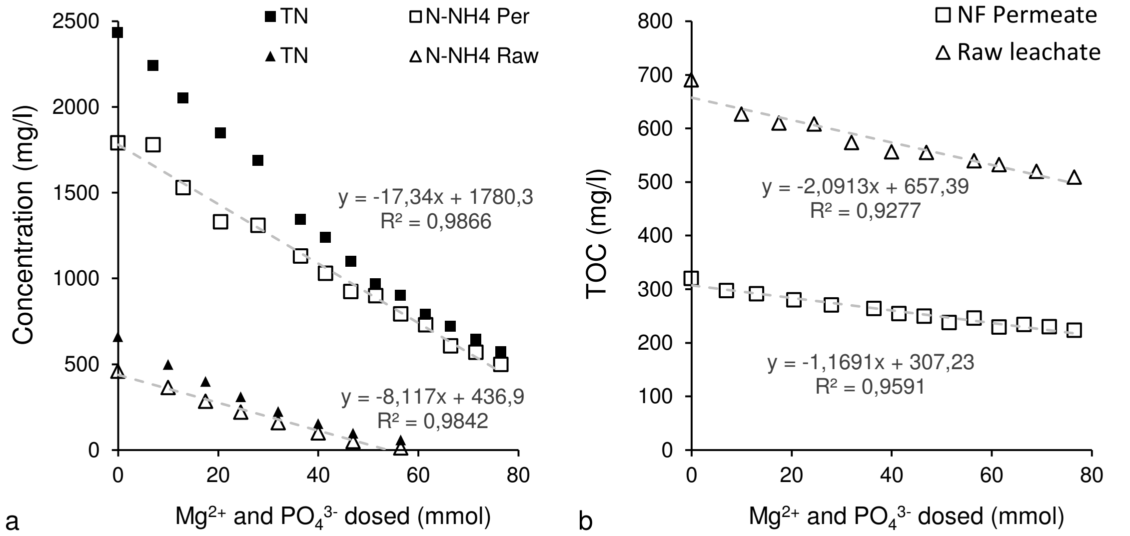

Figure 3: Comparison of (a) N-removal (b) TOC reduction: in permeate and raw leachate with dosage of $MgCl_2$ and H_3PO_4

Upon comparing the slopes of ammonium-nitrogen removal curves, the slope for removal in permeate which is roughly twice that of raw leachate, indicates a better removal capacity or in other words, a better utilization of added chemicals in the former. It becomes further clear with the following comparison. In the case of NF permeate, for an added volume of 41.5 ml of both solutions a removal of 760 mg NH_4^+-N was achieved whereas for raw leachate, a removal of only 361 mg NH_4^+-N was measured for a dosage of 40 ml. This should be attributed to the interactions of added Mg^{2+} and PO_4^{3-} ions with the already existing Mg^{2+} and Ca^{2+} ions in leachate and predominantly with organic molecules. The concurrence of TKN (TKN≈TN, as inferred from table 1) with ammonium-nitrogen suggests a removal of organic bound nitrogen.

The interaction with organics leading to co-precipitation becomes evident from figure 3b, which shows the decrease in TOC concentration in supernatant with the dosing of $MgCl_2$ and H_3PO_4. An overall reduction of about 25% of measured initial TOC concentration was observed in both cases, with a higher rate of removal in raw leachate, which can be again concluded from the slope. Table 2 presents the results of wet chemical analysis of the product from these experiments. The first glance at the TOC content of the two products says it all. NF permeate, which had less than half organic load in comparison to raw leachate, provides a product which also has lesser organic impurities (less than half) than the latter. Furthermore, the deviation in P/N ratio (by mass) is much larger in the product from raw leachate when compared to the theoretical value. This is likely due to the formation of aggregates of magnesium hydrogen phosphate associated with organic matter.

Table 2: Effect of matrix on product purity - results of wet chemical analysis

Product	NH_4^+-N content (mg/kg)	o-PO_4^{3-}-P content (mg/kg)	P:N (w/w)	TOC content (mg/kg)
Theoretical	57,047	126,197	2.21	-
NF permeate (P2), pH 8.75	40,556	102,222	2.52	4,058
Raw leachate (F), pH 8.75	37,120	126,087	3.40	8,281

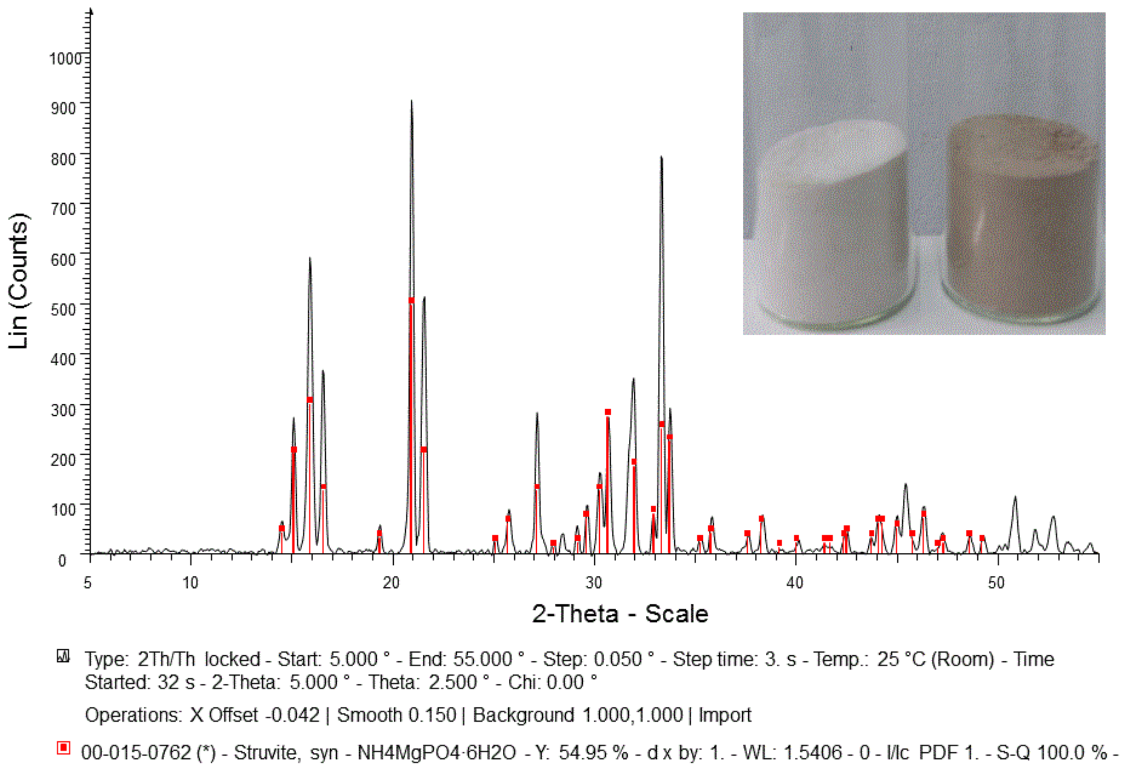

Figure 4: X-ray diffractogram of the precipitation product from raw leachate and photo of products from the two experiments

Figure 4 shows the X-ray diffractogram of the product from raw leachate and a photo of the products from the two experiments for visual comparison, the precipitate from raw leachate being slight brown in colour. The results from XRD were not different from one another for the two samples.

3.2 Effect of pH

Figure 5a shows the measured N-NH_4^+ concentration in the supernatant with the addition of $MgCl_2$ and H_3PO_4 at various pH. There was no effect of pH on the removal kinetics in the range studied. However, it is to be noted that pH has a significant role in the degassing of ammonia which was indicated by decreasing initial concentration with increasing pH (see legend of figure 5a).

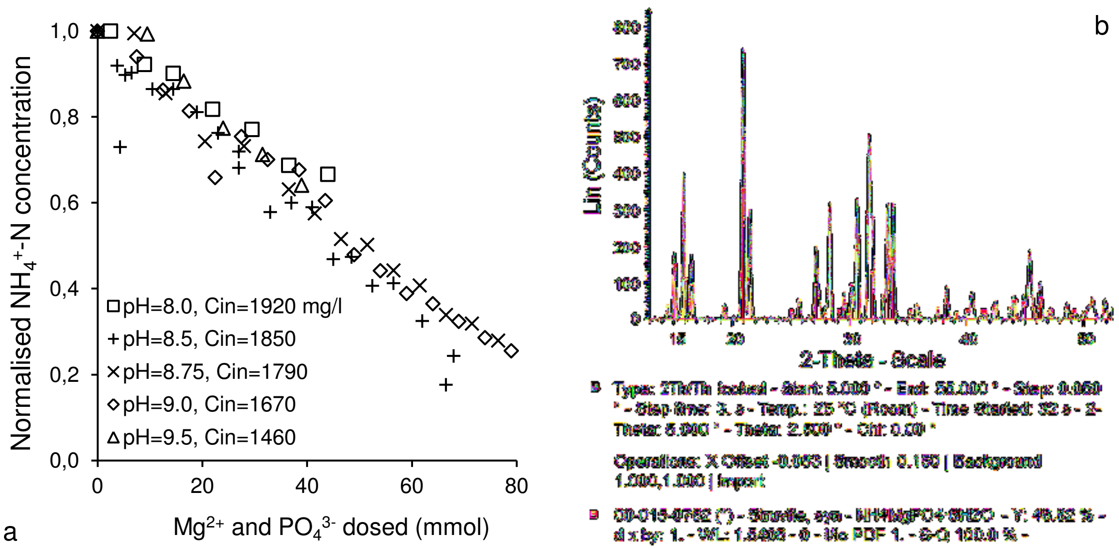

Figure 5: (a) Ammonium-nitrogen concentration in supernatant – effect of pH on the process kinetics, (b) X-ray diffractogram of product from NF permeate, precipitation at pH 9.0

Table 3: Effect of pH on product purity - results of wet chemical analysis

Product	NH_4^+-N content (mg/kg)	o-PO_4^{3-}-P content (mg/kg)	P:N (w/w)	TOC content (mg/kg)
Theoretical	57,047	126,197	2.21	-
pH 8.0	52,976	131,148	2.48	3,741
pH 8.5	45,000	101,111	2.25	3,126
pH 8.75	40,556	102,222	2.52	4,058
pH 9.0	43,333	103,889	2.40	3,789
pH 9.5	45,440	108,791	2.39	4,824

Figure 5b shows the X-ray diffractogram of the precipitate obtained from the experiment with NF permeate at pH 9.0. Although the effect of pH on kinetics was nil, there was a pronounced effect on the purity of the product. Table 3 shows the results from the wet chemical analysis of the dried product from various experiments.

It becomes clear that the product obtained at pH 8.0 has the closest composition to that of theoretical with an ammonium-nitrogen content of 93% and a phosphate-phosphorus content of 103% of theoretical contents. The slight excess of phosphorus can be attributed to the presence of magnesium hydrogen phosphate in the product. The lower nitrogen content in the products from higher pH is possibly due to the simultaneous stripping of ammonia at these conditions which was also reflected by the initial values indicated in the legend of figure 5a. Besides, the possibility for the existence of magnesium hydroxide in the precipitate formed at higher pH exists, which is however not characterised here. It is also to be noted that the TOC content of the product is not changing significantly with the change in pH.

4. Conclusion and outlook

The precipitate obtained from raw leachate was found to be slightly brown in colour, similar to previous research works [1] and has been proven to contain organic impurities reflected by a TOC content of about 8.3 $g/kg_{precipitate}$. The potential for employing such a product for agricultural purposes would be low due to the unknown risks that the impurities can pose. The product should then be dumped into a landfill site as suggested in earlier works [3]. However, the product from NF permeate was characterised to contain only about 3-4 $g_{TOC}/kg_{precipitate}$. The work proves that a co-precipitation of organics is inevitable (also reported in [1, 2]). A TOC reduction of about 25% was measured in the supernatant in both systems. For precipitation of MAP from NF permeate, a pH close to 8.0 was found to offer best results, with an ammonium-nitrogen content of about 93% of the theoretical value. The observed pH optimum is in agreement with pH optimum range of 8.0-10.0 found in the literature.

The potential to enhance the product purity further remains if raw leachate is directly treated with a nanofiltration step and the precipitation is made from its permeate. If the produced struvite is meant for agricultural reuse, indeed an integration with nanofiltration step would result in a better product with lesser risks or hazards when compared with precipitation from raw leachate. Besides the reduction of organic impurities, this would also facilitate zero heavy metal content in the product, as they would be retained in the retentate. Further research with direct nanofiltration and precipitation is necessary to comment on the process economics, as a nanofiltration step before reverse osmosis is likely to reduce the overall energy demand and result in greater flexibility for treatment.

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References

[1] Di Iaconi, C., Pagano, M., Ramadori, R., & Lopez, A. (2010). Nitrogen recovery from a stabilized municipal landfill leachate. Bioresource technology, 1732-1736.

[2] Kabdasli, I., Öztürk, Í., Tünay, O., Yilmaz, S., & Arikan, O. (2000). Ammonia removal from young landfill leachate by magnesium ammonium phosphate precipitation and air stripping. Water Science and Technology, 237-240.

[3] Li, X. Z., Zhao, Q. L., & Hao, X. D. (1999). Ammonium removal from landfill leachate by chemical precipitation. Waste Management, 409-415.

TA-O_10 Combination of biochar and clinoptilolite for nutrient recovery from liquid fraction of digestate

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Introduction and objectives

The liquid fraction of digestate contains nutrients, particularly nitrogen and potassium, predominantly in ionic forms. Ion exchange and adsorption technologies can potentially be applied to recover and concentrate these valuable nutrients.

In a number of publications, ion exchange with clinoptilolite has been shown to be very effective for the removal of ammonium from domestic wastewater. Furthermore, more than 90% removal efficiencies for both ammonium and potassium from source-separated human urine have been reported in the literature [1, 2]. Biochar has recently been suggested as bio-sorbent material for removing various types of contaminants from wastewaters [3]. Combination of clinoptilolite and biochar could be an alternative solution for the concentration of nutrients from the liquid fraction of digestate.

To the best of our knowledge, the combination of clinoptilolite and biochar to remove nutrients from the liquid fraction of digestate has not been investigated.

Methodology

The liquid fraction of pig slurry digestate was sampled from Morsø Bioenergy (Denmark). Clinoptilolite (particle size 1-3 mm) was preconditioned with 0.01 M NaCl in a continuous column system with 1 BV h^{-1} flow rate for 24 h. Holm oak biochar (pyrolysed at 650 °C) was used with particle size 1-4 mm.

Combination of clinoptilolite and biochar was tested in 3 different experimental setups. In the first experimental setup, biochar and clinoptilolite samples were homogenously mixed and filled in one column. The second and the third experimental setup had two columns each where clinoptilolite and biochar were filled in different columns separately. Columns were filled with the mass ratio of 1:1 for biochar and clinoptilolite.

The experiments were run for 72 hours and sampling was done every 24 h. Ammonium, orthophosphate, potassium, total nitrogen, total phosphorus and total organic carbon were analysed in all samples.

Results

The experimental work is in progress and the results will be shown at the conference.

References:

[1] Beler Baykal B, Kocaturk NP, Allar AD, Sari B (2009). The effect of initial loading on the removal of ammonium and potassium from source separated human urine via clinoptilolite. Water Science and Technology 10(60), 2515-2520.

[2] Kocaturk NP, Beler Baykal B (2012). Recovery of plant nutrients from dilute solutions of human urine and preliminary investigations on pot trials. Clean- Soil, Air, Water 40(5), 538-544.

[3] Lehmann J and Joseph S (2009). Biochar for Environmental Management: Science and Technology. Earthscan Ltd, London, UK.

TA-O_11 Wood ashes – A new fertilizer for agriculture

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1. Objectives

Ashes generated by wood-burning industries in Switzerland are currently discharged due to their elevated trace element concentrations that exceed the authorized thresholds for application on agricultural lands as recycling fertilizer. This is a significant loss of natural nutrients, especially as the amount of ashes is growing from the increased interest in renewable green energy. The objectives of this study were to determine (i) the mineralogical and physicochemical composition of wet wood bottom ashes from the wood burning power station Enerbois (Canton of Vaud, Switzerland) and (ii) their possible application in agriculture as a potassium fertilizer.

2. Methodology

2.1 Sampling and analysis of ashes

Wet wood bottom ashes were sampled in 2011 during a four week period that consisted of five to seven daily samples (500 g each). Ashe samples were dried at 40°C and sieved (2 mm screen). Total macro- microelements and trace metal elements (TME) content were determined by ICP-AES after hydrofluoric and perchloric acid digestion (Ciesielski et al. 1997). Sequential extraction procedure (Rauret et al. 2000), x-ray diffraction analyzes (XRD) and scanning electron microscope analyzes (SEM) were used to study the chemical speciation and mineralogical composition of macroelements and TME contained in ashes.

2.2 Greenhouse experiment

Greenhouse experiment was conducted by Agroscope in Changins to study the agronomic effects of ashes as a source of potassium (K) on sunflower, which is a very high demanding K crop. Each pot contained 2 kg dry soil and only one plant. The soil used was a clay soil with 53.8% clay and a slightly acidic pH of 6.5. Total soil content of nitrogen (N), phosphorus (P), K and magnesium (Mg) was 3.4, 0.94, 19.8 and 12.4 g / kg DM, respectively. Soil moisture was maintained at about 70% of field water holding capacity with deionized water. The temperature during this trials was maintained between 20-25°C.

Four treatments were tested: (i) "*Control*" without ash and fertilizers; (ii) "*Ashes*" consisting of application of ash only, which would substitute for K fertilizer; (iii) "*NPMg* + *Ashes*" with N, P, and Mg as chemical fertilizers and (iv) "*NPMg* + *K*" with K, N, P, and Mg, all applied as chemical fertilizers. Each treatment was repeated three times in a randomized complete block design. The amount of nutrients and ashes was calculated to meet sunflower K requirements according to Swiss guidelines for fertilization practices (Sinaj et al. 2009). Chemical fertilizers used were ammonium nitrate (NH_4NO_3), triple superphosphate [Ca $(H_2PO_4)_2.H_2O$], magnesium chloride ($MgCl_2$), and potassium chloride (KCl). Sunflower total dry matter and macro-microelements content were measured, and Student's t-test and Fischer's LSD tests were used to assess the effect of 4 treatments.

3. Results and discussion

3.1 Characterization & speciation

Characterization

The ashes of the wood burning power station Enerbois are very alkaline, which is to be expected due to their high levels of Ca and Mg (Table 1). XRD analyzes indicated that Ca occurs mostly as carbonate [(*calcite*: $CaCO_3$)] and hydroxyl [*portlandite*: $Ca(OH)_2$], which are not very reactive forms explaining the slower action of ashes on changing soil pH compared to quicklime (CaO) (Maltas and Sinaj 2013).

Ashes organic matter (OM) and N content were low due to burning during the combustion process. Ashes are an important source of K and to a lesser extend of P (Table 1). They also contain significant amount of microelements [aluminum (Al), iron (Fe), manganese (Mn) and boron (B)] and TME such as copper (Cu), nickel (Ni), zinc (Zn) and lead (Pb). TME originate from Swiss soil (Luster et al. 2006), and subsequently, concentrate in ashes during wood combustion. As a results, Cu and Ni concentrations in the ash exceeds the Switzerland authorized thresholds for application on agricultural lands as recycling fertilizer (Table 1).

Table 1: Chemical characteristics of wood ashes and Switzerland authorized thresholds for recycling fertilizer application on agricultural lands. Coefficient of variation is indicated in parenthesis.

Wet wood bottom ashes			
pH		13.2 (1%)	
Organic matter (g/kg)		15 (55%)	
Total macroelements (g/kg)		**Total microelements (mg/kg)**	
Calcium	281.3 (2%)	Aluminum	17300 (7%)
Potassium	67.4 (9%)	Iron	12175 (3%)
Magnesium	6.5 (5%)	Manganese	7550 (7%)
Phosphorus	9.2 (9%)	Boron	147 (12%)
Nitrogen	0.07 (27%)	Molybdenum	1.1 (7%)
Total TME (mg/kg)		**Switzerland authorized thresholds**	
Zinc	178 (14%)		400
Copper	110 (21%)		100
Nickel	52 (7%)		30
Lead	21 (53%)		120
Cadmium	<0.6		1
Mercury	<0.02		1

Speciation

The sequential extraction methods was used to study the speciation of each element. The results from this study (Fig. 1) showed that most of macronutrients (K, Ca and Mg) in wet ashes are immediately available (fraction 1) and slowly-available (fractions 2 and 3). More than 50% of K was present in the fraction 1 (immediately available). Phosphorus was mainly present in a slowly or non-available forms, associated perhaps with Ca as calcium phosphate, which forms at high temperatures during the combustion.
Zn, Cu, Ni and Pb (Fig. 1) were almost absent from the fraction 1 (readily available). More than 50% of Zn, Cu & Ni were present in fractions 2 and 3 (slowly-available) while almost the entire Pb was in the residual fraction (considered not-available). The speciation analysis indicates that the risk related to TME in wet ashes in the short term is negligible as long as the pH of the receiving environment remains neutral to basic.

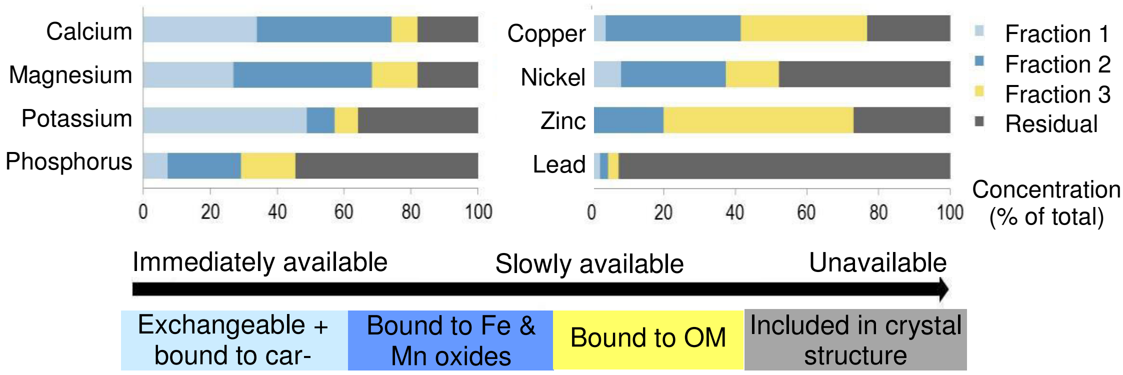

Figure 1: Speciation of macroelements and trace metal elements from sequential extraction procedure (from Maltas and Sinaj, 2014).

3.2 Effect of wood ashes on sunflower biomass production & elements absorption

Biomass production

Total dry matter and nutrients uptake by sunflower are shown in Table 2. Sunflower fertilized with ashes only produced more dry matter than control treatment though not significant ("*Control*" vs. "*Ashes*", p>0.05). However, ashes added to the NPMg fertilizers produced significantly more total dry matter compared to K added to the NPMg ("*NPMg-Ashes*" vs. "*NPMg-K*", p<0.05). This positive effect of ashes has been observed on several crops: oats, wheat, fescue, spinach, peas, corn, poplar and soya (Demeyer et al., 2001). Increased biomass production has been related to the liming effect of ashes especially in acidic soil and/or macro-microelement input.

Table 2: Effect of treatments on total dry matter produced and total amount of macronutrients and TME absorbed by sunflower. Different letters indicate significant differences at Fischer's LSD test (p<0.05).

Treatments	Total dry matter	N	P	K	Ca	Mg	Zn	Cu	Ni
	g/plant	mg/plant							
Control	23.9 c	236.3 c	86.0 b	514.4 c	475.9 b	86.4 b	0.86 ab	0.34 a	0.11 a
Ashes	26.2 bc	257.7 c	96.8 b	704.2 b	514.2 b	58.8 b	0.64 b	0.39 a	0.06 b
NPMg+Ashes	37.6 a	706.1 a	123.9 a	1110.1 a	1051.0 a	168.6 a	0.58 b	0.44 a	0.12 a
NPMg+K	31.0 b	605.2 b	85.0 b	1124.5 a	1117.3 a	187.1 a	0.99 a	0.42 a	0.13 a

Macroelements and TME absorption

As expected, the absorption of K by sunflower was higher in the "ashes" treatment than "control", following K supply by ashes. In addition, the quantities of absorbed K were not significantly different between the two forms of applied K (K - ashes vs. K - fertilizer) (Table 2). Furthermore, the addition of ashes improved significantly the absorption of N and P (Table 2), despite the fact that ashes are practically free of N. The beneficial effect on N uptake is likely related to the effect of liming on soil organic matter mineralization (Maltas and Sinaj, 2013). The positive effect of ashes on P absorption could be explained by the significant amounts of exchangeable P provided by the ashes (Fig. 2) and the positive effect of liming on soil P availability in acidic soils, generally attributed to the dissolution of Al and Fe-phosphates.

The absorption of Mg by the sunflower decreased significantly in the presence of ashes (Table 2) despite the addition of Mg by the ashes. The antagonism between Ca and Mg could explain this effect.

In the presence of ashes, sunflower absorbed less Zn and Ni. This effect was significant in limiting NPMg +K conditions but not in non-limiting conditions (Table 2). This can be explained by the lower amounts in the soil of exchangeable Zn and Ni observed when acidic soil is limed.

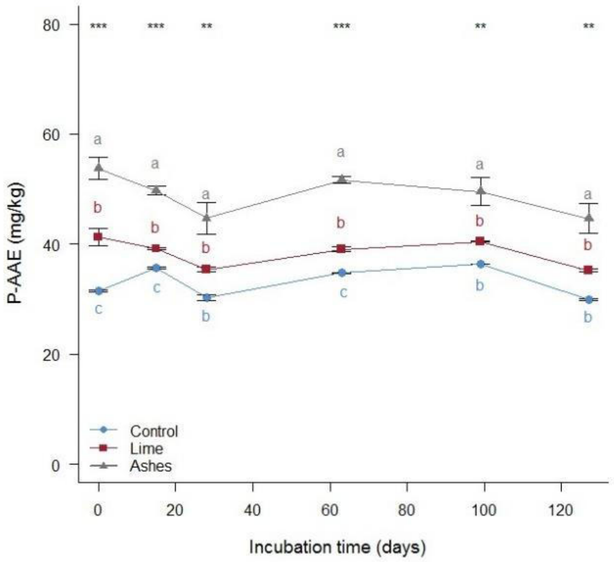

Figure 2: Effects of lime and ashes addition on soil exchangeable P (P-AAE) content. Soils were incubated for 4 months. Different letters indicate significant differences at Fischer's LSD test (**: p<0.01; ***: p<0.001).

Long-term risk of TME contamination

Defining maximum quantities of TME (kg/ha) supplied can prevent long-term risk of TME accumulation in soil. Maximum quantities of recycling fertilizer authorized for manure have been established at a value of 25 t/ha (based on dry matter) over a 3 years period (annex 2.6, ch 3.2.2 of ORRChem). Based on K content of wood ashes (Table 1) and crops K requirements (Sinaj et al., 2009), the amount of ashes applied should not exceed 5 t/ha (based on dry matter). Wood ashes are highly concentrated in K and would require smaller application rates. The suggested 5 t/ha of ashes would supply less TME compared (i) to the established maximum rates of 25 t/ha recycling fertilizer and (ii) quantities generated by conventional agricultural compost (used at the maximum dose of 25 t/ha). Based on the characteristics of Swiss agricultural compost given by Kupper and Fuchs (2007), ashes provide 92 and 33 times less Pb and Ni respectively compared to composts for the same amount of supplied K.

Despite high Ni and Cu contents above the current Switzerland authorized thresholds for recycling fertilizers, content of K limits the amount of ashes to be applied. This amount would be too low to increase significantly TME concentration in soil, given the initial soil concentration. This reinforces the idea of a low short-term risk of using ashes in agriculture and raises the question of the relevance of TME thresholds currently authorized for ashes. These thresholds should take into account the doses for each applied manure, which are calculated based on critical nutrients (N, P and K). Other studies have also highlighted that wood ashes occasionally used at agronomic doses, do not pose short or long-term environmental risk (Demeyer et al., 2001, Hébert and Breton 2008).

4. Conclusion and outlook

The results of this study indicate that wood ashes coming from the wood burning power station Enerbois (i) present a low risk to soils and crops and could be used as potassium fertilizer on acid soils, (ii) contain TME but in “slowly” or “unavailable” forms and (iii) further research is needed on the effects of these ashes on acid and neutral soils at field/farm level.

Acknowledgements

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References

[1] Demeyer, A., Voundi Nkana, J.C., Verloo. M.G. (2001): Characteristics of wood ash and influence on soil properties and nutrient uptake: an overview. Bioresource Technology, 77: 287-295.
[2] Halvin, J.L., Beatoon, J.D., Tisdale, S.L., Nelson, W.L. (2005): Soil fertility and fertilizers: an introduction to nutrient management. Pearson Prentice Hall. Upper Saddle River, NJ. 515 p.
[3] Hébert, M., Breton, B. (2008): Recyclage agricole des cendres de bois au Québec- Etat de la situation, impacts et bonnes pratiques agroenvironnementales. Agrosolutions, 19: 18-33.
[4] Ciesielski, H., Proix, N., Sterckeman, T. (1997): Détermination des incertitudes liées à une méthode de mise en solution des sols et sédiments par étude interlaboratoire. Analusis, 25: 188-192.
[5] Kupper T., Fuchs, J. (2007): Compost et digestat en Suisse. Connaissance de l’environnement 743. Office fédéral de l’environnement, Berne. 124p.
[6] Luster, J., Zimmermann, S., Zwicky C. N., Lienermann, P., Blaser, P. (2006): Heavy metals in Swiss forest soils: Modification of lithogenic and anthropogenic contents by pedogenetic processes, and implications for ecological risk assessment. Geological Society Special Publication, 266: 63-78.
[7] Maltas, A., Sinaj, S. (2011): Intérêts agronomiques des cendres humides de la central Enerbois. Agroscope. 28 p.
[8] Maltas, A., Sinaj, S. (2013): Effets des cendres de bois de la centrale Enerbois sur les propriétés du sol, le rendement des cultures et la qualité des récoltes. Agroscope. 63 p.
[9] Maltas, A., Sinaj, S. (2014): Wood ashes: A new fertilizer for Swiss agriculture. Agrarforschung Schweiz, 5: 232-239.
[10] Rauret, G., Lopez-Sanchez, J.F. , Sahuquillo, A. , Barahona, E. , Lachica, M. , Ure, A.M., Davidson, C.M., Gomez, A., Luck, D., Bacon, J., Yli-Halla, M., Muntau H., Quevauviller, P. (2000): Application of a modified BCR sequential extraction procedure for the determination of extractable trace metal contents in a sewage sludge amended soil reference material, complemented by a three-year stability study of acetic acid and EDTA extractable metal content. Journal of Environmental Monitoring, 2: 228-233.
[11] RS.814.81. (2011): Chemical Risk Reduction Ordinance (ORRChem). 18 May 2005 (Status as of 1 December 2014).
[12] Sinaj, S., Richner, W., Flisch, R., Charles, R. (2009): Données de base pour la fumure des grandes cultures et des herbages (DBF-GCH). Revue Suisse d’Agriculture, 41: 1-98.
[13] Smith, S.R. (1994): Effect of soil pH on availability to crops of metals in sewage sludgetreated soils. I. Nickel, copper and zinc uptake and toxicity to ryegrass. Environmental Pollution, 85(3): 321-327.

TA-O_12 The influence of raw materials and storage time in composts maturity

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1. Objectives

The present work aimed to:

- assess the maturity degree of organic composts/fertilizers actually used in agriculture,
- evaluate the influence of the compost source and the storage time in the maturity degree and
- study the compatibility of three methodologies used to assess the maturity degree.

2. Methodology

Four types of commercialized composts, from different sources, were analyzed: without storage (T0) and with 24 months of storage (T1). The specifications of the compost production are described in Table 1.

Table 1: Operational parameters of compost production.

Operational parameters	Composts			
	A	B	C	D
Raw Material	MSW	MSW + Green wastes	MSW + Green wastes	Poultry manure + ash + carcass meal
Type of Collection	Unseparated	Unseparated	Separated	Separated
Type of process	Reactor	Windrow	Reactor	Windrow
Time of process (months)	6	6	6	4

MSW- Municipal solid waste

The physical-chemical characterization of composts was made according with standard procedures, as reported by Silva et al. [1]. Humic-like substances (HS), extracted from the composts, were also tested. As a control, a commercial liquid fertilizer (Humistar®) was used. HS extraction was carried out according with the methodology described in the Spanish official method for the analysis of fertilizing organic products (R.D. 1110/12, July 1991). The maturity was assessed by Germination Index (GI), Growth Test (GT) and Cation Exchange Capacity (CEC). All tests were performed in triplicate except for GI, for which seven assays were done. The methodologies were:

- GI: Composts aqueous extract or dialysed HS extract were analyzed using cress seeds (*Lepidium sativum)* and radish seeds (*Raphanus sativus*) following the procedure described by Tiquia et al. [2].
- GT: Composts were tested using radish seeds in pots with a mixture of sand, perlite and peat [3].
- CEC: The air-dried composts were treated with ammonium acetate [4].

3. Results and discussion

3.1 Physical-chemical characterization of organic composts/fertilizers

The physical-chemical properties of the composts are summarized in Table 2.

Table 2: Physical-chemical characterization of composts at different maturity stages.

Composts	Moisture (%)		OM (% dry matter)		pH		EC (mS/cm)	
	T0	T1	T0	T1	T0	T1	T0	T1
A	53.7	61.4	64.5	47.8	8.0	8.4	4.9	3.4
B	13.1	8.7	44.5	44.7	8.4	8.3	6.5	6.7
C	13.7	10.1	52.3	50.5	8.4	8.2	8.1	7.1
D	40.0	23.0	33.2	28.2	9.0	8.7	5.6	4.0

OM - Organic matter; EC - Electric Conductivity

In general, the physical-chemical parameters range widely between values, among the analyzed composts, even for those produced from MSW. Briefly, moisture content varied from 13% to 54% for all composts at time T0 and, except for compost A, these values decreased during the storage time. The organic matter content had a rather wide variation, with compost D showing the lowest value at both times. In general, the composts registered an organic matter content decrease over time; however compost A showed the highest decrease. Composts had a slightly alkaline pH (between 8.0 and 9.0). All composts presented electrical conductivity higher than 4 mS/cm, which is considered to be the tolerable conductivity by plants of medium sensitivity [5]. This was not verified for compost A at time T1. Data suggests that significant changes occurred on composts properties during the storage. But, as the pH is the parameter that does not change over storage time, it cannot be used as an indicator of the compost maturation.

3.2 Maturity degree of organic composts/fertilizers

Although the age and maturity might differ amongst the composts analyzed, these products were sold or supplied by the producers as being ready-to-use. Throughout this report, the different compost evaluation was done based on such assumption.

Overall, higher maturity degree is related to higher values for the three tests. For example, values of GI below 50% characterize an immature compost indicating that it has phytotoxic characteristics [6]. According with the GI obtained (Figure 1), for time T0, the organic composts were potentially phytotoxic, indicating a low maturity degree. Composts without storage (T0) and with the similar source (MSW) exhibited different GI, between 0% and 44%. Compost D showed the highest GI value (72%) for cress seed, which can be considered as a mature compost. Overall, the GI values were in the same range as reported by Lasaridi [5], which presented values between 25% and 151%. The phytotoxicity exhibited by some composts may be related with their high EC values (e.g. compost C).

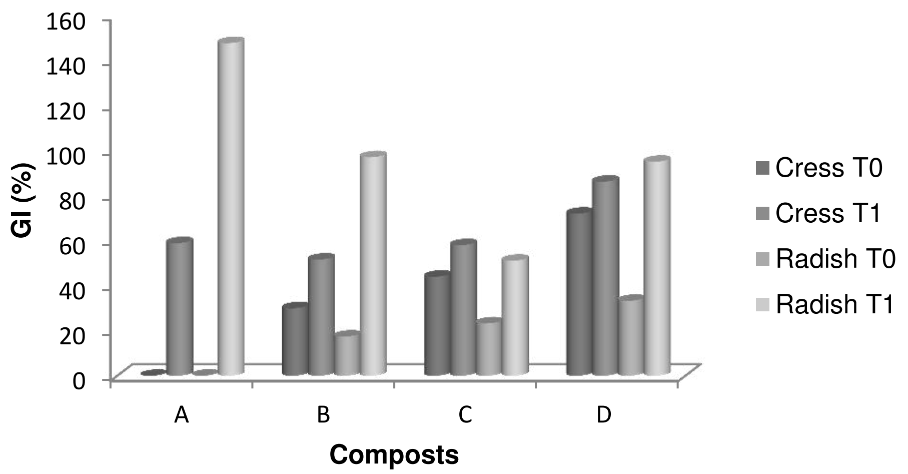

Figure 1: Germination index of cress and radish seeds in composts (A, B, C, D) at time T0 and T1

A similar trend was observed for stored compost (T1). The composts with similar raw material revealed different GI values. These results seem to indicate that the compost source does not influence the maturation degree, and that probably other parameters may influence the maturation, such as the type of collection and process. Nevertheless, for all composts the GI increased, mainly for compost A that showed a clear improvement in the properties with the storage time. At time T0, this compost *had* significant *inhibitory* effect on the seeds germination; however at time T1 the germination levels were higher than 100%. This may be explained by the fact that this compost was not stabilized at time T0 and the biodegradation occurred during the storage, which was confirmed by the highest organic matter decrease. These results indicated that the phytotoxic organic compounds are gradually eliminated during the storage time. Among the two seeds – cress and radish – the cress proved to be a more sensitive plant when compared to radish as reported before in the literature [7]. The germination index results for dialysed HS extract and commercialized liquid fertilizer are summarized in Table 3.

Table 3: Germination index (GI) for Humic Substances (HS) extracts from composts with different storage time and for the commercial liquid fertilizer.

HS extracts	**GI_cress (%)**		**GI_radish (%)**	
	T0	T1	T0	T1
A	141.0	139.6	132.6	165.0
B	nd	105.2	nd	106.3
C	113.9	105.9	138.9	115.0
D	nd	99.3	nd	105.3
Commercial liquid fertilizer	nd	153.9	nd	94.5

nd – not determined

All HS extracts presented GI values higher than the composts. These values are between 99% and 165%. In this case, the data is similar for both evaluated periods (T0 and T1). Results seem to indicate that some phytotoxic compounds are reduced with the extraction process and further dialyse. In fact, the immature composts may contain high concentrations of salt, high amounts of free ammonia, certain organic acids or other water-soluble compounds which can limit seed germination and root development [8]. These results suggest that some of these compounds may be reduced with the extraction and/or dialysed processes and the HS extracts may be applied as fertilizer. This could be a way to valorize composts with low quality and to avoid the HS extraction from natural resources. To further evaluate the maturity, the GT and CEC evaluation were carried out at time T1 (Table 4).

Table 4: Growth test (GT) and Cation Exchangeable Capacity (CEC) for composts with storage (T1).

Composts	**GT (%)**	**CEC (Cmol/kg dry matter)**
A	87.1	79.3
B	62.0	111.8
C	68.8	71.8
D	48.9	25.9

It was found that composts A and D showed the highest (87%) and lowest (49%) values for growth test, respectively. It can be said that the highest EC values of compost C did not inhibit the radish plant growth as happened with the seed germination. This may be explained by the organic matter content at time T1. The organic matter content is very important to soil and thus to the plant growth because increase the water holding capacity, act as buffer against pH changes, improve the aggregation between silt and clay particles and increase the retention of metal micronutrients (limiting their leachability) [9]. It is known that high humified organic matter should increase CEC. In fact, the different capacity of composts to supply essential nutrients, which is confirmed by the highest CEC of MSW composts, could explain the highest growth of the radish in these composts. The high CEC values could be attributed to the accumulation of materials bearing the negative charge, such as lignin derived products, and the increase of carboxyl and/or phenolic hydroxyl groups, which may be produced by the humification occurring during the composting process. In fact, according to Bernal et al. [8] humification produces functional groups, increasing organic matter oxidation and a consequent increase of CEC. This is the key reason why CEC is a parameter that has been used to evaluate the maturity of compost. The values obtained for the CEC are between 26 and 112 Cmol/kg of dry matter, with composts B and D showing the utmost and the lowest value, respectively. Bernal et al. [10] found values of CEC for composts between 36 and 236 Cmol/kg of compost. The same trend was verified in all composts except for compost D, with CEC lesser than 36 Cmol/kg of dry matter. Again, it was observed different profile among the composts with similar raw material.

3.3 Compatibility of three methodologies used to assess the maturity degree

The maturity degree can be evaluated using several methodologies; however in our case the results did not show the same trend. Comparing the data obtained with the 3 methodologies, at time T1, it was observed that compost A had the highest GI and GT values, and compost B the highest CEC value. Therefore, maturity degree may be evaluated by germination or growth tests, but the test plants have to be chosen with care. In fact, it appears that more consistent results can be achieved with GT, CEC and organic matter content evaluation. Probably the use of the compost directly in the GT, with longer test periods, and the direct contact between the test seeds and the compost will be more accurate to overcome the phytotoxicity effects. Perhaps, this test is the one that simulate more accurate the real utilization of these products.
Overall, each test used may influence the evaluation of the maturity degree, indicating the need to perform several tests to assess the effectiveness of the compost maturity.

4. Conclusion and outlook

The composts showed low maturity degree, which may cause germination inhibition. Data suggests that maturation is independent of the compost source and that probably other production features may influence the maturation, namely the type of collection, effectiveness of wastes separation or the type of composting process: open vs. container. Nevertheless, the storage time endorse an increase of maturation. Although the HS extraction from the compost may increase the fertilizer preparation complexity, it was shown that was an effective process for toxic compounds removal from the fertilizer products. The use of these HS extracts may be a solution for valorization composts with low quality and to avoid the HS extraction from natural resources. The results indicate that the various methodologies did not show the same trend, strengthening the need to further study the factors that are behind these observations.

Acknowledgements

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References

[1] Silva, M. E.; Lemos, L.T.; Cunha-Queda, A.C.; Nunes, O.C. (2009): Co-composting of poultry manure with low quantities of carbon-rich materials, Waste Management & Research 27, 119-128.
[2] Tiquia, S.M.; Tam, N.F.Y.; Hodgkiss, I.J. (1996): Effects of composting on phytotoxicity of spent pig-manure sawdust litter, Environmental Pollution 93, 249–56.
[3] Jodice, R. (1989): Parametri chimici e biologici perl a valutazione della qualità del compost, in *Proc. of the Compost Production and Use –International Symposium, S. Michelle all'Adige*, 20-23 June, 363-384.
[4] Sumner, M.E., Miller, W.P. (1996): Cation exchange capacity and exchange coefficients. In: Bartels JM, editor. Methods of Soil Analysis. Part 3. Chemical Methods Madison, Wisconsin: Soil Science Society of America; 1201–30.
[5] Lasaridi, K.; Protopapa, I.; Kotsou, M.; Pilidis, G.; Manios, T.; Kyriacou, A. (2006): Quality assessment of composts in the Greek market: The need for standards and quality assurance, Journal of Environmental Management. 80, 58-65.
[6] Zucconi, F.; Pera, A.; Forte, M.; De Bertoldi, M. (1981): Evaluation toxicity of immature compost maturity, Biocycle 22, 54-57.
[7] Oleszczuk, P. (2010): Testing of different plants to determine influence of physico–chemical properties and contaminants content on municipal sewage sludges phytotoxicity, Environ. Toxicol., 25, 38–47.
[8] Bernal, M.P.; Albuquerque, J.A.; Moral, R. (2009): Composting of animal manures and chemical criteria for compost maturity assessment. A review, Bioresource Technology 100, 5444-5453.
[9] Moreira, R.; Sousa, J.P.; Canhoto, C. (2008): Biological testing of a digested sewage sludge and derived composts, Bioresource Technology 99, 8382–8389.
[10] Bernal, M.P.; Paredes, C.; Sanchez-Monedero, M.A.; Cegarra, J. (1998): Maturity and stability parameters of composts prepared with a wide range of organic wastes, Bioresource Technology 63, 91-99.

TA-O_13 Reduction of ammonia emissions by acidification and injection of cattle slurry applied to perennial grassland

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1. Objectives

Ammonia (NH_3) losses after field application of slurry may be high and are responsible for several environmental problems like deposition of nitrogen (N) to N-sensitive ecosystems. In Denmark a new officially permitted system exists to lower the pH of slurry by adding concentrated sulphuric acid (H_2SO_4) shortly before application in the field. Lowering the pH suppresses the dissociation of NH_4^+ to NH_3 + H^+ and thus lowers NH_3 volatilization. To test different praxis available NH_3 abatement techniques, a collaboration project between Kiel University, Aarhus University and the Danish Knowledge Centre for Agriculture was established. In a field trial injection and acidification of cattle slurry were compared to bandspreading.

2. Methodology

Two sites with similar climatic conditions but contrasting soils were chosen to test interactions between soil conditions and applicability of the examined methods.The field trial was conducted in 2012 and 2013 at 2 perennial grassland sites at the German Coastal Marsh (site A; 40% clay soil) and on a sandy soil (site B) in Southern Denmark. Following treatments were tested: 1. Control, 2. Bandspreading, 3. Injection A (0.175 m slot distance), 4. Injection B (0.35 m slot distance), 5. Acidification A (pH 6.5), 6. Acidification B (pH 6.0). Slurry was applied plotwise at each application date (site 1: n = 8 and site 2: n = 6, respectively) and application rate was determined total ammoniacal nitrogen (TAN) based. To calculate N yield response, increased mineral fertilization treatments (CAN) were established. N-uptake (not at site A in 2012) and dry matter yield was determined and ammonia emissions were measured by a combined method of calibrated chamber measurement and passive flux samplers in each plot [1]. The field trial was set up as a fully randomized block design (4 replicates) with plots of 9m x 9m. Between the plots guard areas of the same dimension were established to reduce the influence of NH_3 drift between plots, resulting in a chess-board design. At site A N_2O emissions were determined by a static chamber approach (Hutchinson and Mosier 1981) in following treatments:1st year (April 2012 - March 2013): 1. Control, 2. Mineral N (320 kg N ha^{-1}) 3. Bandspreading, 4. Injection A, 5. Acidification A and 6. Acidification B. 2nd year (April -October 2013): 1. Control, 2. Bandspreading, 3. Injection A, 4. Injection B and Acidification B. Slurry was taken site specific in 2012 and the same slurry for both sites was used in 2013; pH of the raw slurry was 7.2 – 7.4.

3. Results and discussion

3.1 Slurry and weather

At both sites slurry NH_4^+ concentration varied between 1.4 and 2.9 kg N ton^{-1} and dry matter varied between 5.7 and 11.6% (Tab. 1). For the 1st application dates temperatures were below 10°C, while wind velocity was varying but in average higher at site A due to exposure to the North Sea and dense hedges at site B. The average acid amount needed for acidification of the slurry to pH 6.5 and 6.0 was 2.8 and 4.4 l 96% H_2SO_4 t^{-1}.

Table 4: Slurry characteristics and mean weather conditions over measurement period for all slurry application dates.

Site	Year	Application number	$TAN_{applied}$ [kg N ha^{-1}]	Slurry volume [ton ha^{-1}]	DM concentration [%]	NH_4^+-concentration [kg N ton^{-1}]	Total N concentration [kg N ton^{-1}]	Wind [m s^{-1}]	Temperature [°C]
Site B	2012	1	80	40.0	7.4	2.0	3.2	3.7	6.8
		2	50	25.5	7.2	2.0	3.3	1.7	17.9
	2013	1	80	50.0	5.9	1.6	2.6	4.7	3.3
		2	50	29.4	6.2	1.7	3.1	2.7	14.1
		3	40	25.0	6.1	1.6	2.6	2.1	18
		4	30	17.6	5.8	1.7	2.5	1.9	16.4
Site A	2012	1	80	32.8	7.2	2.4	3.0	2.8	6.4
		2	80	33.3	11.6	2.4	5.0	3.8	17.3
		3	60	21.6	10.5	2.8	5.2	5.9	15
		4	60	21.1	8.5	2.9	4.9	4.6	14.5
	2013	1	80	57.1	5.8	1.4	3.4	5.5	9.4
		2	80	50.0	6.1	1.6	3.4	6	14.6
		3	60	37.5	6.1	1.6	2.6	3.6	16.5
		4	60	37.5	5.7	1.6	2.6	2.9	15.6

3.2 Ammonia losses

NH_3 losses ranged from 0.5 – 39.9% of $TAN_{applied}$ and the average loss of all application dates was 14% of $TAN_{applied}$. Average NH_3 losses of the reference method band spreading were for site A 19.1% and 15% and for site B 11.0% and 7.4% in 2012 and 2013, respectively. The lower level

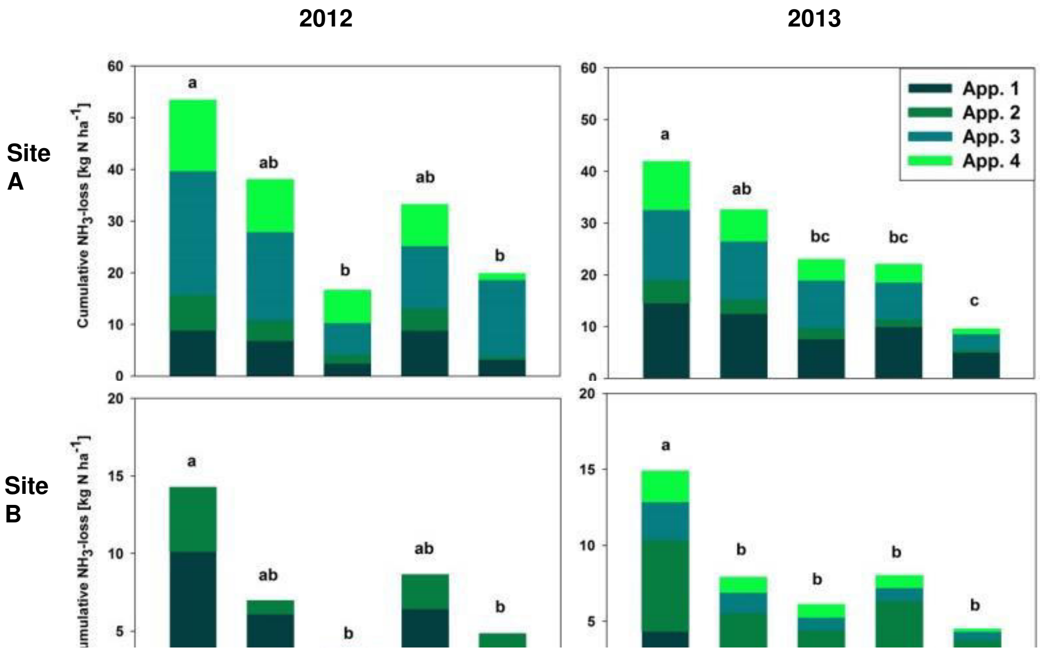

Figure 2: Cumulative absolute ammonia losses of single applications at site A and B in 2012 and 2013 for all slurry treatments. Different letters indicate significant differences within one year and site (ANOVA, ...LSD test, $p < 0.05$).

of losses found at site B might be caused by higher wind velocities at site A. Maybe also faster infiltration of slurry on sandy soil might have played a role for lower average losses of NH_3 at site B (Sommer et al. 2006). Significantly highest losses of NH_3 always were found for bandspreading

(Fig. 1); On average, injection A and B and acidification A and B showed reduced losses of 31.4%, 60.6%, 42.2% and 68.9% compared to bandspreading, respectively.

3.3 N_2O emissions

The emissions of N_2O from the first experimental year were on a low level between 0.6 and 1.8 kg N_2O-N ha^{-1} (0.12 - 0.38% of N_{total}), while slurry treatments showed no significant differences. The mineral reference (CAN, 320 kg N ha^{-1}) showed significant highest N_2O emissions (Fig. 2). In 2013 overall losses were on a higher level ranging from 0.4 to 4.2 kg of N_2O-N ha^{-1} (0.12 - 0.38% of N_{total}). Highest losses could be observed from Injection B and lowest losses from the control. But also Injection A and Acidification B showed higher N_2O emissions, though not significant different to control and band spreading. But this may indicate that injection of slurry or lowered/saved ammonia losses not always will enhance denitrification processes. But in specific situations N_2O emissions may be higher. Here a higher concentration of nitrogen in the open slot from injection B may be causing enhanced denitrification. The level of N_2O emissions in this trial is low compared to other studies.

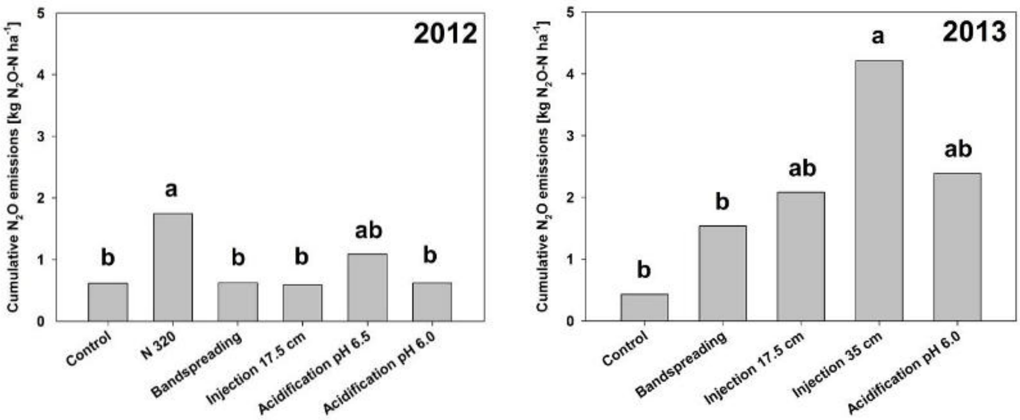

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Figure 2: Cumulative N2O emissions for selected treatments in 2012 and 2013 at Waygaard. Different letters ……… indicate significant differences between the treatments within one year (ANOVA, LSD test, $p<0.05$).

3.4 Dry matter and N yield

By tendency dry matter yields increased with saved NH_3 losses, but considering both sites and all application dates, slurry treatments did not differ significantly in dry matter yield.
Cumulative N uptake (Tab. 2) at site B in 2012 was highest for Acidification B, while band spreading showed lowest N uptake. In 2013 no significant difference in N uptake could be observed from the slurry treatments at site B. At site A in 2013 band spreading, injection A and B and acidification A showed no significant difference in N uptake while significantly highest N yield was observed from acidification B. This implies, that not only saved N from lower NH_3 losses causes higher N uptake from the plants. After strong acidification on Marsh soil (pH 7.3) a higher mobility of N and also other nutrients could be influencing nutrient uptake of the grass.

Table 2: Cumulative Nitrogen uptake [kg N ha^{-1} a^{-1}] of site A in 2013 and site B in 2012 (2 cuts) and 2013. Letters in brackets indicate significant differences (ANOVA, LSD test, $p<0.05$).

Site and year	Band spreading	Injection A	Injection B	Acidification A	Acidification B
Site B, 2012	148.0 (b)	152.5 (ab)	148.0 (b)	160.3 (ab)	171.9 (a)
Site B, 2013	208.5 (a)	207.6 (a)	196.6 (a)	211.1 (a)	197.2 (a)
Site A, 2013	308.8 (b)	322.8 (b)	329.2 (b)	311.5 (b)	428.4 (a)

4. Conclusion and outlook

Ammonia losses may be high after field application with band spreading, but already practice available techniques could help reducing this problem. All measures to lower NH_3 losses proved to be effective under the same field conditions and are already available for practice. Under sandy site conditions injection methods proofed to work effective while at the marsh site injection sometimes yielded lower NH_3 abatement success under dry soil conditions. Strong acidification of slurry on a marsh soil with high pH may also enhance nutrient uptake by the plants. Reducing NH_3 losses by more than 50% with acidification or injection with wide slot distance compared to the actual good practice of band spreading are realistic. But restrictions in both methods may occur. Depending on slurry type (pig / cattle / anaerobic digestate) different amounts of acid are necessary. Especially anaerobic digestates from biogas production have a demand of approx. 8 – 10 l H_2SO_4 t^{-1}. Therefor the crop sulphur demand and also liming to neutralize the applied acid have to be considered. Efficiency of slurry injection will also be determined by soil conditions while under conditions like stony or dry soil penetration by the injectors may be disturbed.

Acknowledgements

This work was funded by the “Aktivregion Nordfriesland Nord” of the Federal State Schleswig-Holstein and the FNR (Agency of renewable resources). We also thank Ove Edlefsen for support in organization and Peter S. Nielsen, Kjeld Callesen and Steffi Rost for technical work in the field.

References

[1] Gericke, D., Pacholski A., Kage, H., 2011. Measurement of ammonia emissions in multi plot field experiments. Biosys. Eng. 108 (2), 164-173, 2011.

[2] • Hutchinson, G. L., Mosier, A. R., 1981. Improved soil cover method for field measurement of nitrous oxide fluxes. Soil Sci Soc Am J 45, 311 – 316.

[4] Sommer, S. G., Jensen L. S., Clausen, S. G., Sogaard, H. T., 2006. Ammonia volatilization from surface-applied livestock slurry as affected by slurry composition and slurry infiltration depth. J Agric Sci 144, 229 – 235.

TA-O_14 Drying of separated manure digestate solids – Effects of acidification, temperature and ventilation on nitrogen loss

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1. Objectives

The objective of this study was to assess the effects of different acidification levels and drying conditions on the total and ammonium nitrogen content of dewatered digestate solids. We hypothesized that:

- Gradual acidification of the solids will progressively reduce nitrogen losses during thermal treatment,
- Higher drying temperatures will result in higher nitrogen losses, and
- Forced-air assisted drying will significantly increase the nitrogen loss from the solids.

2. Methodology

Solids, produced by decanter centrifugation of slurry-based (70% cow and 20% pig slurry feedstock) digestate, were collected fresh from Morsø Biogas plant, Denmark. The pH of the dewatered digestate was adjusted from the initial value of 9.2 to pH of either 6.5 or 5.5, by addition of 12.5 µl or 17.5 µl (g^{-1} fresh solids) concentrated sulfuric acid, respectively. The drying operating conditions included four different drying temperatures (70 °C, 100 °C, 130 °C and 160 °C) and two moisture evaporation regimes, regulated by application of external forced aeration or no forced aeration. The batch drying experiment was conducted in a laboratory conductive oven. Solids were placed in aluminium foil containers (250 ml) and sealed with their respective lids. Container lids were perforated with 25 holes (Ø=0.5mm), to promote moisture evaporation. Samples under externally ventilated conditions, received external air (420 ml/min) through a tube fitted centrally in the lid. In preliminary trials the time needed to dry the solids to a dry matter content of 85% (wet weight based) under each operational condition was estimated and this time divided into 4 drying periods of equal length. A triplicate set of samples remained in the oven for either one, two, three or all four drying periods. Samples were then removed, placed in a desiccator and subsequently their weight and pH was recorded. Before being kept frozen for later analyses, part of the solids was extracted with 1 M KCL for inorganic nitrogen determination. Total nitrogen and carbon (C) were determined on an elemental analyzer coupled to a mass spectrometer.

Table 5. Main characteristics of the digestate solids used in the present study.

Dry matter content (% of FM)	pH	NH_4^+-N (g Kg^{-1} DM)	Total-N (g Kg^{-1} DM)
26.4	9.2	15.1	32,5

3. Results and discussion

3.1 Solids drying characteristics

Drying of the solids revealed an almost linear moisture evaporation progress with time, across all acidification levels and drying temperatures (Fig.1). Drying at increased operating temperatures markedly decrease drying time. Time required for the solids to reach the desired DM content (85 % of wet weight) was decreased by almost 80%, when solids were dried at 160 °C, compared to drying at 70 °C, both under forced aeration (Fig. 1). The higher energy input induced to the system under elevated temperatures, accelerates moisture diffusion and reduces the drying time. Comparably, application of external forced aeration during drying, decreased the drying time by 79% and 20% at 70 °C and 160 °C, respectively, compared to drying without forced ventilation (data not shown). Headspace air is more quickly saturated at low temperatures resulting in an extended drying time. Introduction of fresh air in the system, increases the convective transfer of moisture into the airstream which may explain the detected interaction between temperature and ventilation rate. The drying rate of the acidified solids appeared to be slightly higher than for the non-acidified until the 3rd time interval, across most drying temperatures (100 °C to 160 °C). An improvement in sludge dewaterability after acidification has been found earlier [1], possibly due to the release of water held by extracellular polymers (EPS). Release of EPS bounded water might have facilitated the faster evaporation process during drying of acidified solids. After the end of the drying, the solids acidified to pH 5.5 showed significantly ($p<0.05$) lower moisture content in half of the tested drying conditions (data not shown).

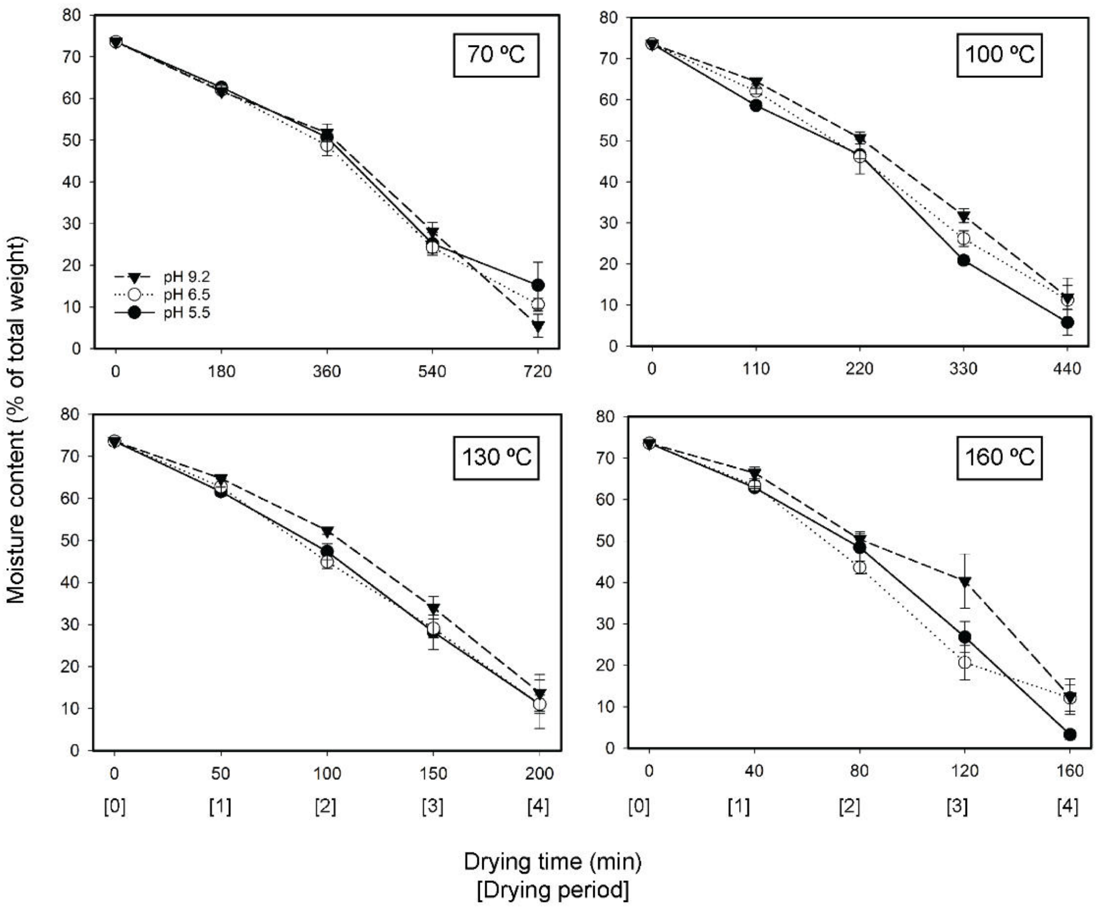

Figure 1: Moisture loss curves of solids during drying under different operating temperatures and acidification levels,in ventilated drying conditions. Drying times corresponding to the four drying periods for each temperature are denoted on the x-axis. Error bars represent standard deviation (n=3)

3.2 Chemical composition

Nitrogen: Clearly, there was a strong effect of acidification on the NH4-N content of the dried solids (Fig. 2). While the NH4-N concentration in the non-acidified solids decreased sharply at the end of the drying, solids acidified to pH 6.5 and 5.5 were able to retain 72.8% and 82.9% of the original NH4-N content (mean recoveries across all drying temperatures), respectively. NO3-N concentrations were always below the detection limits, indicating NH3 volatilization as the primary source of inorganic nitrogen losses. The alkaline nature of the non-acidified solids, promotes the dissociation of NH4+ to the easily volatilized NH3. Release of NH3 from manures strongly depends on the environmental conditions that manures are exposed to and thus drying under ventilated conditions facilitates NH3 volatilization [2]. Acidification of the solids to pH 6.5 and 5.5 proportionally inhibited the formation of NH3 in the aquatic phase of the solids and reduced the NH4-N losses. Another possible explanation for the high NH4-N recovery in the acidified samples may be the formation of ammonium sulfate ((NH4)2SO4) granules, after the reaction of H2SO4 with the NH4+ content of the solids.

The initial values of NH4-N was increased in the acidified solids compared to the non-acidified ones Fig. 2). Moreover, the organic N (Norg) concentration was lower in the case of the acidified solids. Sulfuric acid addition to the solids may have induced partial hydrolysis of the solids organic matter similar to previous reports [3] , and consequently mineralization of organic N. However, no significant different in Norg concentration was highlighted between solids acidified to pH 6.5 and 5.5, possibly due to the recalcitrant nature of the solids.

The effect of drying temperature on the NH_4-N content of the solids was significant only in the case of non-acidified solids dried under no external ventilation, where higher drying temperatures resulted in higher NH_4-N losses during drying (data not shown). Furthermore, we found no temperature effect on Norg of the solids, within each acidification level.

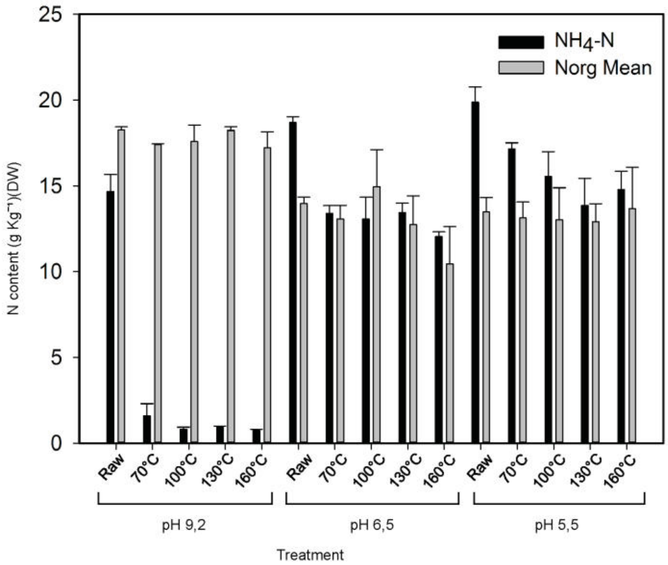

Figure 2: Changes in NH_4-N and organic N content between raw and finally dried digestate solids under different drying conditions and acidification levels, in ventilated drying conditions. Error bars represent standard deviation (n=3)

pH and Carbon content: Changes in the solids pH during drying revealed a contrasting pattern between acidification levels (Fig. 3). After the conclusion of drying, the non-acidified solids showed a gradual decrease of pH compared to acidified solids. In the latter, the pH increased slightly initially and then remained relatively stable or declined slightly during drying, across all drying conditions. The pH of the manures is regulated by the acid-base buffering systems, mainly ammonium and bicarbonate, in the solids, which in turn is affected by NH3 and CO2 emissions. The pH decrease of the non-acidified could be attributed to the acidifying effect of NH3 volatilization and might indicate higher NH3 volatilization compared to CO2 emissions. The stability of pH in the acidified samples is likely to be related with a more balanced emission of NH3 and CO2.

The C concentration of the solids decreased significantly after acidification. The raw un-acidified solids contained 37,2 (± 0.3) g C kg^{-1} DM while the acidified had 34.7 (± 0.28) g C kg^{-1} DM and 34.3 (± 0.11) g C kg^{-1} DM when acidified to pH 6.5 and 5.5, respectively. Drying temperature and ventilation did not significantly decrease the carbon content of the solids, which remained relatively stable across all drying conditions (data not shown).

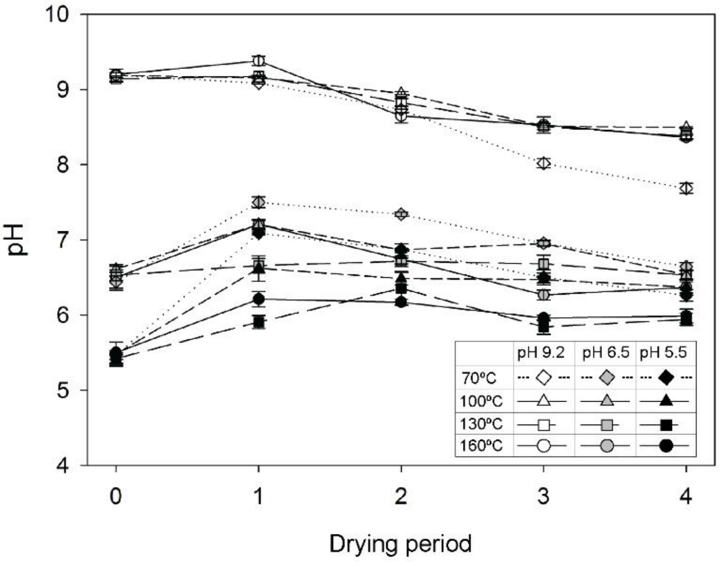

Figure 3: Solids pH evolution during drying in ventilated samples, under different operating temperatures and acidification levels. Error bars represent standard deviation (n=3). When error bars are not visible, they fall within symbols

4. Conclusion and outlook

We have shown that moderate additions of sulfuric acid before thermal drying can be an effective pre-treatment of the digestate solids in cases where management strategies focus on the retention of nitrogen in the dried product. The results of this study indicated the suitability of acidification as a pre-treatment of the solids, in a wide range of drying conditions. The thermo-chemical treatment of the solids results in a concentrated organic fertilizer with potentially much higher nitrogen availability than dried, non-acidified solids. Future research should address the effect of acidification and drying on the overall nutrient load of the solids as well as the agronomic impact of the products on plant nutrient uptake efficiency.

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References

[1] Y. Chen, H. Yang, G. Gu, Effect of acid and surfactant treatment on activated sludge dewatering and settling, Water Res. 35 (2001) 2615–2620.
[2] S.G. Sommer, N. Hutchings, Ammonia emission from field applied manure and its reduction—invited paper, Eur. J. Agron. 15 (2001) 1–15.
[3] P. Kennedy, H. Kennedy, S. Papadimitriou, The effect of acidification on the determination of organic carbon, total nitrogen and their stable isotopic composition in algae and marine sediment, Rapid Commun. Mass Spectrom. 19 (2005) 1063–1068.

TA-O_15 Increasing thermal drying temperature of biosolids reduced nitrogen mineralisation and N_2O emissions

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1. Objectives

Sewage sludge is frequently treated at waste water treatment plants (WWTPs). Due to the relatively-high availability of nutrients in sludge, particularly nitrogen (N), the treated product can be used in agriculture.

Thermal drying is an increasingly common sludge treatment process; however, the effect of drying method and conditions on N release is not well understood. We investigated the effect of thermal drying method (in a belt-drying process at a waste water treatment plant, WWTP, or in a laboratory oven) and drying temperature (70, 130, 190, and 250 °C) on the N content of anaerobically-digested (AD) sludge, then subsequent N release and nitrous oxide (N_2O) and carbon dioxide (CO_2) emissions in soil under field-moist (pF 2) conditions.

2. Methodology

2.1. Preparation of sludge materials

AD sludge was collected from Randers WWTP, Denmark, in July 2013. WWTP-dried sludge (AD-FD), was collected from the same facility, and had been dried on a belt drier (95 °C input temperature and 200 °C thermal fluid circuit temperature). For the other dried treatments, AD sludge was dried in a laboratory-oven at 70, 130, 190 or 250 °C to less than 5% GMC (AD-LD70, 130, 190, 250).

2.2. Incubation test with dried sludges

AD and dried sludges were added to 45 g (d. w.) of sandy loam soil at a 2 % d. w. rate in plastic pots (D = 4.7 cm, H = 7.0 cm). Soil water contents were adjusted to pF 2, and samples were incubated at 15 °C in the dark for 120 days (close to the average Danish summer temperature). Soil ammonium (NH_4^+) and nitrate (NO_3^-) concentrations were analysed on day 0, 3, 7, 14, 42, 80, 120, and soil N_2O and CO_2 emissions were analysed on day 0, 1, 3, 5, 7, 10, 16, 24, 48, 80, 120. The percentage of mineralisable N in sludge (*Min-N*), and the percentage of sludge total N emitted as N_2O-N (N_2O%) or percentage of sludge total C emitted as CO_2-C (CO_2%) were calculated from these results.

- Total combustible C and N: Sludge samples were dried and analysed on an elemental analyser. Total combustible N was used as the value of organic N for subsequent calculations. The total N content of sludge was determined by adding total combustible N and inorganic N content.
- NH_4^+/NH_3 and NO_3^- concentrations: Soil or sludge was mixed with 1 M KCl at a 1:4 or 1:40 ratio respectively, shaken for one hour and filtered. The extracts were analysed using a FIAstar 5000 analyser (Foss Analytical, Denmark). These concentrations were used to calculate *min-N* (below).
- *Min-N*: The equation to calculate the percentage of sludge mineralisable organic N (*min-N*) was taken from the literature [1]:

$$Min - N\ (\%) = \frac{(N_s - N_a - N_c)}{N_{org}} * 100 \qquad \text{Eq. 1}$$

Where N_s is the inorganic-N concentration in sludge-amended soil at the end of the incubation period, N_a is the amount of NH_4^+-N supplied to the soil in the sludge, N_c is the mineral-N concentration of the control soil, and N_{org} is the amount of organic-N added to the soil by the sludge. All units are mg N kg^{-1}.

- Percentage of total sludge N or C emitted as N_2O or CO_2 (N_2O% and CO_2%): First, N_2O and CO_2 gas fluxes were analysed. Soil and sludge mixes were placed in sealed glass jars for up to two hours. 10 ml gas samples were taken using syringes at the start at end of closing, and stored in 3 ml vials. Concentrations were then analysed on a gas chromatograph (Bruker 450C, Germany), from which gas fluxes were calculated. The mean hourly gas flux between two time points was multiplied by the time difference in hours, and the result was divided by the total N or C in the sludge, giving N_2O% or CO_2%.
- Statistics: One-way ANOVAs were conducted, followed by Tukey's honest significant difference test.
- pH: Sludge or soil samples were put in deionised water for 1 hour before pH analysis using a PHM210 pH meter (Radiometer analytical, UK).

3. Results and discussion

3.1 Sludge N concentrations and mineralisable organic N

Sludge NH_4^+/NH_3 concentrations in anaerobically-digested sludge (AD) were reduced by at least 29% after thermal drying, and decreased most in sludges dried at high temperatures (~87% reduction in AD-LD190 and AD-LD250, Table 1). This was expected, as the drying process typically volatilises NH_4^+ within sludge as NH_3, of which most is lost. Organic N contents in the sludge were similar between dried and un-dried sludges (Table 1).

Table 6: Dried and fresh sludge N properties. Treatments: AD = dewatered AD sludge, AD-FD = WWTP-dried sludge and AD-LD70, AD-LD130, AD-LD190, and AD-LD250 = laboratory-dried AD sludge dried at 70, 130, 190 or 250 °C respectively. SE = standard error, n = 3. Letters indicate treatments that are significantly different from one another ($p > 0.05$).

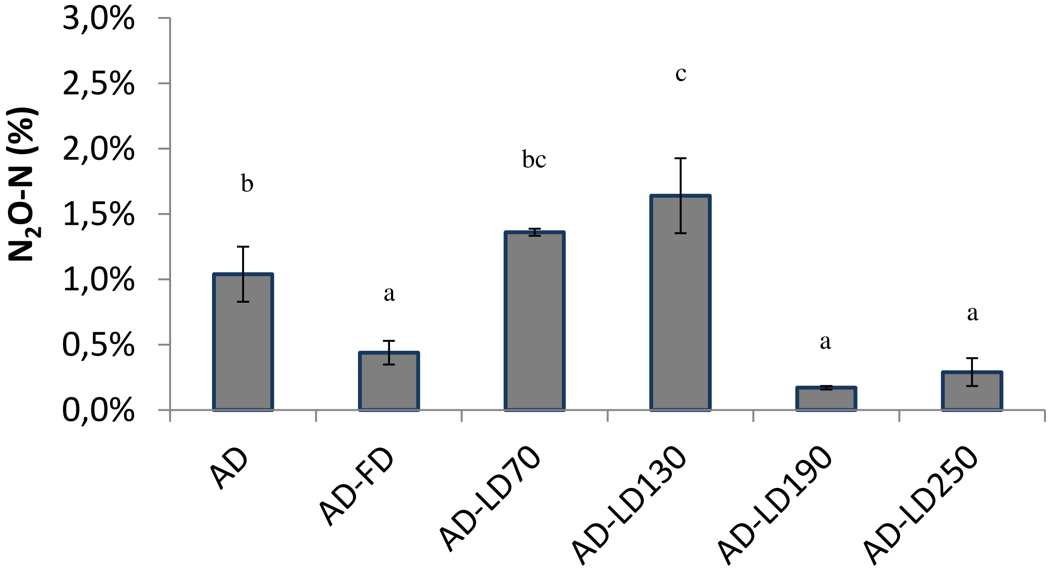
Sludge	Treatment					
parameter	**AD**	**AD-FD**	**AD-LD70**	**AD-LD130**	**AD-LD190**	**AD-LD250**
Drying method	NA	Belt dryer	Lab oven	Lab oven	Lab oven	Lab oven
Drying temperature (°C)	NA	95 - 200	70	130	190	250
NH_4^+-N / NH_3-N g kg^{-1} d.w. (SE)	8.98 (0.37) a	3.43 (0.04) b	6.38 (0.11) c	4.64 (0.03) d	1.09 (0.03) e	1.15 (0.07) e
Organic N g kg^{-1} d.w. (SE)	45.0 (0.5) a	46.0 (0.3) a	45.1 (0.2) a	45.8 (0.6) a	44.8 (0.5) a	45.8 (0.5) a
Total C g kg^{-1} d.w. (SE)	284.0 (2.4) a	293.3 (1.4) b	300.4 (4.0) b	300.5 (4.3) b	297.8 (2.4) b	290.2 (1.5) a
pH (SE)	8.43 (0.02) e	7.32 (0.02) d	6.91 (0.02) c	6.65 (0.05) b	5.79 (0.03) a	6.47 (0.07) b
Min-N % (SE)	46.4 (2.9) c	50.5 (1.4) c	35.4 (0.4) bc	30.0 (7.3) ab	28.1 (0.9) ab	19.3 (1.2) a

Thermal drying did not increase *min-N* in the sludge compared to undried sludge (the AD treatment) regardless of the drying technique used (in a WWTP facility or in a laboratory oven, $p > 0.05$, Table 1). *Min-N* decreased with increasing laboratory drying temperature, e.g. from 35.4 to 19.9% between 70 and 250 °C. The data indicated that drying sludge does not increase *min-N*, while reducing the N content of the sludge. From this we concluded that thermal drying of the AD sludge did not improve its N fertiliser properties.

It is possible that the drying process altered the physical and biochemical properties of the N retained in sludge (mostly amides and proteins), rendering them less available to soil. For example, Maillard reactions occur at temperatures of 170°C or above and may cause the creation of recalcitrant chemical bonds and agglomerations of particles [3].

3.2 N_2O and CO_2 production

N_2O% ranged between 0.17 and 1.64%, and sludge lab-dried at 130 °C (AD-LD130) had the greatest N_2O% (Figure 1, group c, $p < 0.05$). In all cases, N_2O% was close to or lower than the 1% emission factor defined by the IPCC for fertilisers [2] (apart from sludges dried at 70 and 130 °C), and indicated that adding AD or dried sludge to soil did not vary greatly from the standard emission factor values given for other organic fertilisers. It was not clear why the sludge dried at 130 °C had a greater N_2O% than the un-dried AD sludge.



CO2% also differed significantly between treatments; between 8.9 and 28.6% of sludge C was emitted over 120 days (Figure 2, $p < 0.05$). AD-LD190 and AD-LD250 had the lowest CO2% (Figure 2, $p < 0.05$), which indicated that sludges dried at the highest temperatures were the most recalcitrant. The previously-mentioned Maillard reactions affect both nitrogenous and carbonaceous compounds in organic materials at 170 °C, and so may at least partially explain the low CO2% and N2O% from sludges dried at high temperatures.

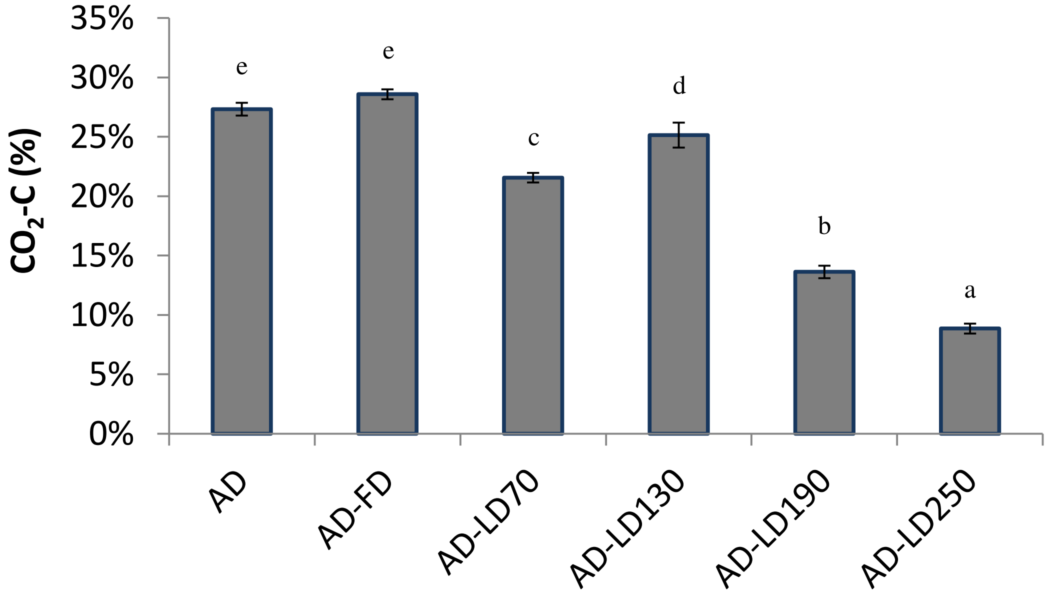


Figure 3: The effect of sludges on the percentage of sludge total C emitted as CO_2-C, (CO_2%) over 120 days. Filled bars indicate mean, error bars indicate significant error (n = 3). The treatment letters signify the following: AD = soil dewatered AD sludge, AD-FD = soil + WWTP-dried sludge, and AD-LD70, AD-LD130, AD-LD190, and AD-LD250 = soil + laboratory-dried AD sludge dried at 70, 130, 190 or 250 °C respectively. Letters above bars indicate treatments that are significantly different from one another ($p > 0.05$)

4. Conclusion and outlook

Thermal drying reduced total and inorganic N contents of AD sludge, and did not increase sludge *min-N*. From this evidence, we concluded that drying this AD sludge was not a suitable treatment to increase N availability. Sludges dried at the highest temperatures had the lowest N content, *min-N*, and CO_2%, which indicated they were more recalcitrant than the others. The effects of processes such as Maillard reactions at high temperatures on sludge physical and biochemical characteristics may explain these findings. More research is needed to confirm the influence of drying, and drying temperature on mineralisation of N from anaerobically digested biosolids.

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References

[1] Smith, S. R. & Durham, E. (2002): Nitrogen release and fertiliser value of thermally-dried biosolids. J. Chart. Inst. Water Environ. Manag. 16, 121–126

[2] De Klein, C., Novoa, R.S.A., Ongle, S., Smith, K.A., Rochette, P., Wirth, T.C., McConkey, B.G., Mosier, A., Rypdal, K., (2007): N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. Volume 4: Agriculture, Forestry and Other Land Use, 2006 IPCC Guidelines for National Greenhouse Gas Inventories

[3] Carrère, H., Dumas, C., Battimelli, A., Batstone, D.J., Delgenès, J.P., Steyer, J.P., Ferrer, I., 2010. Pretreatment methods to improve sludge anaerobic degradability: a review. Journal of Hazardous Materials 183, 1–15. doi:10.1016/j.jhazmat.2010.06.129

TA-O_16 Composting or ensiling cattle slurry solid fraction to improve product quality and limit nutrient losses

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1. Objectives

Separating cattle slurry into a solid and liquid fraction is gaining more interest by dairy farmers due to the strict phosphate fertilizer restrictions. After separation, the solid fraction (SF) contains more phosphorus, what limits the application in nutrient-rich soils such as in Flanders. Furthermore, to export the SF, sanitation is important. We evaluated if composting or ensiling the SF, whether or not mixed with byproducts, could (i) optimize the quality of the processed SF as fertilizer and soil improver, and (ii) reduce nutrient losses during storage.

2. Methodology

Storage experiments were conducted in an open-air composting facility with a concrete floor between April-June 2014. Five SF treatments with a volume of 24 m³ were compared:

- Composting SF (*SF_C*)
- Composting SF (16 m³) with straw (4 m³) and grass (4 m³) (*SF+S+G_C)*
- Ensiling SF (16 m³) with straw (4 m³) and grass (4 m³) (*SF+S+G_E)*
- Composting SF (12 m³) with cattle farmyard manure (CFM) (12 m³) (*SF+CFM_C)*
- Composting SF with clinoptilolite, a zeolite with a cation-binding capacity (2% on dry weight) (*SF+Clin_C)*

The compost piles were covered with a TopTex cover which allows gas exchanges, while the silage was covered with a plastic to ensure anaerobic conditions.
Temperature, CO_2 levels and moisture content were monitored 3-4 times per week on four different points per pile. The piles were aerated when temperatures reached 65°C or CO_2 levels exceeded 16%, using a compost turner (TG 301, Gujer Innotec AG, Switzerland). To analyze the product quality, samples (n = 4) were taken at the start and at the end of the experiment (after 61 days). Samples were analyzed for physico-chemical compost quality parameters (Table 1). During the experiment gaseous emissions (NH_3, N_2O, CO_2 and CH_4) were quantified once a week with a photoacoustic gas monitor (INNOVA 1312A, LumaSense Technologies, Denmark), after removing the pile covers. A N mineralization experiment was run for soil amended with the end products (products were added based on an equal N input of 170 kg N ha^{-1}). Soil mineral N was measured according to ISO 14256-2. A pot experiment was set up based on a pot trial with English rye grass (products were added based on an equal N input of 100 kg N ha^{-1} and P_2O_5 was added to adjust for an equal P input). Control treatments were included; each treatment consisted of thee replicates.

Table 1: Overview of the analyzed parameters of the feedstock (solid fraction of cattle slurry, grass and straw) and the end products after 61 days of composting or ensiling.

Parameter	Unit	Guideline
Dry matter content	% FM^{-1}	EN 13040
Organic matter content	% DM^{-1}	EN 13039
NO_3^--N and NH_4^+-N	kg Mg^{-1} DM	BAM 4/05 (manure)
pH-H_2O	-	EN 13037
Electrical conductivity	µS cm^{-1}	EN 13038
Total N	kg Mg^{-1} DM	Dumas EN 13654-2
Total P, K, Mg, Ca, Na	kg Mg^{-1} DM	CMA 2/IV/19 - ICP-OES
(Hemi-)cellulose and lignin content	% DM^{-1}	Van Soest et al. (1991)
Germinal seeds	-	CMA /2/IV/10

FM-fresh matter; DM-dry matter

3. Results and discussion

3.1 Temperature and CO_2 evolution in the piles

The temperature of the compost piles increased directly after the start of the experiment. Adding straw and grass or CFM resulted in higher temperatures during the first 20 days compared to composting of pure SF (Figure 1). We assume that adding more structure lead to more O_2 and microbial activity in the pile. Moreover, the feedstock mixture at the start was dryer when straw and grass were added (see 3.3), which facilitated O_2 transport and lead to the maintenance of higher temperatures (Brito et al., 2012). On the other hand, addition of straw and grass or CFM, lead to lower CO_2 levels. Only by adding straw and grass or CFM, temperatures rose above 55°C for 15 days, indicating the destruction of pathogens (Lung et al., 2001) and weed seeds (Eghball & Lessing, 2000). Ensiling resulted in lower temperatures (maximum 45°C), but higher CO_2 levels when removing the plastic cover from the silage, compared to composting.

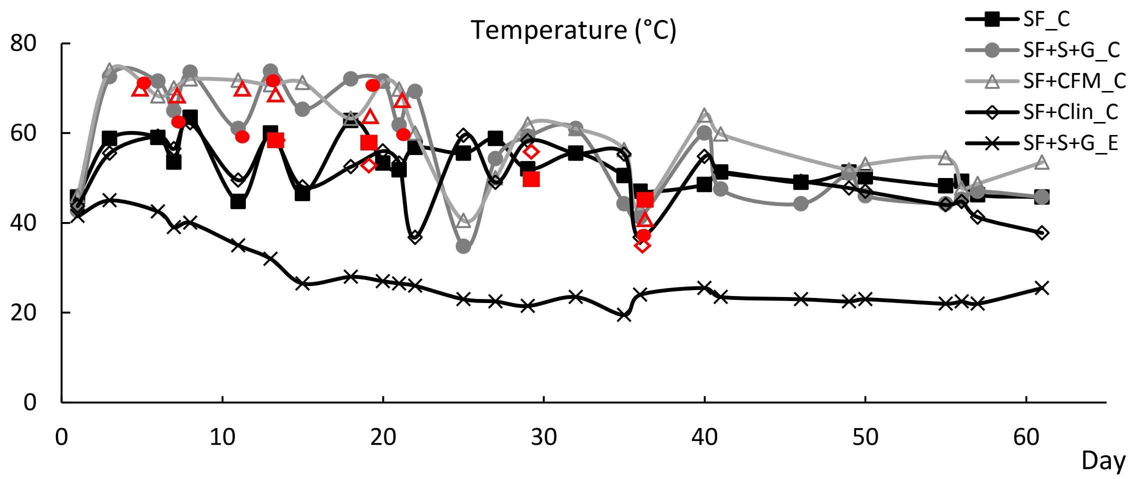

Figure 1: Temperature profile during the composting/ensiling process. SF_C = composting solid fraction of cattle slurry (SF); SF+S+G_C = composting SF with straw and grass; SF+CFM_C = composting SF with cattle farmyard manure; SF+Clin_C = composting SF with clinoptilolite; SF+S+G_E = ensiling SF with straw and grass. Red symbols indicate when the piles were turned.

3.2 Gaseous emissions

Adding straw and grass or clinoptilolite reduced the NH_3 emissions. The added carbon source (straw and grass) could be used by the thermophilic microorganisms, inducing the microbial immobilization of NH_4^+-N (Li et al., 2013), while the zeolite clinoptilolite has a cation-binding capacity (Inglezakis et al., 2004), through which NH_3 emissions decreased. When removing the plastic cover from the silage, higher CO_2 and CH_4 emissions were measured compared to the composted SF with straw and grass. Adding CFM to the SF resulted in more NH_3 emissions during the first month, what can be explained by the higher pile temperatures.

3.3 Product quality

Adding straw and grass to the SF resulted in a lower NH_4^+-N concentration (by adding structure and O_2) and higher total N content (due to the N-rich grass); thus lowering the C/N ratio of the feedstock. Moreover, the DM content, NO_3^--N concentration and C/P ratio of the feedstock increased (Table 2). Adding CFM to the SF resulted in a higher OM and DM content, a lower total N content, and a higher C/N and C/P ratio (Table 2).
Generally, **composting** decreased the OM content due to decomposition of the feedstock, while the DM content increased due to the high temperatures and the evaporation of water, resulting in a decrease of the C/N and C/P ratios (Table 2 and 3). The decomposition of OM resulted in a stabilization of the composts, confirmed by a decreased potential biodegradability. Composting SF with straw and grass or CFM resulted in a higher DM content and lower NO_3^--N concentration and NO_3^--N/NH_4^+-N ratio of the end products compared to composting SF without amendments (Table 3). Moreover, composting SF with CFM resulted in a product with a lower P content and higher C/P ratio compared to composting SF. In contrast to the composting process, **ensiling** SF with straw and grass resulted in only limited OM decomposition (Table 2 and 3). There was a small decrease in the potential biodegradability, indicating a limited degradation. Ensiling resulted in a higher OM content and a lower DM content compared to the compost of SF with straw and grass.

Moreover, the silage had a lower NO_3^--N and higher NH_4^+-N concentration, and thus a lower NO_3^--N/NH_4^+-N ratio; a lower P content and a higher C/N and C/P ratio. With ensiling, relatively more (hemi-)cellulose and less lignin were observed (Table 3). The limited degradation process indicates that OM decay was delayed due to ensilage and will continue after the ensilage product is applied to the soil.
A positive net N mineralization for all products was measured (data not shown). The net mineralization was higher for the silage (42 mg kg^{-1} DM) compared to the compost with SF and straw and grass (17 mg kg^{-1} DM) after 100 days. Adding clinoptilolite resulted in a slower N mineralization and less N was mineralized (46 mg kg^{-1} DM) compared to the compost with only SF (51 mg kg^{-1} DM) after 100 days.
The higher net N mineralization of the ensilaged SF in the incubation experiment was also observed in the pot experiment where a higher DM crop yield was observed for the silage treatment (2938 ± 153 compared to 1937 ± 58 kg ha^{-1} for the compost treatment). There were no differences in crop yields between the other treatments after 60 days (data not shown).

Table 2: Chemical characterization of the feedstock at the start of the experiment (n=4)

	SF_C and SF+Clino_C	SF+CFM_C	SF+S+G_C and SF+S+G_E
Organic matter content (% DM^{-1})	78.9 ± 2.1^a	80.3 ± 0.5^b	84.7 ± 1.5^a
Dry matter content (% FM^{-1})	24 ± 0.3^a	27.3 ± 0.1^b	25.6 ± 0.3^c
NO_3^--N (kg Mg^{-1} DM)	0.014 ± 0.006^a	0.006 ± 0.001^a	0.020 ± 0.010^b
NH_4^+-N (kg Mg^{-1} DM)	2.17 ± 0.25^b	0.57 ± 0.04^b	1.74 ± 0.40^a
N (kg Mg^{-1} DM)	21.8 ± 0.8^b	19.6 ± 0.2^a	25.8 ± 1.2^c
P (kg Mg^{-1} DM)	7.1 ± 0.5^b	5.3 ± 0.2^a	5.3 ± 0.3^a
C/N (-)	20.1 ± 0.8^b	22.8 ± 0.1^a	18.3 ± 0.6^c
C/P (-)	62 ± 6^a	84 ± 3^b	88 ± 6^b
Hemicellulose	18.9 ± 2.3^a	18.7 ± 1.1^a	20.5 ± 3.3^b
Cellulose	30.2 ± 1.5^b	31.7 ± 1.0^b	26.7 ± 2.5^a
Lignin	13.2 ± 0.2^c	11.7 ± 0.6^b	8.1 ± 1.1^a
Potential biodegradability	3.7 ± 0.2^a	4.3 ± 0.3^a	5.8 ± 0.9^b

FM-fresh matter; DM-dry matter

Table 3: Chemical characterization of the end products after 61 days of composting/ensiling (n=4)

	SF_C	SF+Clino_C	SF+CFM_C	SF+S+G_C	SF+S+G_E
Organic matter content (% DM^{-1})	62.8 ± 1^{ab}	60.8 ± 1^a	64.1 ± 1.2^b	62.3 ± 1^{ab}	83.7 ± 1.9^c
Dry matter content (% FM^{-1})	26.1 ± 0.3^b	25.5 ± 1^b	47.1 ± 1.2^d	35.4 ± 3.8^c	20.4 ± 1.2^a
NO_3^--N (kg Mg^{-1} DM)	1.023 ± 0.097^d	1.259 ± 0.307^d	0.056 ± 0.015^b	0.254 ± 0.066^c	0.005 ± 0.002^a
NH_4^+-N (kg Mg^{-1} DM)	1.23 ± 0.12^a	0.92 ± 0.37^a	0.56 ± 0.04^a	1.14 ± 0.27^a	5.59 ± 0.79^b
NO_3^--N / NH_4^+-N (-)	0.835 ± 0.112^c	1.660 ± 1.002^c	0.100 ± 0. 031^{ab}	0.238 ± 0.099^b	0.001 ± 0.000^a
N (kg Mg^{-1} DM)	30.8 ± 1.4^a	28.7 ± 2.4^a	32.5 ± 0.8^a	33.2 ± 1.8^a	30.0 ± 3.6^a
P (kg Mg^{-1} DM)	11.6 ± 0.3^c	10.7 ± 0.5^b	9.8 ± 0.2^b	11.7 ± 0.2^c	5.1 ± 0.5^a
C:N (-)	11.3 ± 0.3^a	11.8 ± 1^a	11 ± 0.8^a	10.5 ± 0.8^a	15.7 ± 1.6^b
C:P (-)	30 ± 0.9^c	32 ± 2^c	36 ± 3^b	30 ± 1^c	92 ± 11^a
Hemicellulose	7.3 ± 4.3^a	9.3 ± 2.0^a	7.6 ± 1.8^a	9.2 ± 3.0^a	21.0 ± 0.8^b
Cellulose	13.2 ± 3.1^a	12.5 ± 2.4^a	8.6 ± 1.8^a	8.1 ± 2.4^a	28.3 ± 2.0^b
Lignin	27.4 ± 1.3^b	26.7 ± 1.9^b	24.0 ± 1.7^b	24.9 ± 0.8^b	13.5 ± 2.0^a
Potential biodegradability	0.7 ± 0.2^a	0.8 ± 0.1^a	0.7 ± 0.1^a	0.7 ± 0.2^a	3.8 ± 0.5^a

FM-fresh matter; DM-dry matter

4. Conclusion and outlook

Table 4 summarizes the effects of the different treatments of SF on the composting or ensiling process and the quality of the end products. Composting SF in open air was not convenient due to the lack of O_2 and the risk of serious gaseous losses. Mixing SF with structure-rich feedstock or clinoptilolite could reduce this risk to a certain extent. In contrast to composting, only a limited degradation was noticed when ensiling SF. Ensiling can thus be seen as a controlled storage method. Soil amendment with ensilaged SF resulted in a higher N mineralization and crop growth compared to the application of composted SF. Future research could focus on a more detailed monitoring of the gaseous emissions during storage and on nutrient losses (through emission or leaching) after application of the products to the soil. Furthermore, sanitization of the end products should be validated, e.g. with continuous temperature measurements over the whole length of the piles.

Table 4: Effect of different SF treatments on process and product quality. SF_C = composting solid fraction of cattle slurry (SF); SF+S+G_C = composting SF with straw and grass; SF+Clin_C = composting SF with clinoptilolite; SF+S+G_E = ensiling SF with straw and grass. Score: LLL (very low) - LL (low) - L (rather low) - 0 (no effect) - H (rather high) - HH (high) - HHH (very high). The risk for ammonia losses for SF+S+G_E was measured after removing the plastic cover at the end of the experiment.

		SF_C	SF+S+G_C	SF+CFM_C	SF+Clino_C	SF+S+G_E
Process						
Physical parameters	Temperature - sanitization	H	HH	HH	H	L
	Aerobic circumstances	L	H	H	L	LL
	Moisture content	L	LL	LLL	L	H
OM degradation		H	H	H	H	L
Product Quality						
Stability	Potential biodegradability	LLL	LLL	LLL	LLL	L
	NO_3^--N / NH_4^+-N	L	LL	LL	L	LLL
Nutrient content	C/N	H	H	H	H	HH
	C/P	H	H	H	H	HH
N mineralization		L	LL	/	LLL	H
Crop growth		H	HH	/	H	HHH

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References

[1] Brito, L.M.; Mourão, I.; Coutinho, J., Smith, S.R. (2012): Simple technologies for on-farm composting of cattle slurry solid fraction, Waste Management, 32, 1332–1340

[2] Eghball, B.; Lessing, G.W. (2000): Viability of weed seeds following manure windrow composting, Compost Science and Utilization, 8, 46-53

[3] Inglezakis, V.; Loizidou, M.; Grigoropoulou H. (2004): Ion exchange studies on natural and modified zeolites and the concept of exchange site accessibility, Journal of Colloid and Interface Science, 275, 570-576.

[4] Li, Y.; Li, W.; Wu, C.; Wang, K. (2013): New insights into the interactions between carbon dioxide and ammonia emissions during sewage sludge composting, Bioresource Technology, 136, 385–393

[5] Lung, A. J.; Lin, C.M.; Kim, J.M.; Marshall, M.R.; Nordstedt; R., Thompson, N.P.; Wei, C. I. (2001): Destruction of Escherichia coli O157:H7 and Salmonella enteritidis in cow manure composting, Journal of Food Protection, 64, 1309-1314

TA-O_17 Scaling up pig slurry composting from laboratory to farm

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1. Objectives

The liquid character of the pig slurry, together with its high concentration of N as NH_4-N, may limit its composting. So, solid-liquid separation is a necessary pre-treatment before subsequent composting of the solid fraction. The objective of the present work is to develop a composting strategy for the treatment of the solid fraction of pig slurry and its establishment at the farm level. Three partial objectives were identified:

- To select the appropriate bulking agent for composting the solid phase of pig slurry by a shelf-heating test and microbial degradability test;
- To establish the methodology concerning the proportion of bulking agent;
- To adapt the developed procedure to the farm level, using the basic farm equipment.

2. Methodology

Three composting experiments were run at different scales: laboratory reactors for selecting the bulking agent; pilot-plant for defining proportions; and farm scale for establishing the technology in a pig farm. Solid fractions of pig slurry (SPS) were separated by screw press without flocculants:

- Laboratory reactors: Four mixtures were prepared using the SPS1 with 4 bulking agents: cotton gin waste (CW), maize stalks (MS), barley straw (BS), and garden pruning wastes (PW), at the ratios (% weight): SPS:CW = 83:17; SPS:MS = 92:8; SPS:BS = 94:6; SPS:PW = 87:13, in order to keep the moisture content 65-70 % without any leaching. Mixtures were placed in 5 L batch reactors (in duplicate), consisting of thermal insulated cylinders with 14 cm inner-diameter and 40 cm height. Temperature probes connected to a data logger (HOBO-Data Logger) were placed in the centre of the reactors, and internal and external temperatures were recorded automatically. The biodegradability of the mixtures at the beginning and at the end of the experiment was determined by respiration tests (CO_2-C production during 10 days).
- Pilot plant: Two mixtures were prepared by mixing SPS2 (from fatteners) with CW at two proportions (% weight): P1= 70:30, and P2 = 56:44. Trapezoidal piles of 2000 kg each were composted by the Rutgers static pile system with forced on-demand aeration (set at 65 ºC). The piles were turned two times for homogenisation, the moisture was controlled weekly (> 40 %) and the temperature was monitored using two probes for each pile. The bio-oxidative phase lasted for 107 days, then, composts were left to mature for two months (total time 167 days).
- Farm level: A trapezoidal pile (19.3 m^3) was prepared with SPS3 (from sows and piglets) and CW at 75:25 % (w), using a tractor for the mechanical turning (5 turnings); the moisture was adjusted (> 40 %) at the time of sampling. The temperature evolution was monitored during the bio-oxidative phase (HOBO-Data Logger) which lasted for 142 days, and 45 days of maturation (total time 187 days).

Samples were taken from the laboratory reactors at the end of the experiment after homogenisation of all the material in the reactor; samples from pilot plant and farm level experiments were taken periodically by mixing seven subsamples from seven representative sites of the pile, from the whole profile (from the top to the bottom of the pile). In laboratory reactors, the energy produced by the microorganisms was calculated considering the specific heat capacity of the samples (C_p) [1] and the differences in temperature between the internal and the external values: $q(J) = m \times C_p \times \Sigma\Delta T$; being: m the mass (dry weight) of the mixture and $\Sigma\Delta T$ the accumulated increase of temperature (the difference between the internal and the ambient temperatures).

3. Results and discussion

3.1 Laboratory reactors

According to the thermal profile of the mixtures (Fig. 1), the mixture with CW results in the highest temperatures and a rapid increase in temperature on the first day, increasing 12 ºC with respect to the external values, in comparison with the 8, 7 and 3 ºC increases for the mixtures with PW, BS and MS, respectively. This indicates that the microbial activity was highest in the mixture with CW. All the mixtures were able to maintain the rise in temperature for at least 3-4 days, before starting to decrease. At day 7, only the mixtures with GP and MS were turned to improve the homogeneity and the aeration of the mixtures, as the temperature had dropped to values close to the external values,. After this, the temperature increased very little in both mixtures, showing a small second peak of temperature, which indicated that the materials were still biologically active. After 10 days, the energy produced by the mixtures increased in the order (kJ): MS (96) < PW (166) < BS (296) < CW (398).

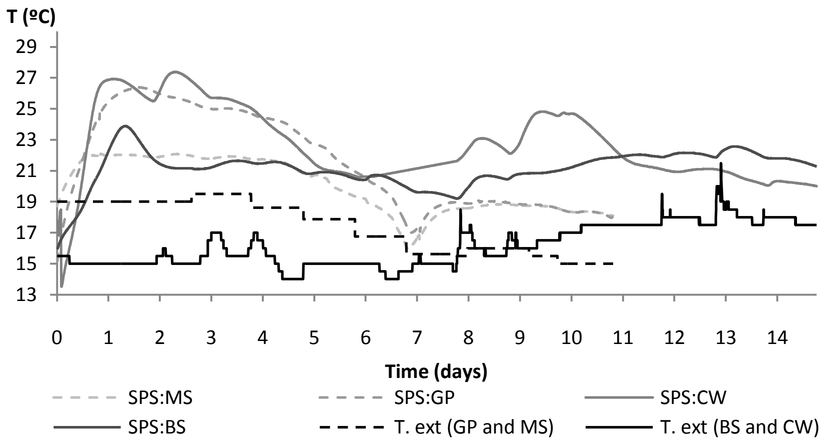

Figure 1: Temperature profile of the mixtures of solid fraction of pig slurry with different bulking agents in laboratory batch reactors (T. ext (GP and MS): external temperature in experiment with garden pruning and maize stalks; T. ext (BS and CW): External temperatures in the experiment with barley straw and cotton gin waste).

The biodegradability (CO_2-C production) of the mixtures CW and PW was lower at the end of the experiment than at the beginning; the lowest value for PW, indicates the highest stabilisation. Since CO_2-C production is a sign of microbial activity and organic matter (OM) degradation, its slowdown indicates that easily accessible compounds were already degraded. However, in the mixtures with BS and MS, the CO_2 emission did not change statistically during the experiment, and the latter mixture reached low temperature values (Fig. 2), indicating low degradability.

Table 1: Respiration test (CO_2-C emission in 10 days) of the mixtures of solid fraction of pig slurry with different bulking agents for samples taken at the beginning and end of the experiment (mg g^{-1} dry matter; n=3). Results for different bulking followed by the same letter for each sampling time are not statistically significant according to the Tukey test; n.s.: not significant P>0.05; *, **: significant at P<0.05 and 0.01, respectively.

Sampling time	GW	MS	BS	PW	ANOVA
Initial	26.7±7.0a	12.1 b	17.8 ab	24.2 a	*
End	15.6±3.6b	12.6 ab	15.6 a	10.8 b	*
ANOVA	*	n.s.	n.s.	**	

The energy produced by the microorganisms after 1 day of the experiment correlated with the CO_2-C release of the initial samples (r=0.966; P<0.01), while the energy produced after 10 days correlated to the CO_2-C released from the samples at the end of the experiment. Therefore, both parameters; the energy released as heat and the CO_2-C production indicate the degradability of the material, as the respiration index and temperature are directly related to microbial activity. The results show that cotton gin waste and garden pruning enhanced the composting of the solid fraction of pig slurry, speeding-up the process.

3.2 Pilot plant

The thermal profiles of both composting piles showed fast increases in temperature during the first week of experiment, both piles achieving values close to 65ºC (Fig. 2a). Such quick increases indicate fast biodegradability of the materials due to the presence of easily degradable forms of OM in the mixtures. Both piles showed similar patterns, reaching temperatures above 55ºC for more than 35 days, ensuring hygienic conditions in the composts (*Salmonella* was absent in 25 g and *E. coli* numbers were < 10 UFC g^{-1} in both mature composts). But the pile with greater proportion of bulking agent (P2) showed higher temperatures and a longer thermophilic phase with greater aeration demand than pile P1, indicating a stronger exothermic process in P2. So, the energy generated by microorganisms and converted into temperature of the composting mass depends on the thermal properties of the cotton gin waste used in a greater proportion in P2.

The OM degradation by the microorganisms was shown as a decrease in OM concentration of the composting mass from the beginning to the end of the experiment: in P1 from 79.1 to 57.3 %; in P2 from 73.3 to 57.2 % (dry weight), respectively. Such mineralisation also reduced the TOC concentration in the piles, the mass balance revealed that 50 % of TOC was lost during composting (52 % in P1 and 47% in P2). The OM mineralisation followed a first-order kinetic process: $OM_{min} = A \times (1-e^{-kt})$. The parameters of the equation indicates that an increase in bulking agent proportion did not statistically change the concentration of degradable OM (A parameter: 56.4±2.5 % in P1; 57.4±8.0 % in P2), but decreased the degradation rate (k decreased from P1 0.0803±0.0183 d^{-1} to P2 0.0235±0.0084 d^{-1}). Also, according to the product of A×*k* the initial OM mineralisation rate was faster in the pile prepared with a higher proportion of SFP (4.53 % OM d^{-1}), than for P2 (1.35 % OM d^{-1}). This indicates high microbial activity at the beginning of the process, as a consequence of the high amount of easily degradable material from SFP. However, the pile with high bulking showed a slow OM mineralisation rate due to the recalcitrant compounds present in the cotton gin waste, such as lignin [2]. Therefore, the high temperature values maintained for a long time in P2 may be associated with the thermal characteristics of cotton gin waste, such as the calorific value (14.7 MJ kg^{-1}) and the specific heat capacity of cotton (1.34 KJ kg^{-1} K). These are lower than frequently used bulking agents and beef manure or turkey litter [3], which conditioned the heat loss through the aeration. It is known that temperature variations during composting are the result of the thermal balance between heat generated by the microorganisms and heat loss.

The agronomical quality of the compost can be defined by the nutrient content, degree of maturity, organic matter humification and absence of phytotoxicity, pathogenic microorganisms and heavy metals (Table 2). The composts have good agricultural properties in terms of nutrient concentration, OM humification, sanitisation and maturation degree (Table 3). The Spanish legislation classified compost P1 and P2 as “class C”, due to the concentrations of Cu (326 and 262 mg kg^{-1}, respectively) and Zn (947 and 720 mg kg^{-1}, respectively), limiting their use to 5 t ha^{-1} per year.

Table 2: Characteristics of the mature compost: P1 and P2 from the pilot plant and F from the pile prepared at the farm.

Compost	pH	OM (%)	TOC (g kg^{-1})	TN (g kg^{-1})	NH_4-N (g kg^{-1})	NO_3-N (g kg^{-1})	C_{EX} (g kg^{-1})	C_{HA} (g kg^{-1})	P (g kg^{-1})	K (g kg^{-1})	GI (%)
P1	6.7	57.3	303	35.4	0.68	8.9	33.8	17.5	15	22.2	81.5
P2	6.8	57.2	292	31.1	0.47	5.7	24.8	9.7	11	22.3	85.0
F	7.2	59.7	288	28.1	1.9	4.5	29.2	18.1	28	20.8	80.2

3.3 Farm composting

The temperature evolution in the composting pile at the farm showed a fast development of the process, reaching thermophilic values (> 40°C) in the first week, with temperatures ≥ 66 ºC, this period lasted for 70 days (Fig. 2b). After 90 days the temperatures progressively decreased to external values, indicating the end of the bio-oxidative phase, which lasted for 120 days. The temperature profile led to sanitisation of the mature compost: *Salmonella* was absent (in 25 g) and *E. coli* numbers were < 10 UFC g^{-1}.

The mass balance indicated a strong degradation of the OM, the first-order kinetic equation gave a potentially mineralisable OM of 54.7 %, with a rate constant of 0.0179 $days^{-1}$; giving an initial mineralisation rate of 0.98 % OM day^{-1}.Then, a slower OM degradation occurred during the mechanical turning used in on-farm composting that in the Rutgers static pile system. The automatic on-demand aeration temperature used in the pilot-plant was more efficient than the mechanical turning for temperature control and OM degradation. Of the OM, 45 % remained in the final mature compost as stabilised OM. The characteristics of the OM of the mature compost indicated a good degree of maturity and humification (Table 2), reaching higher values of cation exchange capacity (95.7 $cmol_c\ kg^{-1}$) than the limits suggested for compost maturity and the germination index (GI) clearly indicated the absence of phytotoxicity. The compost has a high content of OM and TN (Table 2), giving a C/N ratio of 10.3, with high inorganic-N, mainly as nitrates, directly available to plants. Also, the total concentration of nutrients P and K was high, providing valuable properties as an organic fertiliser. The heavy metal concentrations, Zn and Cu (5400 and 400 mg kg^{-1}, respectively), were higher than the values found in the compost from the pilot plant experiment, which can limit its quality and use. This was due to the use of slurry from piglets, as Zn is usually included in piglets' diets in order to avoid some digestive diseases and improve growth.

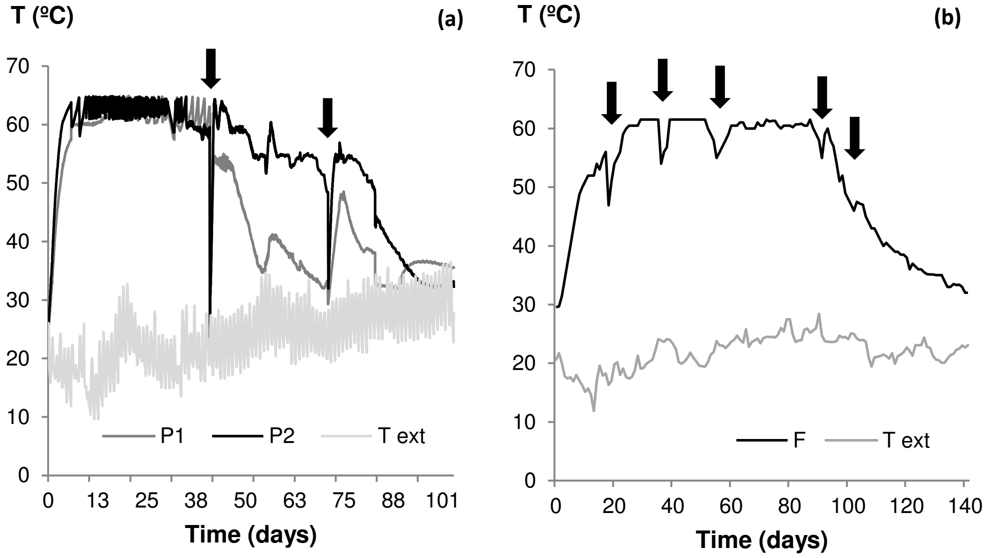

Figure 2: Temperature profile of the solid fraction of pig slurry with cotton gin waste at two proportions (P1 and P2) during pilot-scale composting (a) and at the farm level (b). T. ext: external temperature.

4. Conclusion and outlook

The laboratory and pilot plant experiments are useful for defining the methodology for composting of the solid fraction of pig slurry and the procedure can be easily established on pig farms, using basic equipment. The cotton gin waste has been demonstrated to be an adequate bulking agent for co-composting the solid fraction of pig slurry. Its thermal properties conditioned the temperature profile and ventilation demand; however, the proportion of SFP controlled the OM degradation. A proportion of 70:30 or 75:25 by weight of SFP with cotton gin waste is adequate for the correct development of the process at the real farm scale, but frequent mechanical turning is necessary for efficient aeration and fast OM degradation. A greater proportion of bulking agent requires increased ventilation of the pile to control temperature which may imply extra-cost. Pig slurry from piglets should be avoided for composting as Zn can limit compost quality.

Acknowledgements

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References

[1] Santos, A., Bustamante, M.A., Moral, R., Bernal, M.P. (2015): Carbon conservation strategy for the management of pig slurry by composting: Initial study of the bulking agent influence. Mitig. Adapt. Strateg. Glob. Chang. (in press) DOI: 10.1007/s11027-014-9593-0.

[2] Nolan, T., Troy, S.M., Healy, M.G., Kwapinski, W., Leahy, J.J., Lawlor, P.G. (2011): Characterization of compost produced from separated pig manure and a variety of bulking agents at low initial C/N ratios. Biores. Technol. 102, 7131-7138.

[3] Ahn, H.K., Sauer, T.J., Richard, T.L., Glanville, T.D. (2009): Determination of thermal properties of composting bulking materials. Biores. Technol. 100, 3974-3981.

TA-O_18 The influence of feeding on excreta characteristics of dairy cows

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1. Objectives

Nitrogen (N) is essential for any form of life and thus indispensable to sustain food production. Along the transformation process from organically bound plant nitrogen into animal protein, mobile and at the same time reactive N compounds are generated and excreted as fecal and urinary N which are recycled. However, N cycling is associated with N losses, the primary pathways being volatilization of ammonia, nitrous oxide and elementary nitrogen into the atmosphere and leaching of nitrate into aqueous systems. N losses reduce the fertilizer value of manure and the N use efficiency of animal production systems.

The most sustainable strategy to mitigate NH_3 emissions should focus on reducing its direct pre-cur-sor, i.e. the urea fraction in animal manure. Urea, a metabolite of the protein metabolism, con-sti-tutes, with 50 to 85% of total urinary N, the major nitrogenous component of urinary N in ruminants (Dijkstra et al. 2013). The large body of evidence for positive correlations between N-input, ruminal protein balance, milk urea, urinary N or urea N excretion and ammonia emissions shows that diet characteristics are clearly related with the excretory patterns of dairy cows (Külling, 2000; Reijs, 2007; Van Duinkerken et al. 2011). The objective of the present study was to identify feeding measures and indicators for low N and particularly low urinary N excretion of dairy cows under Swiss conditions.

2. Methodology

In a meta-analytical approach, 13 Swiss feeding and N-balance trials mostly taken from doctoral theses were analyzed to establish relationships between N-input, N-excretion and excretory pattern and diet characteristics. The N-balance trials were based on quantitative urine and feces collection for barn-fed cows. In case of pasture trials, the n-Alcane technique was applied to estimate feed intake and digestibility. 80 % of the data set represents repeated, individual cow observations which explain the extent of data variability. A subset of 31 dry cow observations as part of a larger study (Münger, 1997) was included in the evaluation. These dry cows were subject to a restrictive feeding regime with low protein supply and thus represent the low end of N turnover. The statistical evalua-tion focused on univariate regressions to describe relationships between dietary factors and excre-tory patterns. Where appropriate, data was grouped by diet type (season), lactation stage or study. In total, 404 data sets were evaluated which are distributed over the whole lactation cycle. Winter and summer diets are represented in almost equal parts.

3. Results and discussion

3.1 Animal, diet and excreta characteristics

Table 1 summarizes average and range of animal, diet and excreta characteristics. The applied winter and summer diets and performance levels reflect situations encountered under practical farming conditions. The large range in dietary protein supply comprising protein deficient (CP <120 g/kg DM) to protein surplus diets (CP > 180 g/kg DM) clearly affected the excretory patterns with a distinct seasonal influence. Summer diets produce on average 11 kg more manure than winter diets primarily the urine volume making the difference. The higher protein content and modified pro-tein/energy ratios of grassland based summer diets are important influencing factors. The contribu-tion of urinary N to total N excretion increased on average from 40 % to 60 % for winter and summer diets, respectively, underlining the higher emission potential of manure produced during the vege-tation period. This is supported by the augmented milk urea content indicating imbalances at the ruminal and animal level and thus causing a high N footprint of 15.3 g N/kg ECM and low N use efficiency of 27 %.

3.2 Manure volume and N excretion

N excretion is determined by N content in feces and urine and excreta volume. As shown in Figure 1, fecal N evolves little with dietary CP while urinary N concentration fluctuates considerably by a factor of up to 4. Assuming constant urinary concentration would lead to very inaccurate predictions of urinary N excretion. Moreover, the change in dietary protein content is accompanied by a shift in urinary N compounds. At low dietary CP content of 100 g/kg DM, the ratio of UUN/UN varied in our data set between 30 and 40 % while in diets exceeding 150 g CP/ kg DM, 70 – 80 % of urinary N is excreted as urea-N (data not shown).

Table 1: Average and range of animal, diet and excreta characteristics grouped by season and cow type

Parameter	**Lactating cows** **Winter diets**	**Lactating cows** **Summer diets**	**Dry cows** **Winter diets**
	Animal factors		
ECM, kg/day	23.0 (1.4-42.5)	25.8 (9.4-48.0)	-
N-milk, g/kg	5.52 (3.9-8.3)	5.51 (3.7-7.2)	-
Milk urea, mg/dl	21.9 (5.0-50.5)	30.9 (11.9-57.5)	-
	Dietary characteristics		
DMI, kg/day	16.6 (8.2-24.6)	17.6 (8.7-28.5)	7.5 (5.2-14.8)
DOM, g/kg DM	652.3 (539-746)	713.1 (587-795)	579.7 (523-651)
CP, g/kg DM	148.3 (98-239)	178.9 (101-259)	94.3 (65-186)
NEL, MJ/kg DM	6.25 (5.3-7.8)	6.61 (5.7-7.0)	5.1 (4.1-6.4)
Concentrate, % diet	18.4 (0-50)	13.5 (0-42)	0
	Excreta characteristics		
Feces volume, kg/day	38.5 (19-63)	34.9 (16-59)	20.4 (10-34)
Urine volume, kg/day	21.5 (6-51)	34.8 (16-65)	7.0 (3-18)
Fecal N, g/kg DM	27.6 (22-36)	32.1 (18-42)	18.5 (15-29)
Urinary N, g/kg	5.3 (2.4-11.8)	6.3 (3.0-12.3)	4.0 (2.1-8.0)
Excreta/DMI, kg/kg	3.8 (2.8-5.6)	4.6 (2.9-7.4)	3.7 (2.9-4.4)
	N turnover		
N intake, g/day	399.9 (157-758)	502.2 (219-824)	120.8 (65-440)
Milk-N/N-intake, %	30.7 (14-66.1)	26.9 (8.6-49)	-
Fecal-N/N-intake, %	37.9 (21.5-59.7)	28.2 (18.9-49.7)	52.2 (25.7-69.2)
UN/N-intake, %	26.1 (9.8-63.3)	43.9 (25.7-69.2)	24.0 (10.9-38.1)
UN/excreta-N, %	40.5 (14.8-64.4)	60.6 (35.2-75.2)	31.7 (17.4-48.3)
Excreta-N/ECM, g/kg	11.6 (5.6-28.8)	15.3 (5.2-47.1)	-

DM-dry matter; FM-fresh matter, ECM-energy corrected milk, DMI-dry matter intake, DOM-digestible organic matter, CP-crude protein, UN-urinary N

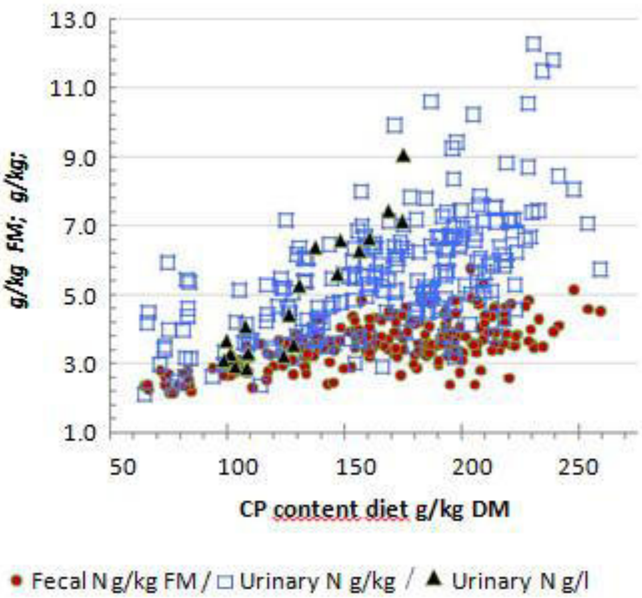

Figure 1: Relationship between dietary crude protein and fecal (g/kg FM) and urinary N (g/ kg, g/l) content in dairy cows

Manure output varied between 16 and 51.4 kg for dry cows and between 28.2 and 104.2 kg for lactating cows. Dry matter intake (DMI) can be used as a predictor of manure and feces output according to the following formulae: Manure all cows = 4.055*(DMI) + 1.0735 (R^2 = 0.71); Feces lactating cows = 2.1914*(DMI) + 2.8212 (R^2 = 0.56); Feces dry cows = 2.161*(DMI) + 2.9973 (R^2 = 0.64).

Considering the role of urine in regulating the osmotic balance and the excretion of protein and mineral metabolites, it can be expected that urine volume is more closely related to diet composition than feces which is confirmed by our findings. As shown in Figure 2, more urine is excreted at high protein diets. A power function is fitted to the data set of Münger (1997) describing the non-linear character of the relationship. At high dietary protein content, dry cows can produce as much urine as lactating cows.

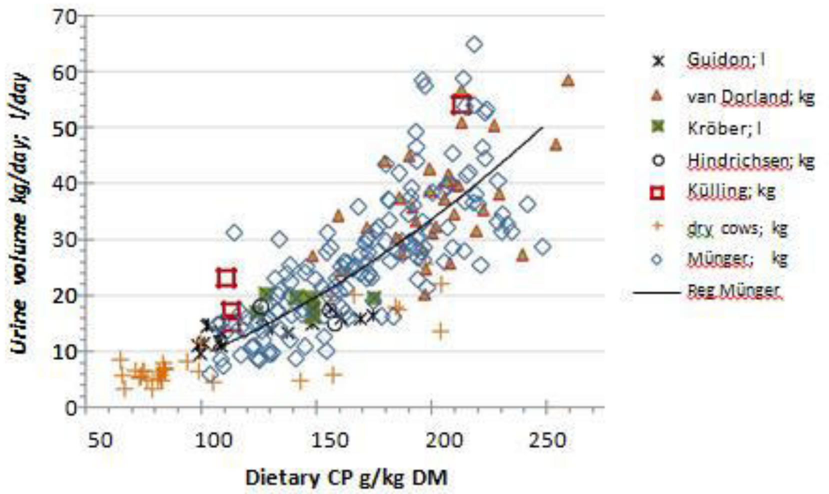

Figure 2: Relationship between dietary crude protein and urine volume in dairy cows. Urine volume (data set Münger) = $0.002*(CP)^{1.8401}$ (R^2 = 0.645).

As seen for feces volume, dry matter intake explained 80 % of the variation in fecal N excretion. Dry matter intake proved to be a better single predictor than N intake (Fecal N lactating cows (g/day) = 8.2048*(DMI) + 2.4795 (R^2 = 0.81)).

In contrast to fecal N, urinary N excretion increases with the dietary protein content in a non-linear manner (Figure 3). The power function fitted to the data set of Münger illustrates the general relationship. However, a certain study effect is visible. For a given protein content, the variation in urinary N output remains high pointing to additional influencing factors such as dietary imbalances between protein and energy supply at the ruminal and animal level.

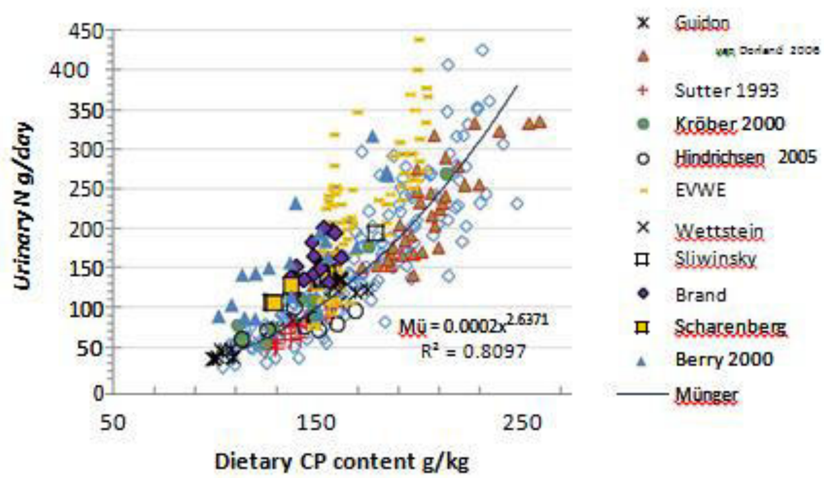

Figure 3: Relationship between dietary crude protein and urinary N in lactating dairy cows. Urinary N g/day (data set Münger) = $0.002*(CP)^{2.637}$ (R^2 = 0.81).

At the ruminal level, the so-called ruminal protein balance (RNBch = PMN-PME) quantifies the mi-crobial protein synthesis from fermentable N (PMN = CP*[1-{1.11*(1-deCP/100)}]) and energy sources (PME = 145*FOM). Excess ruminal N is one of the major causes for high urinary urea and N output (Figure 4). Ruminal imbalances are observed for both, winter and summer diets. The resulting uri-nary N output raises faster for summer diets compared to winter diets. The positive correlation also exists for dry cows. Analogous effects are observed for the protein/energy ratios CP/NEL and N/DOM (deCP = protein degradability; DOM = digestible organic matter; FOM = fermentable or-ganic matter).

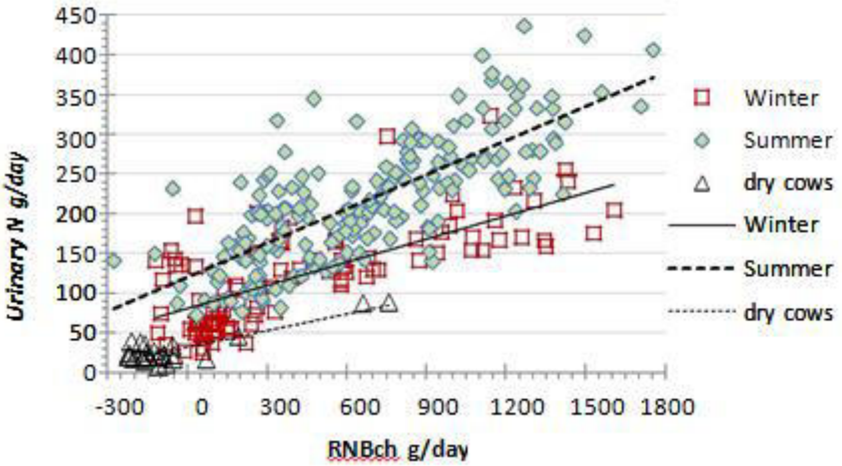

Figure 4: Relationship between dietary ruminal protein balance and urinary N excretion in dairy cows. Urinary N excretion winter (g/day) = 0.0975*(RNBch) + 80.27 (R^2 = 0.52); Urinary N excretion summer (g/day) = 0.1438*(RNBch) + 120.43 (R^2 = 0.58);

4. Conclusions and outlook

Feeding influences excreta characteristics and excretory patterns of dairy cows. Fecal N is closely related to dry matter intake while the highly variable urine characteristics correlate with protein and energy metabolism. Improving N use efficiency and reducing N losses first of all means minimizing avoidable urinary urea sources. N intake is a driving force for N losses which is superposed by pro-tein-energy interactions. For low N emissions, the various protein/energy ratios at the dietary and ruminal level and the total protein supply need to be optimized to match requirements. Particu-larly summer diets prove to be difficult to balance in cases of absent or limited use of energy rich feed stuffs and prevailing pasture systems. Established relationships can be used for modeling pur-poses. The extension of single regressions to multiple regressions will improve prediction error.

References

[1] Berry, N. R. (2000). Production efficiency and nutrient cycling of Brown Swiss dairy and Scottish Highland suck-lers on high altitude pastures under varied feeding conditions. Diss. ETH Nr. 13727, Zürich.

[2] Dikstra J., Oenema O., van Groenigen J. W., Spek J. W., van Vuuren A. M., Bannink A. (2013). Diet effects on urine composition of cattle and N_2O emissions. Animal, 7, 292-302.

[3] Hindrichsen, I. K. (2005). Methanogenesis in dairy cows and their slurry as affected by diet type and carbohy-drate characteristics. Diss. ETH Nr. 15852, Zürich.

[4] Külling, D. R. (2000). Influence of feed composition and manure type on trace gas emissions from stored dairy manure. Diss. ETH Nr. 13872, Zürich.

[5] Kröber, T. F. (2000). Methionine, lysine and leucine in dairy cows: limitations and their use to improve protein utilisa-tion and minimise nitrogen losses from manure. Diss. ETH Nr. 13568, Zürich.

[6] Münger, A. (1997). Energie- und Stickstoffverwertung bei Milchkühen verschiedener Rassen. Diss. ETH Nr. 11929, Zürich.

[7] Reijs, J. W. (2007). Improving slurry by diet adjustments: a novelty to reduce N losses from grassland based dairy farms. PhD thesis, Wageningen University, The Netherlands.

[8] Sutter, F. (1993). Einfluss einer reduzierten Proteinversorgung auf den Protein- und Energieumsatz von Milch-kühen bei Laktationsbeginn. Diss. ETH Nr. 10101, Zürich.

[9] Van Dorland, A. H. (2006). Effect of white clover and red clover addition to ryegrass on N use efficiency, per-formance, milk quality, and eating behaviour in lactating dairy cows. Diss. ETH Nr. 16867, Zürich.

[10] Van Duinkerken G., Smits M. C. J., André G., Vereijken P. F. G., Sebek L. B. J., and Dijkstra J. (2011). Milk urea concentration as an indicator of ammonia emission from dairy cow houses in a situation with restricted grazing. J Dairy Sci. 94, 321-335.

TA-O_19 Innnovative technique to improve phosphorous recycling yield from wastewater treatment plant (WWTP) sludge by producing high quality fertilizer (struvite)

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1. Objectives

The goal of our work is to improve P recycling as quality fertilizer (struvite) by adapting an innovative process to recover both soluble and immobilized phosphorus (P) from wastewater treatment sludge [1][2]. The innovation was to favoring both biological and chemical P dissolving. The objectives of this study were to optimize dissolving of Pand to assess suitability of the liquid to enter the struvite crystallization step.

2. Methodology

Forms of P in waste activated sludge (WAS): The cold perchloric acid fractionation method [3] was used to characterize P WAS from 6 different WWTP based on chemical (S3 and S4) and/or biological P removal (S1, S2, S5, S6).

Chemical acidification: Tests were run to estimate the amount of P chemically dissolved during biological tests. Hydrochloric acid was added and the pH was maintained 15 mn at each pH value under continuous stirring and then lowered from 1 unit till reaching 2. Acidified WAS was sampled at every pH unit, cations and anions concentrations were measured in the supernatants. The equation of the curve dissolved P/pH was used to calculate the part of P chemically dissolved during the biological tests.

Biological phosphorus dissolution potential assessment: A test to assess the biological phosphorus dissolution potential (BPDP) from a sludge was designed. A mixture of sludge to be tested and white sugar (sucrose) was kept in anaerobic conditions under steering during ten days in specific vials allowing sampling without air introduction.

BPDP of model and real co-substrates: To estimate the potential contribution of co-substrate to P dissolution depending on their biochemical composition, BPDP from model and real co substrates were measured. Dried white egg (mainly ovalbumine) was used as model for proteins and butter for lipids. Waste from an organic vegetarian restaurant, organic fraction of household waste, lactoserum, carrots, apples and flotation greases from WWTP were tested as “real” co-substrate.

Analysis: Total solids (TS), volatile solids (VS), total Kjeldahl nitrogen (TKN) and chemical oxygen demand (COD) were measured using standard methods (APHA, AWWA, WEF, 1998).
TP was analyzed with ascorbic acid method using automate colorimetric methods on QuikChem® FIA+ from Lachat Instruments with QuikChem method 10-115-01-1-P after mineralisation by a mix of nitric and sulfuric acid (1/4) under pressure (1bar) at 110°C during 1 hour.
Ionic composition of supernatant was measured with anion and cation chromatography with Metrohm 940 Profesional Vario IC respectively with Metrosep A sup 5 and Metrosep C4 -250/4,0 columns. Volatile fatty acids (VFA) were analysed as described by Peu et al.[4].

3. Results and discussion

3.1 Forms of phosphorus in WAS

The different mechanisms involved in dissolving P from WAS by a biological process are P release by phosphate accumulating organisms (PAO), chemical dissolution of mineral solid P and hydrolysis of dissolved organic matter due to pH decrease induced by volatile fatty acid production (expected pH around 4.5). The relative part of each of these processes will depend on forms of P in WAS. In sludge from pure chemical treatment (S3 and S4) the amount of insoluble P was more than 85%, probably mainly as iron or alum salts. Soluble and solid mineral P soluble at pH 4.5

were 20-45% of the total P in sludge from WWTP using biological P removal (S1, S5 and S6). S2 was described as a biological process but a large part of the influent was coming from food industry using FeCl3 as pre-treatment. So forms of P were close to those in the chemical P removal process sludge. Due to the biological P removal process, insoluble P should be mainly stored as polyphosphate which could be released by available carbon addition so the best results were expected from WAS from S6 (Figure 1).

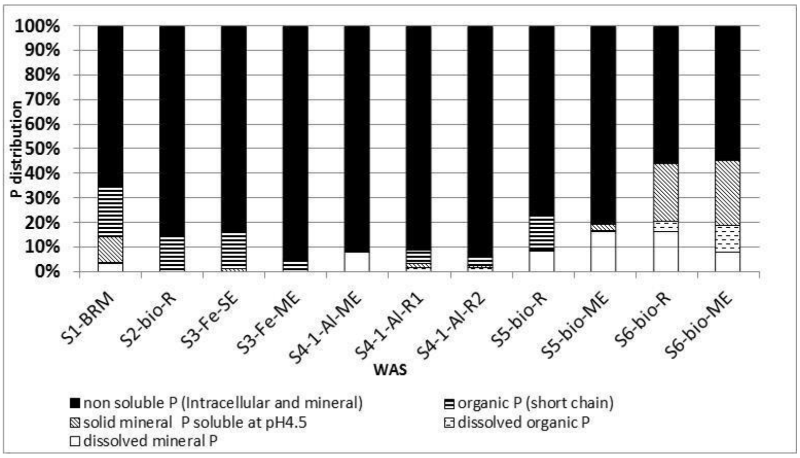

Figure 1: Forms of P in sludge (BRM: membrane bio reactor, bio : biological P removal, Fe and Al: chemical P removal, R recirculated, SE : secondary sludge after thickening, ME : mix of primary and secondary sludge after thickening)

3.2 Chemical acidification

Chemical acidification was performed on the mixed sludge from S3, S4, S5 and both on the recirculated and mixed sludge from S6. As expected, chemical dissolution rate was low for sludge from chemical P removal process (S3 and S4) when pH was above 3. The best P dissolution rate was measured for diluted biological WAS (S6-bio-R). However soluble P concentrations hardly reached the 50 mg/L threshold set by Cornel et al [5] for process profitability and it is thus preferable to implement it on concentrated sludge. So S6-bio-ME was used for further experiments.

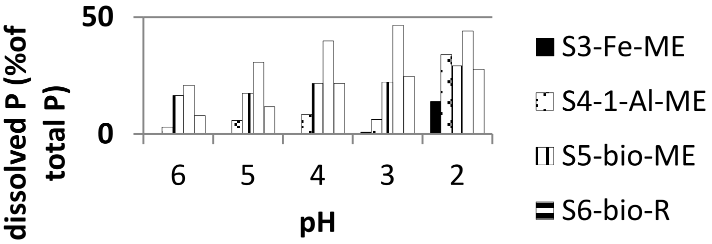

Figure 2: Chemical dissolution rate (bio : biological P removal, Fe and Al: chemical P removal, R recirculated, ME : mix of primary and secondary sludge after thickening)

The evolution of dissolved P concentration, depending on pH during chemical acidification, can be described by a polynomial function. The curve obtained from S6-bio-ME is presented in Figure 3.

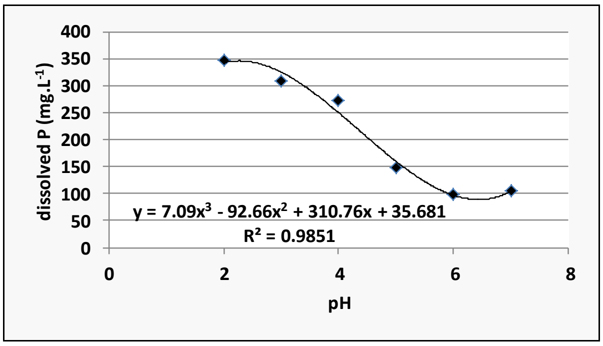

Figure 3: Dissolved P depending on pH during chemical acidification tests (S6-bio-ME)

3.3 Biological phosphorus solubilizing potential (BPDP)

Tests with sucrose ($BPDP_{max}$): A first set of experiments had shown that an inoculum obtained by maintaining WAS under acidifying conditions (feed with sugar during several days) did not improve significantly the kinetic and the amount of dissolved P so further tests were performed without inoculum (results not shown).

Several coS/S ratios were tested. Low pHs (3.5) were reached at the higher loads (2 and 5 gCOD/gVS) which was related to the higher production of VFAs. The pH quickly stabilized (between 24-48 h) around 5.7, 3.4 et 3.3 for the loads of 0.1, 2 and 5 gCOD/gMV, respectively. For the higher loads, a first maxima of dissolved P was quickly reached (~400 mg/L at 24 h) followed by a slight diminution which could be due to inhibition and a necessary adaptation on several days, and another maxima around 540 mg/L at 144 h. In the test at 0.5 gCOD/gVS, soluble P seemed strongly related to pH variations. A pH minimum was observed at 48 hours (about 4) corresponding to the maximum dissolved P which was closed to 530 $mg.L^{-1}$ (figure 4). For further experiments the coS/S was 0.5 gCOD/gVS.

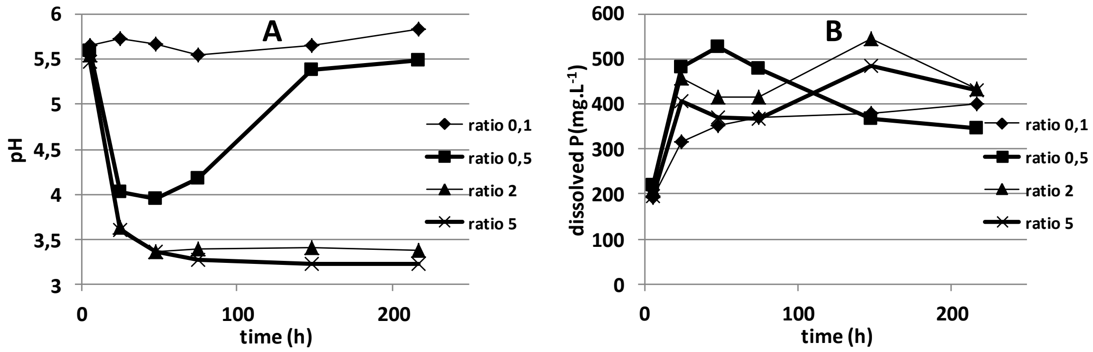

Figure 4: pH (A) and dissolved P (B) during BPDP tests

In conclusion the best results ($BPDP_{max}$) were obtained with white sugar, without inoculum, at 35°C, with a co-substrate/substrate ratio (coS/S) of 0.5 (gCOD/gVSS) between 24 and 48 h.

Tests with model co-substrates:.Depending on the COD/VS ratio, pH remained between 6 and 7 for proteins and between 5 and 6.5 for lipids. Nonetheless rather high soluble P concentrations were observed (416- 493 mg/L) for proteins which means that P release was not only induced by acidification. Similarly to sucrose, maxima was quickly reached for higher loads but with a slightly longer time (48h) compared to sucrose. This may be explained by the fact that ovalbumin and butter are more complex molecules which have to be degraded before to be used as short chains fatty acids by PAO.

The part of chemical dissolved P during those tests was calculated from the equation obtained in figure 3. When dissolved P was maximal, the part released by the metabolism was 40 % for sucrose and 46 and 48% for proteins and lipids. The difference of dissolved P was due to chemical dissolution and was explained by the higher pH reached with proteins and lipids (figure 5). The hypothesis was that the biological P release was higher when pH was above 5 but that to keep P dissolved, pH had to be lower than 5. This was confirmed by a new set of experiments, repeated under controlled pH at 3, 4, 5 and 6 by adding hydrochloric acid or sodium hydroxide. The VFA analysis had shown that formic acid accumulation was responsible for lower pH reached with sucrose and that the low pH inhibited formic acid catabolism.

Finally the BPDP of dried white eggs and butter were calculates as 67 and 72% of the BPDP max.

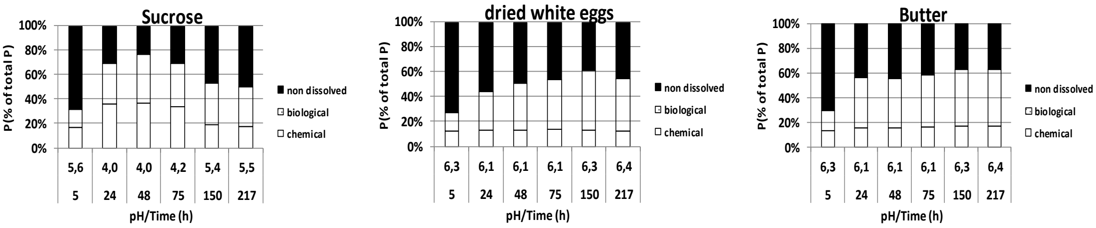

Figure 5: part of P chemically or biologically dissolved during tests.

Tests with wastes as co-substrates: In order to better understand the mechanisms and to classify the waste depending on their potential to dissolve P during the hydrolysis/acidifying step an experimental design to study mixture of model co-substrates in different proportion and 6 wastes that could be used in full scale process have been tested. A model calculating pH and dissolved P from the biochemical composition of the waste was obtained from the experimental design. Some trends could be drawn when it was applied to wastes as co-substrates but it has to be implemented to be used as a real predictive tool.

The results of the BPDP tests performed on co-substrates are presented in table 1.

Table 1: Biochemical composition and BPDP of six wastes that could be used as co-substrates to initiate P biological dissolution.

Sample	Organic restaurant	Household organic waste	Lactoserum	Flotation greases	Carrots	Apples
COD	0.3	0.3	0.06	0.12	0.14	0.08
Carbohydrates	60	31	78	45	89	95
Proteins	8	22	19	6	6	2
Lipids	32	47	3	49	5	3
BPDP (%BPDPmax*)	41	42	49	38	68	61

* BPDPmax was 530mg.L^{-1} for the WAS tested in this study.

As expected, sweet co-substrates were the most efficient to dissolve P because of their capacity to lower the pH and to keep in solution the P biologically released. In all the cases the amount of ammoniacal nitrogen produced by mineralization that occurred during the biological acidification aimed to a N/P ratio above 1.5 which is considered to be the one that maximizes struvite crystallization from the liquid phase [2].

4. Conclusion and outlook

This study had shown that, depending on the biochemical composition of the co-substrate, biological acidification allowed to dissolve up to 75% of the P in WAS from combined biological and chemical P removal process. The COD brought by the co-substrate promoted Prelease by PAO and low pH, reached with sweet co-substrates, contributed to keep the P released dissolved. The pH was driven by the metabolism of formic acid. Further experiments should allow bettering understanding of the mechanisms in order to proposing ways for favoring P dissolution during the process. Nitrogen mineralization was enough to obtain the N/Pratio required to struvite crystallization.

References

[1] M.-L. Daumer, S. Picard, P. Saint-Cast, P. Dabert, Technical and economical assessment of formic acid to recycle phosphorus from pig slurry by a combined acidification-precipitation process, Journal of Hazardous Materials. 180 (2010) 361-365.

[2] Capdevielle, A., Sykorova, E., Biscans,B., Beline,F. Daumer, M-L.(2013). Optimization of struvite precipitation in synthetic biologically treated swine wastewater—Determination of the optimal process parameters, J.of Hazardous Material, 244-245, 357-369

[3] M.-L. Daumer, F. Béline, M. Spérandio, C. Morel, Relevance of a perchloric acid extraction scheme to determine mineral and organic phosphorus in swine slurry, Bioresource Technology. 99 (2008) 1319-1324.

[4] Peu P., Beline F., Martinez J. (2004). Volatile fatty acids analysis from pig slurry using high-performance liquid chromatography. International Journal of Environmental Analytical Chemistry, 84 (13), 1017-1022

[5] Cornel, P. & C. Schaum (2009) Phosphorus recovery from wastewater: needs, technologies and costs. *Water Science & Technology,* **59**(6)**,** 1069–1076.

TA-O_20 Phosphorus fertilizer from sewage sludge ashes by thermochemical treatment – Benefits and challenges

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1. Objectives

Mineral phosphorus (P) fertilizers are solely produced from phosphate rock, a fossil and thus limited resource. Europe completely depends on imports since there are no relevant phosphate rock deposits on the continent. Furthermore, phosphate rock is contaminated with heavy metals such as Cd and U that pollute the farmlands and pose environmental risks [1]. In Germany, more than 560,000 t/a of phosphorus (P) is removed from the soils by farming and husbandry and thus has to be replaced. About 150,000 t/a P of this are applied as mineral fertilizer [2]. Sewage sludge ash (SSA) might be a promising source for recycling fertilizer since it contains large amounts of P (up to 13 %). However, fertilizer from SSA has to comply with the respective ordinances, particularly the heavy metal limit values stated in the fertilizer ordinance, and requires sufficient P bioavailability. Thus, we conducted a complete survey of SSA from German mono-incineration facilities [2] and developed a thermochemical treatment for SSA to reduce toxic elements and increase P bioavailability.

2. Methodology

Approximately 300,000 t/a SSA accrue in 26 German mono-incineration facilities. 24 of these took part in the survey, mostly with monthly samples covering a period of one year. Taking into account the respective ash amounts, we covered more than 97 % of all German SSA enabling a complete assessment of the SSA in terms P recovery potential, heavy metal content and P bioavailability. SSA samples were dried, grinded, homogenized, and dissolved with microwave-assisted digestion (HNO_3/$HClO_4$/HF). The elemental composition was determined with inductively coupled plasma optical emission spectroscopy and mass spectroscopy, respectively (ICP-OES; ICP-MS). P solubility in neutral ammonium citrate (P_{NAC}) was chosen as indicator for P bioavailability and determined according to the respective standard.
SSA was treated in a rotary kiln at 900 - 1000 °C under reducing conditions (Figure 1). Sewage sludge (SS) was added as reducing agent to the SSA. Under reducing conditions some volatile toxic elements such as As, Cd, Hg and Pb were separated via gas phase. Additionally, alkaline additives (sodium or potassium -sulfates, -carbonates or -hydroxides) were added for increasing the P bioavailability because they react with the Ca- and Al-phosphates present in the SSA to highly bioavailable mineral phases such as $NaCaPO_4$ or $KCaPO_4$. The thermochemical process was investigated in lab-scale in crucibles, was optimized in medium scale rotary kilns and finally demonstrated in an industrial production campaign. The raw materials sewage sludge ash (SSA), sewage sludge (SS) and sodium sulfate were mixed in the ratio 5:1:2. The product of the demonstration trial is currently tested in a field trial.

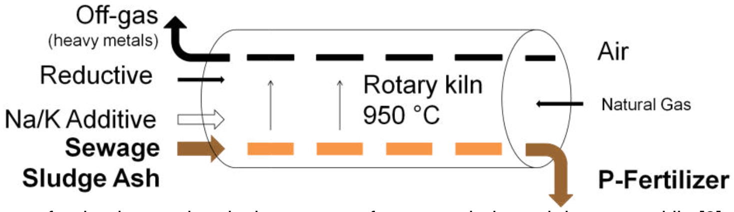

Figure 1: Process scheme for the thermochemical treatment of sewage sludge ash in a rotary kiln [8]

3. Results and discussion

3.1 Survey for German sewage sludge ashes

Table 1 shows mass fractions of phosphorus and selected trace elements in German sewage sludge ashes. The median P concentration in SSA was 7.9 %. The P concentrations are further separated according to source of the respective sludge. SSA that originated solely from municipal sludge contained 9.0 % P in the mean, whereas those from industrial sources showed the lowest concentrations with 2.3 %. SSA from both municipal and industrial sources was in between with 4.9 % [3]. This was expected, since the majority of P in the wastewater stream originates from human excreta and dishwashing detergents. Taking into account the P concentrations in the SSA and the respective ash amounts, this results in an annual P recovery potential of up to 19,000 t P [3]. Again, the majority of this potential is with SSA from municipal sludge (approximately 11.000 t/a P), whereas SSA from mixed sources contribute about 7.500 t/a P. SSA from industrial sources is insignificant on that account (500 t/a P). Thus, more than 12 % of the so far required mineral fertilizer could be replaced by secondary phosphates. The concentrations of Cd and U in the ashes (3.3 mg/kg and 5.8 mg/kg) are significantly lower than in phosphate rock and mineral fertilizers. By using SSA as secondary raw material for fertilizer production, the input of these harmful substances to farmlands might be significantly reduced. Furthermore, table 1 shows element concentrations relevant for fertilizer use and the respective labeling and limit values as stipulated by the German fertilizer ordinance. Whereas the median values are all well below the respective limit values, several single values were above. Taking into account the respective ash amounts this means that two thirds of SSAs are not suitable for direct fertilizer use due to heavy metal contents above the limit values. Furthermore, the P_{NAC}-solubility mean value of SSA was only 31 % but should be towards 100% for fertilizer application. Thus, further treatment is inevitable in most cases to reduce inorganic contaminant content and increase P bioavailability.

Table 1: Phosphorus- and trace element mass fractions in German sewage sludge ashes. Closer information in [3] and [4]

Element [% DM]	Min	Max	Mean	Median	No. of samples	P recovery potential [t/a]
P	1.5	13.1	7.3	7.9	252	18,812
P (municipal)	3.6	13.1	9.0	9.1	163	10,939
P (municipal/Industrial)	2.8	7.5	4.9	4.8	69	7,319
P (industrial)	1.5	3.8	2.3	2.3	20	554
Element [mg/kg DM]	**Min**	**Max**	**Mean**	**Median**	**No. of samples**	**Limit value [5]**
As	4.2	124.0	17.5	13.6	252	40
Cd	<0.1	80.3	3.3	2.7	252	9 1)
Cr	58	1502	267	159.7	252	-
Cu	162	3467	916	785	252	900
Hg	0.1	3.6	0.8	0.5	143	1
Ni	8.2	501	105.8	74.8	251	80
Pb	<3.5	1112	151	117	252	150
U	1.6	26	5.2	4.9	252	-
Zn	552	5515	2535	2534	252	5000

DM-dry matter

Calculated for P= 7.9 % because limit value is stipulated as 50 mg Cd / kg P_2O_5

3.2 Thermochemical treatment of sewage sludge ashes in a rotary kiln

The two main goals in the thermochemical treatment of sewage sludge ash is increasing P bioavailability and decreasing the mass fraction of toxic trace elements. In order to assess different additives, it is displayed over the ratio alkali/P of the starting material mixture in Figure 2. Generally, the P_{NAC}-solubility of SSA (35 %) could be significantly increased for alkali/P ratios > 1. P_{NAC}-solubility > 85 % was achieved by treatment with Na_2SO_4, Na_2CO_3, NaOH, K_2CO_3 and KOH. The surplus of potassium addition to achieve high P_{NAC}-solubility compared to sodium addition is necessary because potassium is partly lost via off gas. The lower P_{NAC}-solubility for K_2SO_4 (maximum 70 %) is linked to the high melting point of K_2SO_4 (1070 °C) which is above the temperature of the thermochemical treatment that requires melt phases for reaction.

Thermochemically treated SSA was tested in plant experiments (variants with Na_2CO_3 and Na_2SO_4). The fertilizer effect of the ash based products was comparable to triple superphosphate [7]. The current extensive pot and field tests in the EU project P-REX further support the fertilizer effect of the thermochemical products. The results is published in the deliverable D 8.1 on the homepage of the EU-project P-REX (www.p-rex.eu).
The described thermochemical process with Na_2SO_4 was successfully tested in a demonstration trial with an output of 2 t recycling fertilizer. The product had a P content of 7.7 % and a high P_{NAC}-solubility (82 %) in the representative product period (Table 2). In general, the mass fraction in the calcined product for matrix elements and non-volatile heavy metals (Cr, Cu, Ni and U) were 80 +/- 2% of the mass fractions in SSA (dilution by factor 1.25). The reason for this is the dilution of the SSA by the addition of Na_2SO_4. Additionally, the mass fraction of the volatile elements As, Cd, Hg and Pb were significantly decreased by evaporation under the reducing conditions in the kiln. The removal calculated on the mass balance was 61 % for As, 80 % for Cd, 68 % for Hg, 39 % for lead and 9 % for Zn [6].

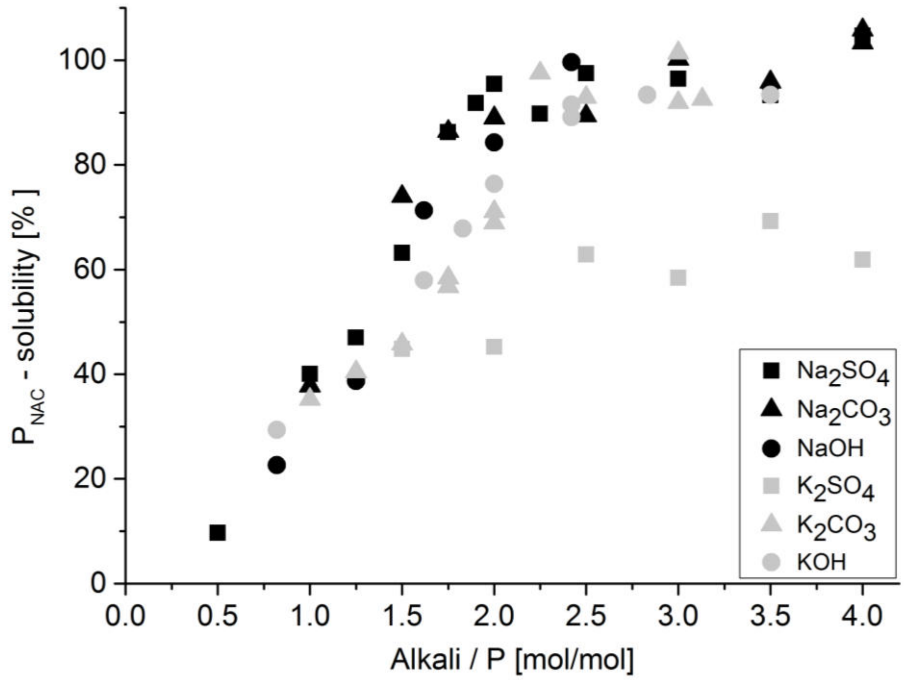

Figure 2: P_{NAC}-solubility of products from SSA calcined at 1000°C with Na_2SO_4, Na_2CO_3, NaOH, K_2SO_4, K_2CO_3 and KOH as a function of Na/P or K/P ratio [mol/mol].Na/P or K/P ratio corresponds to the molar ratio in the starting material [6]

Table 2: Mean elemental compositions and standard deviations of SSA (n = 3), SS (n = 3) and product samples (n = 7-10) for representative period in the production campaign of 2 t recycling fertilizer [6]

	SSA		**SS**		**Product**	
	mean	SD	mean	SD	mean	SD
	mg/kg					
P	93,700	3160	42,000	840	77,100	1700
P_{NAC}	35,100	1100	-		62,800	1400
As	11.1	0.4	4.2	0.2	3.6	1.7
Cd	2.1	0.2	0.6	0.1	0.3	0.2
Cr	159	11	43.7	0.9	127	5
Cu	767	43	204	4	601	8
Hg	1.1	0.1	0.4	0.1	0.3	0.1
Ni	73.3	3	17.7	0.4	56	3
Pb	123	0.8	29	1.3	60	5
U	11.2	0.3	2.1	0.1	7.2	0.1
Zn	2330	29	621	7	1710	84

4. Conclusion and outlook

German sewage sludge ashes show a significant P recovery potential and are able to reduce the uranium and cadmium input to farmlands significantly. However, additional action or treatments might be necessary to reduce the heavy metal content and to increase the P bioavailability.

The thermochemical treatment in a rotary kiln with alkali additives is a suitable process to transform sewage sludge ash into a product which could be used as P-fertilizer. The P-bioavailability is increased to a level comparable to TSP. The content of heavy metals is below the limit values of the German fertilizer ordinance.

References

[1] Dissanayake, C. B.; Chandrajith, R. (2009): Phosphate mineral fertilizers, trace metals and human health, Journal of the National Science Foundation of Sri Lanka, 153-165.

[2] Gethke-Albinus, K. (2012): Verfahren zur Gewinnung von Sekundärphosphaten aus flüssigen Stoffströmen und deren Einfluss auf die deutsche Phosphorbilanz, PhD thesis RWTH Aachen.

[3] Krüger, O., Adam, C. (2015): Recovery Potential of German Sewage Sludge Ash, Waste Management, in press

[4] Krüger, O., Grabner, A.,Adam, C.. (2014) Complete survey of German sewage sludge ash, Environmental Science and Technology 48, 11811-11818.

[5] Düngemittelverordnung (2012): Verordnung über das Inverkehrbringen von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten und Pflanzenhilfsmitteln (Düngemittelverordnung - DüMV)

[6] Herzel, H.; Krüger, O.; Hermann, L., Adam, C. (2015): Sewage sludge ash – a promising secondary phosphorus source for fertilizer production, Science of the Total Environment, under revision

[7] Severin, M., Breuer, J., Rex, M., Stemann, J., Adam, C., Van den Weghe, H., Kücke, M. (2014): Phosphate fertiliser value of heat treated sewage sludge ash. Plant Soil and Environment; 60, 555 - 561.

[8] Herzel, H.; Adam, C.; Hermann, L.; Mallon, J.; Schaaf, M.; Eicher, N.; Stemann J. (2015): Comparative review of ash processes. P-REX deliverable D4.1, available for download here: www.p-rex.eu

TA-O_21 Innovative bioresource management technologies for recovery of ammonia and phosphorus from livestock and municipal wastes

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1. Background

The recovery of nutrients from wastes for re-use as concentrated plant fertilizers is a new paradigm in agricultural and municipal waste management. Currently the potential impact of manure on the environment represents one of the world agriculture's major challenges. Many areas of intensive livestock production generate more manure nutrients than available cropland can assimilate due to a net import of nutrients. Nutrient pollution is one of the most widespread, costly and challenging environmental problems in industrialized nations. It is caused by too much nitrogen (N) and phosphorus (P) in the environment. Nutrient pollution has diverse and far-reaching effects on the economy, impacting many sectors that depend on clean water. The U.S. tourism industry loses $1 billion each year, mostly from losses in fishing and recreational activities because of nutrient-polluted water bodies [1]. As result of phosphorus pollution, algal blooms in drinking water sources can drastically increase treatment costs and shortages in water supplies. The agricultural sector remains the major source of NH_3 emissions in both Europe and USA. Ammonia emissions from animal husbandry operations (dairy, beef, poultry and swine) were estimated at 2.5 million tons/year in 2015 [2]. Therefore, significant efforts are required to abate ammonia emissions from livestock operations. In this context, new technologies to recover the N and P are needed. The aspect of N and P reuse is also important for farmers because of increasing demand and cost of inorganic fertilizers. Fertilizer prices have escalated in recent years, thus there is renewed interest on developing technologies to recover and recycle nutrients from wastes.

2. Objective

Treatment technologies can play an important role in the management of livestock manure by providing a more flexible approach to land application and acreage limitations and by solving specific problems such as odors, pathogens, water pollution, ammonia emissions, greenhouse gas emissions, and phosphorus and heavy metal contamination of soils. In this paper we show development of systems and methods by USDA to recover nitrogen, phosphorus and value-added materials from livestock wastes.

3. Recovery of Materials and Nutrients

Advanced treatment technologies are being developed to address manure problems in confined livestock production. The examples show a shift from municipal treatment methods in the near past to a new body of knowledge with methods adapted to the specific characteristics of these wastes and a different purpose for treatment. Further, for the first time we are seeing technologies developed for agricultural waste crossing the discipline boundaries and being adopted by the municipal wastewater treatment industry. For example, the flow diagram in Figure 1 illustrates three main alternative approaches to swine manure management. The first approach is to develop dry systems such as the deep bedding where fresh manure is mixed with a bulking agent or use inclined belts under the slatted floor to separate urine and solids so that all or part of the manure leaving a building is directly handled as a solid. The second approach is to improve or retrofit existing liquid systems so that volatile solids and organic nutrients are recovered and separated from the waste stream and treated with a variety of technologies to generate value-added products. Solid-liquid separation of the raw manure increases the capacity of decision making and opportunities for treatment. The separation up-front allows recovery of the organic compounds, which can be used for the manufacture of compost materials and other value-added products or energy production. These products may include stabilized peat substitutes, plastic composites, humus, biochars, bio-oils, organic fertilizers, amino-acids, soil amendments, energy, and quick-wash phosphorus. The remaining liquid needs to be treated on the farm. A variety of biological, physical or chemical processes can be used to achieve specific nutrient management goals and environmental standards that may include recovery of phosphate and ammonia concentrates.

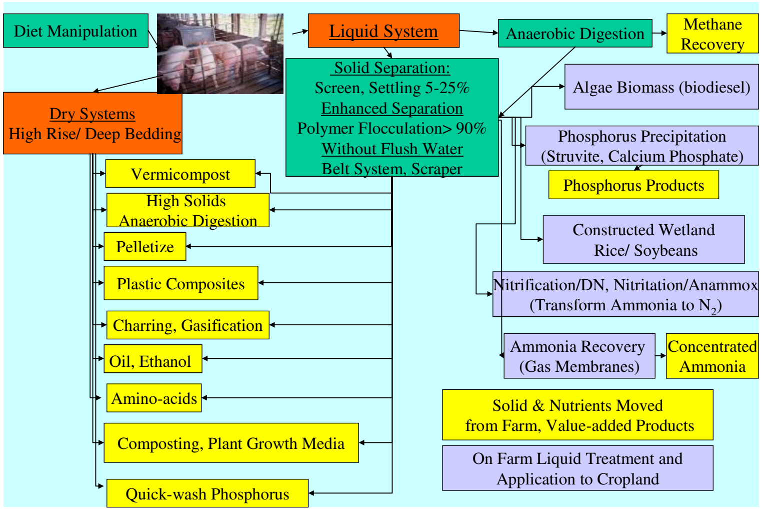

Figure 1: Alternative technologies to manage excess nutrients and create value added products from swine wastes.

A third approach is to use anaerobic digesters (AD) to recover methane and energy from the carbon in the liquid manure. The biogas recovery systems collect methane from the manure and burn it to generate electricity or heat. Production of biogas from manure using anaerobic digesters is projected to be important worldwide. However, in areas of intensive livestock production, new technologies need to be developed in conjunction with AD to address surplus nitrogen and nitrogen removal and/or recovery of concentrated phosphorus from AD effluents in a form that can be removed from the watershed.
In addition to nutrients (e.g., nitrogen and phosphorus), manure and wastewater from animal feeding operations have the potential to contribute other pollutants such pathogens, heavy metals, odor and ammonia to the environment. So a combination of two or more treatment technologies is often needed to meet multiple environmental goals.

3.1 Solids and phosphorus recovery in Third Generation EST system

We tested several configurations of a calcium phosphate recovery process on swine farms showing consistent results producing a marketable fertilizer with > 90% plant available P [3]. It was technically feasible to flocculate and dewater both the P and raw manure in a simultaneous operation, significantly reducing cost of equipment [4]. The latest full-scale project evaluated and demonstrated the viability of a third generation Environmentally Superior Technology (EST) manure treatment technology [5]. It was designed to further reduce cost of manure treatment through pre-concentration of diluted manure before polymer application. The technology first separated solids and liquids in the flushed manure with the aid of settling and polymer flocculants; subsequently, the ammonia nitrogen was treated with nitrifying bacteria adapted to high-strength wastewater and cold temperatures; and lastly, the soluble phosphorus was separated via calcium phosphate precipitation in the absence of ammonia (Figure 2). This combination of treatments substantially eliminated the release into the environment of odors, pathogens, ammonia and heavy metals, which are environmental standards required for new or expanding swine operations in North Carolina. A decanting tank was effective to concentrate the solids in the diluted flushing manure stream and reduced the total manure volume processed by the solid separator press; in addition, it increased polymer use efficiency 5.4 times. On a mass basis, the system removed from the liquid 98.6% of the total suspended solids (TSS), 100% of ammonia, 91.9% of total phosphorus, 95.4% of copper, 97.0% of zinc and 100% of odor compounds including skatole, phenol, cresol, indole, and volatile fatty acids [5].

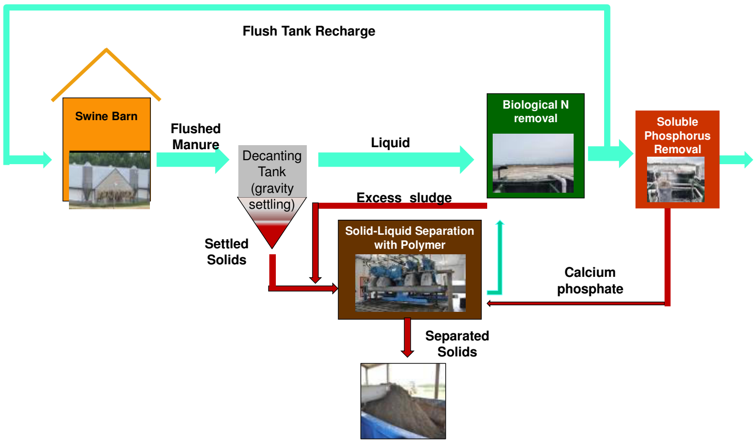

Figure 2: Recovery of solids and phosphorus from diluted swine effluents in a third generation Environmental Superior Technology

3.2 Phosphorus recovery from solid manures and biosolids (Quick-Wash Process)

The "quick wash" process was developed for extraction and recovery of phosphorus from poultry litter and animal manure solids, but research has shown that the approach is equally effective with municipal biosolids [6]. In the quick wash process, phosphorus is selectively extracted from solid manure or poultry litter by using mineral or organic acid solutions. Following, phosphorus is recovered by addition of liquid lime and an organic poly-electrolyte to the liquid extract to form a calcium-containing P precipitate (Figure 3).

Figure 3: Quick-wash process for extraction of phosphorus from waste (www.quickwash.com)

The process generates two products: 1) washed solid residue, and 2) concentrated recovered phosphorus material. The quick wash process selectively recovers more than 80 % of the phosphorus from solid waste while leaving most of the nitrogen in the washed solid residue. Consequently, the washed solid residue has a more balanced nutrient composition for crop production and environmentally safe for land application. The concentrated phosphorus materials contain more than 90% of its phosphorus in plant available form that provides a recycled phosphorus source for use as crop fertilizer. The inclusion of this process in a waste management system offers producers and municipalities a new opportunity to recover P as a valuable product.

3.3 Ammonia Recovery

The recovery of nitrogen (N) from wastes will be important in agriculture because of the high cost of commercial N fertilizers and the environmental damage of the release of reactive nitrogen. We are developing new systems and methods that use gas-permeable membranes at low pressure to recover significant amounts of ammonia when operated in: 1) liquid manure effluents such as lagoon, AD and raw liquid manure [7, 8], and 2) barns to remove the ammonia from the air [9, 10].

The new process includes the passage of gaseous ammonia through a micro-porous hydrophobic membrane and subsequent capture and concentration in a stripping solution on the other side of the membrane. For liquid applications, the membrane manifolds are submerged in the liquid, and the gaseous ammonia is removed from the liquid before it escapes into the air (Figure 4). For air applications, the membrane manifolds are suspended above the litter, and the gaseous ammonia is removed inside the barns close to the litter (Figure 5).

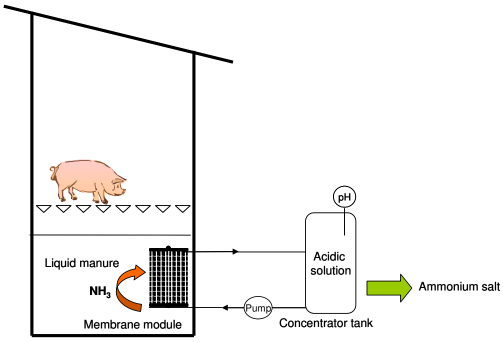

Figure 4: Gas-permeable membrane system to harvest the nitrogen from liquid manure and reduce ammonia emissions.

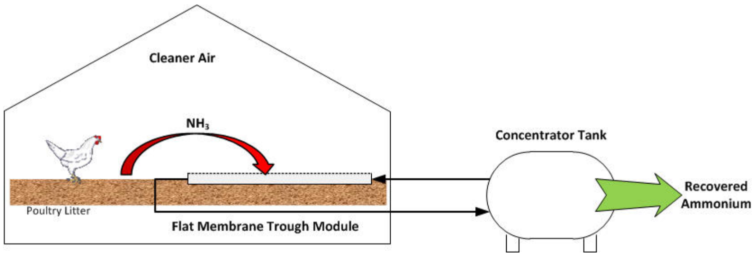

Figure 5: Gas-permeable membrane system to harvest the nitrogen from liquid manure and reduce ammonia emissions.

4. Conclusions

The recovery of N and P from wastes will be increasingly important in agriculture because of the high cost of fertilizers and the environmental damage from the release of reactive N and P. Treatment technologies have been developed to address manure nutrient recovery. Many are now in the commercialization stage after years of extensive on-farm testing. The examples show a shift from municipal treatment methods in the near past to a new body of knowledge with methods adapted to the specific characteristics of these wastes and a different purpose for treatment. Further, for the first time we are seeing technologies developed for agricultural waste crossing the discipline boundaries and being adopted by the municipal wastewater treatment industry.

References

[1] EPA (2015): Nutrient pollution. http://www2.epa.gov/nutrientpollution/

[2] EPA (2014): National emission inventory—Ammonia emissions from animal husbandry operations. Draft report, January 30, U.S. Environmental Protection Agency.

[3] Vanotti, M.B.; Garcia; M.C.; Szogi, A.A.; Hunt, P.G.; Millner, P.D. (2012): Method for Recovery of Phosphorus from Animal Wastewater, WEFTEC 2012, 85th Annual Water Environment Federation Conference, New Orleans.

[4] Vanotti, M.B.; Szogi, A.A.; Fetterman, L.M. (2010): Wastewater treatment system with simultaneous separation of phosphorus and manure solids, U.S. Patent 7,674,379, U.S. Patent and Trademark Office.

[5] Vanotti, M.; Hunt, P.; Rice, M.; Kunz, A.; Loughrin, J.; Williams, M. (2014): Generation 3 treatment technology for diluted swine wastewater using high-rate solid-liquid separation and nutrient removal processes, Paper Number 1901300, Proc. of ASABE Conference, Montreal.

[6] Szogi, A.A.; Vanotti, M.B.; Hunt, P.G. (2014): Process for removing and recovering phosphorus from animal waste, U.S. Patent 8,673,046, U.S. Patent and Trademark Office.

[7] Vanotti, M.B.; Szogi, A.A. (2015): Systems and methods for reducing ammonia emissions from liquid effluents and for recovering the ammonia, U.S. Patent 9,005,333, U.S. Patent and Trademark Office.

[8] García-González M.C.; Vanotti M.B. (2015): Recovery of ammonia from swine manure using gas-permeable membranes: Effect of waste strength and pH, Waste Management 38, 455-461

[9] Szogi, A.A.; Vanotti, M.B.; and Rothrock, M.J. (2014): Gaseous ammonia removal system, U.S. Patent 8,906,332, U.S. Patent and Trademark Office.

[10] Vanotti, M.B.; Millner, P.D.; Sanchez Bascones, M.; Szogi, A.A.; Brigman, P.W.; Buabeng, F.; Timmons, J.; Hashem, F.M. (2015): Development of Pilot Modules for Recovering Gaseous Ammonia from Poultry Manure, Proc. From Waste to Worth: Advancing Sustainability in Animal Agriculture. http://www.extension.org/pages/72886/

TA-O_22 Soil changes and nutrient uptake induced by organic residues

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1. Objectives

The main effect of organic residues as fertilizer is its nutrient supply capacity; important nutrient release takes place during the decomposition process but lessens as the process advances to residue stabilization (Edwards et al, 2011; Porta et al, 2003; Baldock and Bruce, 2012). Other beneficial functions of organic matter in soils are their physical and chemical effects which can cause favorable conditions for plant development; desirable properties attributed to organic matter are its high CEC and its capacity to complex metal micronutrients keeping them available to plants. These are important reasons for using organic residues, not only because of its nutrient supply capacity but also as soil improvers.

Under the hypothesis that plants respond, not only or necessarily to nutrient supply, but to soil changes induced by compost addition, the objective of this paper was to study the soil changes and micronutrient uptake by plants related to composted organic residues.

2. Methodology

Residues were applied for evaluation, as a relatively stable compost prepared with grass from pruning and sheep manure in a 3:1 proportion. These composted residues (C) were mixed 1:1 with a clay soil (S) from an agricultural field near Colegio de Postgraduados in the Central Valley of México. One of these mixtures was evaluated as a treatment (SC), two were respectively added with 20% (SC-20) and 30% (SC-30) agrolite, a low density mineral obtained from amorphous volcanic glass as a commercial product; a fourth treatment consisted in soil complemented with 20-20-20 (N-P-K) fertilizer (SF). They were all compared with soil alone, making a total of five treatments. Agrolite was added to avoid compaction and find out if this had an effect on soil and plant development; fertilizer was added based on a previous experience where micronutrient absorption was favored by its presence.

The species Ammi majus (bishop's weed, lady's lace) an herbaceous plant pertaining to the Apiacea (Umbelliferae) family, was used to measure nutrient uptake and dry matter production (DM); this is an annual species with ornamental potential although its current main reason for study is its content of coumarins and furanocoumarins, substances used in commercial products for skin disease treatment (Ojala et al, 2000; Deepshikha et al, 2002; Krolicka et al, 2001).The experiment was conducted under greenhouse conditions in a completely randomized design with five replicates.

Two months old plantlets were transplanted to pots containing the five treatments described; the whole plants were harvested three months after transplant, dried at 70 ºC, weighed and ground for Mn, Fe, Cu, and Zn analysis. Micronutrients were extracted using a 3:1 sulfuric-perchloric acid mixture and measured by ICP atomic emission. The substrates were analyzed for electric conductivity (EC) and pH in a 1:2.5 suspension with distilled water; organic matter (OM) by wet digestion (Walkley-Black); and cation exchange capacity (CEC) using 1 N ammonium acetate adjusted to pH 7. These variables are being studied because in previous experiences with organic residues, changes were found (Ruíz-Bello et al, 2012).

3. Results and discussion

3.1 Soil changes and DM production

Dry weight production varied from 11.17 to 17.29 g (Table 1), the lowest was obtained in soil alone (S) which was statistically equal to that obtained with soil-compost treatment (SC); this result suggests that there is no direct effect of composted residues on *Ammi majus* DM yield. Two reasons may explain this result, one is that *A. majus* is a species that grows well in poor soil conditions suggesting that it does not have high nutrient requirements; second, the fact that the compost used was about one year old with a low C/N ratio (13.90) and it had low content of available nutrients (not shown). Soil stable organic matter (humus) is attributed a C/N relationship of 11 to 12 (Porta et al, 2003; Baldock and Broos, 2012).

Table 1: Dry matter production, soil and substrate (mixes) characteristics.

Treatment	DM (g)	pH	OM (%)	EC (mS)	CEC (cmol.kg^{-1})
1 S	11.35	7.95	3.54	5.19	45.31
2 SC	11.17	8.05	5.66	2.34	45.02
3 SC-20	17.29	8.20	6.65	1.23	39.67
4 SC-30	13.09	8.16	3.32	1.21	40.34
5 SF	12.49	7.69	5.12	1.30	33.05

S: soil; C: compost; 20:20% agrolite; 30: 30% agrolite; F: fertilizer.

On the other hand, there were significant changes on pH and EC related to residues application (Table 2) which seems to have affected DM yield. The highest DM production was obtained in soils added with compost plus 20 and 30% agrolite (SC-20 and SC-30) where pH increased and EC decreased significantly (α=0.05) compared to soil alone. Therefore, DM increase can be attributed to changes occurred in the soil and not to compost *per se*.

CEC values were high as expected in a clay soil (40% clay) but they were not affected by residues since there were no significant differences between soil alone (S) and those treatments with residues added. There was however, a significantly lower CEC (α=0.05) in treatment added with fertilizer (SF) which could be explained by blocking of exchange sites but it is not clear at this point.

Table 2: Mean differences of soil characteristics among treatments (α=0.05).

Treat.	Group	Treat.	Group	Treat.	Group	Treat.	Group
* (DM)		(pH)		(EC)		(CEC)	
3 (17.29)	A	3 (8.20)	A	1 (5.19)	A	1 (45.31)	A
4 (13.09)	A	4 (8.16)	A	2 (2.34)	AB	2 (45.02)	A
5 (12.49)	AB	2 (8.05)	AB	5 (1.30)	B	4 (40.34)	AB
1 (11.34)	B	1 (7.95)	B	3 (1.23)	B	3 (39.67)	AB
2 (11.17)	B	5 (7.69)	C	4 (1.21)	B	5 (33.04)	B

Means with the same letter are not significantly different.
*Numbers in parenthesis are the values of the corresponding parameter.

3.2. Soil changes and nutrient uptake.

Fe, Cu, Mn, and Zn critical content in *Ammi majus* were not available, but there were not visible signs of deficiency, suggesting that substrates supplied enough of them (Table 3); they were studied because of their possible complexation with OM which could favor availability. However, absorption of no one of them was affected by compost and no relationship between them and DM production was observed

No effect of residues was observed in OM content as measured by wet digestion, which is based on easily oxidized carbon and does not include stable carbon as expected to exist in a low C/N ratio compost.

Table 3: Micronutrient concentration in *Ammi majus* (bishop's weed, lady's lace).

Treat.	Fe	Cu	Zn	Mn	(pH)
			mg.kg^{-1}		
1 S	118.83	11.49	7.40	14.95	(7.95)
2 SC	179.96	12.10	7.42	15.71	(8.05)
3 SC-20	150.42	11.58	9.02	13.22	(8.20)
4 SC-30	183.15	10.76	7.14	11.58	(8.16)
5 S-F	208.56	11.22	8.75	23.04	(7.69)

S: soil; C: compost; 20: 20% agrolite; 30: 30% agrolite; F: fertilizer.

Mn concentration in plant however, was significantly higher (α=0.05) when fertilizer was added (Figure 1) which also decreased pH significantly with respect to the other treatments (Table 2). This shows, as in previous experiences, that Mn uptake is stimulated by other nutrients application, most probably due to the low pH induced by them as it seems to have been in this case.
As observed, not all the elements behave in the same way, while there were not any significant differences among them attributable to compost, still Mn absorption seems to be stimulated by compost as observed in treatment SC, and its absorption was strongly increased by N-P-K. Similar results have been obtained with this element and Cu in chrysanthemum, whose absorption was positively related to P and compost application, regardless of the nutrient content in the residues. Agrolite, although not significantly compared to soil alone, had a negative effect on micronutrients uptake.

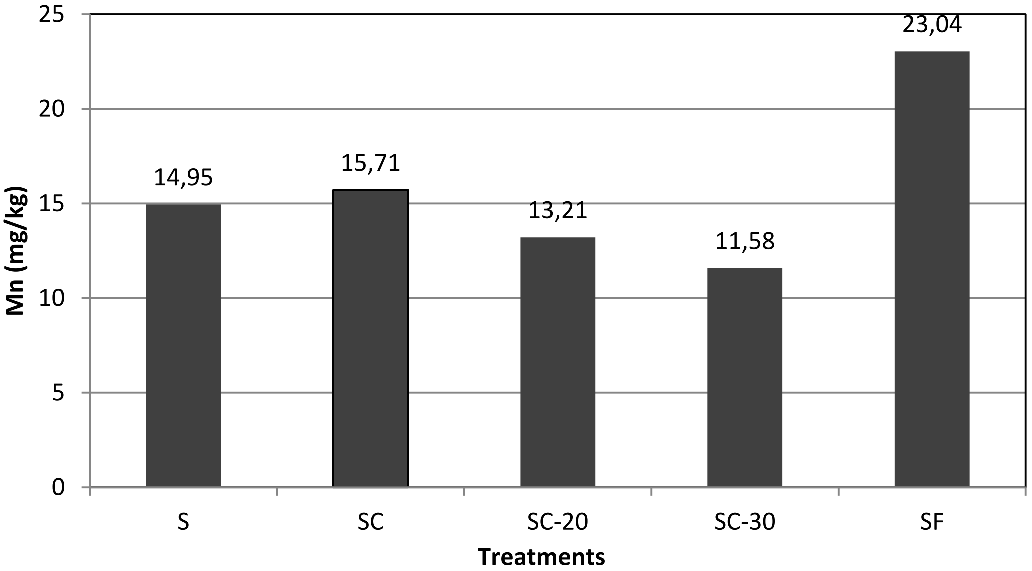

Figure 1. Mn Concentration in plant *(Ammi Majus)*

4. Conclusion and outlook.

The hypothesis that plants respond to soil changes induced by compost addition conveys the idea that compost produces changes in soil properties in the first place. In this investigation it was demonstrated that those changes do occur, as it was observed in pH increase and EC decrease when compost was added.

Research in the use of residues in agriculture has focused in their nutrient supply capacity and plant response to those nutrients. Little research is looking into what happens in the soils that could be the cause of plant response, though experience has shown they respond to both, nutrients and change. The first has been demonstrated by plant response to Ca when added as lime to change pH in acid soils; even when there is not any pH change, plants have responded positively to Ca which is scarce in acid soils. Plant response to changes is demonstrated by soil acidification to make soil P available instead of applying it. In this case, *Ammi majus* DM was increased by pH and EC changes induced by the composted residues.
Biodegradable organic residues can be regarded as a source of diseases, as related to toxic substances or pathogen agents; the risk can be diminished or eliminated through composting (Schwarz et al, 2009; Kuter et al, 1988). However, while decomposing organic residues may supply nutrients during the process of decay, when decomposed to a relatively stable condition (low C/N ratio) they may not supply an important amount of nutrient but still have desirable properties that are conveyed to the soils when applied to them. Composted residues used in this experiment were not high in nutrient content and did not increase OM most probably because of the short duration of the experiment; however there was an overall positive effect on plant response. No one of the micronutrient absorption was affected by residues.
Plant response also depends on the species; a wide variety of ornamental plants have been investigated and their response to different kind of residues with positive results (Ahmed et al, 2011; Larcher et al, 2011; García-Albarado et al, 2010; Gallardo y Valenzuela, 2003); although, there

are some species, mainly fruit and grain producing, where compost does not provide all the necessary nutrients (Muñiz-Rodríguez, 1998; Papafotiou et al, 2004) or even has negative effect (Erdogan et al, 2011). *Ammi majus* which is an ornamental and medicinal herb responded to composted residues.

Overall, the hypothesis in this research was proved since plant responded to changes in soil induced by residues, namely pH and EC. The objective also was accomplished since changes were attained and their effect could be observed.

Acknowledgements

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References

[1] Ahmed, Y. M.; Shalaby, E. A. and Shanan, N. T. (2011): The use of organic and inorganic cultures in improving vegetative growth, yield characters and antioxidant activity of roselle plants (*Hibiscus sabdariffa*). African Journal of Biotechnology 11(19), 1988-1996.

[2] Baldock, J. A. and Broos, K. 2012. Soil Organic Matter. IN: Handbook of Soil Sciences: properties and processes. Second Ed. CRC Press.

[3] Edwards, C. A.; Subler, S. and Arancon, N. 2011. Quality Criteria for Vernicomposts, IN: Vermiculture Technology: earthworms, organic wastes, and environmental management. CRC Press.

[4] Erdogan, R.; Zaimoglu, Z.; Budak, F.andKoseoglu, C. 2011. Use of sewage sludge in growth media for ornamental plants and its effects on growth and heavy metal accumulation. Journal of Food Agriculture & Environment.2011. 2(9): 632-635. Part 2.

[5] Gallardo C. y O Valenzuela.2003. Influencia de propiedades físicas y químicas de diferentes sustratos sobre la emergencia de especies ornamentales herbáceas anuales. Facultad de Ciencias Agropecuarias – UNER. Revista Científica Agropecuaria. 7(1): 69-74

[6] García-Albarado, J. C.; Trejo-Téllez; L. I.; Velázquez-Hernández, M. A. Ruíz-Bello, A.; Gómez-Merino, F. C. 2010. Growth of petunia in response to different proportions of compost in substrate. Revista Chapingo Serie Horticultura Volume: 16 Issue: 2 pages 107-113.

[7] Krolicka, A.; Staniszewska, I.; Bielawski, K.; Malinski, E.; Szafranek, J. and Lojkowska, E. 2001. Establishment of hairy root cultures of *Ammi majus*. Plant Science. 160: 259–264

[8] Kuter, G. A.; Hoitink, H. A. J. and Chen, W. 1988. Effects of municipal sludge compost curing time on suppression on *Pythium* and *Rhizotocnia* diseases on ornamental plants. Plant Dissease. 72: 751-756

[9] Deepshikha, P.; Madhumati, P. and Srivastava, P.S. (2002): Variation in xanthotoxin content in *Ammimajus* L. cultures during in vitro flowering and fruiting. PlantScience 162, 583-587.

[10] Larcher; F.; Berruti, A.; Gullino, P. and Scariot, V. 2011.Reducing peat and growth regulator input in *camellia* pot cultivation.Horticultural Science. 1(38): 35-42.

[11] Muñiz-Rodríguez, E. 1998. Utilización de biosólidos industriales en la producción de dos especies de plantas ornamentales. Tesis de Maestría en Ciencias en Suelos. Bell & Howell Information and Leaming Company. 300 North Zeeb Road. P.O. Box 1346. Ann Arbor, MI, 48106-1346. Director: Oscar Martínez Torres. Universidad de Puerto Rico, Mayaguez, Puerto Rico.

[12] Ojala, T.; Remes, S.; Haansuu, P.; Vuorela, H.; Hiltunen, R. and Haahtela, K. (2000). Antimicrobial activity of some coumarin containing herbal plants growing in Finland. Ethnopharm. 73: 299–305.

[13] Papafotiou M.; Physhaloua M.; Kargasb G.; Chatzipavlidisc I. and Chronopoulosa J. 2004. Olive-mill wastes compost as growing medium component for the production of poinsettia. Scientia Horticulturae. 2(102): 167–175.

[14] Porta, J.; López-Acevedo, M. and Roquero, C. 2003. Edafología para la agricultura y el medio ambiente, Cap. 8: Componentes orgánicos del suelo.

[15] Ruíz-Bello, A.; Trejo-Téllez, L.; Rodríguez-Mendoza, M. N. y Martínez-Villegas, E. 2012. Parámetros de madurez en diversos materiales composteados. EN: Tópicos edafológicos de actualidad. Universidad Autónoma de Zacatecas y Sociedad Mexicana de la Ciencia del Suelo.

[16] Schwarz, E.; Seoane, S.; Núñez, A. y López, M. M. E. (2009): caracterización y evaluación de compost utilizando como sustrato para planta ornamental. Compost Science & Utilization 4(17), 210-219.

TA-O_23 Multi-criteria indices to evaluate the effects of repeated organic amendment applications on soil quality

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1. Objectives

The recycling on soil of organic waste products (OWP) is known to improve soil properties like soil C storage, soil aggregate stability, nutrient availabilities for crops or soil functional biodiversity and biological activities. It may also lead to soil contamination. All effects occurred simultaneously and must be considered in the evaluation of the practice. This study aims at deciphering the long-term positive and negative impacts of repeated applications on soil physical, biological and chemical quality.

2. Methodology

A long term experiment has been initiated in 1998 (Feucherolles, France) [1]. The field is cropped with maize-wheat succession and 4 different OWP are applied in 4 replicates, every 2 years, a municipal solid waste compost, MSW; a biowaste compost, BIO; a co-compost of sewage sludge and green waste, GWS and a farmyard manure, FYM (Table 1). It includes also control plots with mineral N fertilization (CONT+N) and no OWP application.

Table 1: Mean physical and chemical characteristics of OWP applied in Qualiagro between 1998 and 2011. Co-compost of sewage sludge and green waste (GWS), a biowaste compost (BIO), a municipal solid waste compost (MSW) and a farmyard manure (FYM). I_{ROC} is an indicator of organic matter stability [2]. Same letter indicate no significant difference between treatments (Kruskal-Wallis test).

		GWS	BIO	MSW	FYM	*p value*
Dry matter	%	63.3 ± 8.2 a	70.1 ±8.5 a	67.8 ± 12.0 a	39.6 ± 9.1 b	*<0.0001*
Quantity applied	t $DM.ha^{-1}$	16.4 ± 2.7	19.1 ± 4.2	12.0 ± 3.2	13.2 ± 2.0	-
orgC	$g.kg^{-1}$ DM	265 ± 44 b	208 ± 47 c	308 ± 45 a	320 ± 67 a	*<0.0001*
Tot N	$g.kg^{-1}$ DM	23.5 ± 2.7 a	17.4 ± 4.5 b	17.6 ± 2.0 b	21.9 ± 3.1 a	*<0.0001*
orgC/totN	-	11.4 ± 2.1 b	12.1 ± 8.5 b	17.8 ± 4.0 c	14.7 ± 2.8 a	*<0.0001*
pH	-	7.5 ± 0.6 c	8.1 ± 0.5 b	7.5 ± 0.5 bc	9.1 ± 0.3 b	*<0.0001*
Tot P (P_2O_5)	$g.kg^{-1}$ DM	29.7 ± 7.8 a	11.0 ± 3.9 bc	8.0 ±1.7 c	12.6 ± 2.4 b	*<0.0001*
$CaCO_3$	$g.kg^{-1}$ DM	25.8 ± 10.4 c	99.0 ± 55.7 a	69.8 ± 22.7 ab	47.1 ± 16.6 b	*<0.0001*
Exchangeable Ca	$g.kg^{-1}$ DM	39.2 ±12.7 b	61.2 ± 19.6 a	60.3 ± 7.1 a	28.8 ± 8.4 c	*<0.0001*
Exchangeable K	$g.kg^{-1}$ DM	14.5 ± 4.5 c	21.0 ± 2.9 b	10.4 ± 3.0 c	35.6 ± 2.5 a	*<0.0001*
Exchangeable Mg	$g.kg^{-1}$ DM	4.9 ± 0.6 bc	4.7 ± 0.6 c	8.4 ± 4.4 a	5.9 ± 1.0 b	*<0.001*
I_{ROC}	% OM	77 ± 9 a	75 ± 6 a	49 ± 13 c	67 ± 7 b	*<0.001*

DM-dry matter; orgC-organic carbon; totN-Nitrogen total; tot P-Phosphorus total

- Measured parameters on soils and crops in the experiment between 1998 and 2011 have been classified into various classes soil fertility (SF), soil biological activities (SBA), soil biodiversity (SBD), soil physical properties (SPP), soil sanitary status available (SSA), soil sanitary status total (SST) and crop productivity and quality (CP)
- In each class, non-discriminating or correlated above r>0.8 parameters have been removed. The remaining parameters are considered as indicators. Finally, 9 indicators remained in SF (CEC, pH…), 6 in SBA (Urease activity, basal respiration…), 6 in SBD (Earthworm biomass and fungal diversity…), 3 in SPP (Aggregate stability, bulk density), 4 in SSA (Exchangeable Ni, Pb…) and 4 in SST (Total Cu, Mo…) and 7 in CP (Yield, protein content in grains…). Principal Component Analysis (PCA) was applied with the remaining parameters for each class
- Indicators responses curves have been defined ("more is better", "less is better" or "optimum") and soil quality index (SQI) has been calculated for each class using a weighting scheme [3]

3. Results and discussion

3.1 Effects of OWP on categories of impacts

Crop productivity and soil fertility: For crop productivity, crop yield, calcium and copper in grains were the main driving variables (Figure 1a). There was no significant difference between organic treatments. OWP were clearly separated from the CONT+N which has higher proteins content in grains. Thus it is very important to consider both crop yield and quality in the evaluation of organic treatment on crop results. Indicators of soil fertility were relevant to discriminate OWP (Figure 1b) and organic treatments were clearly separated from the controls. All organic treatments except MSW were characterized by larger total N contents in soil, the FYM and BIO treatments by higher pH and ratio K_2O/Mg and GWS treatment by larger soluble boron content. The OWP were differentiated by their efficiency at increasing soil organic matter (visible through total N in soil, organic matter content being removed because of the correlation with total N in this case) with larger effects of the FYM, BIO and GWS treatments compared to MSW. Additionally, FYM and BIO increased more soil pH in this loamy soil susceptible to soil acidification without liming. On the other hand, GWS increased the availability of P in soil and with this organic treatment both N and P could be used in crop fertilization as substitute for mineral fertilizers.

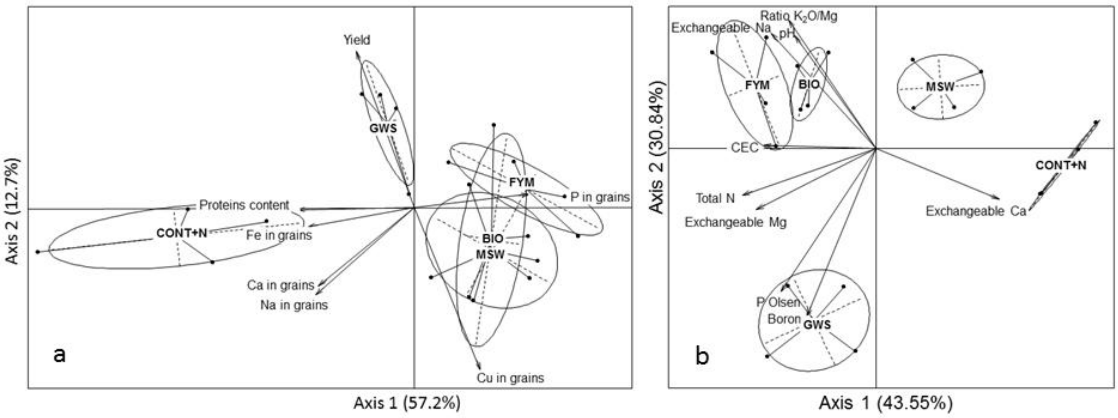

Figure 1: PCA with indicators included in the crop productivity index (a) and in the soil fertility index (b).

Soil sanitary status available and total: OWP were placed in the opposite of CONT+N (Figure 2a, b). For the soil sanitary status “available” (Figure 2a), the control treatment was characterized by larger concentrations in exchangeable trace elements than the organic treatments among which the BIO treatment presented the lowest concentrations. Regarding the soil sanitary status total (Figure 2b), the GWS treatment was separated from the other organic treatments with larger Cu concentration in soils. The organic treatments increased total trace elements in soil compared to mineral fertilizer, but decreased the available fractions of trace elements. In all treatments after 8 OWP applications, trace element concentrations remained within average ranges measured in similar soils.

Soil biological activities and biodiversity: The organic treatments were clearly separated from the control treatment and from each other treatments (Figure 3a,b). As regards soil biodiversity, the FYM was separated from the others because of larger earthworm biomass. Concerning soil biological activities two groups of organic treatments could be differentiated: MSW and BIO (with higher Arylamidase activities) versus GWS (β-glucosidase and phosphatase activities) and FYM (laccase and urease activities), all organic treatments being separated from the control treatment (low enzymatic activities). OWP enhance soil biodiversity and soil biological activities compare to mineral fertilizer, with intensities function of their OM quality. The addition of C substrates stimulate the growth of soil organisms (bacteria, fungi, free-living nematodes and earthworms included in the biodiversity category) which activities should further improved nutrients availabilities for plants and soil porosity.

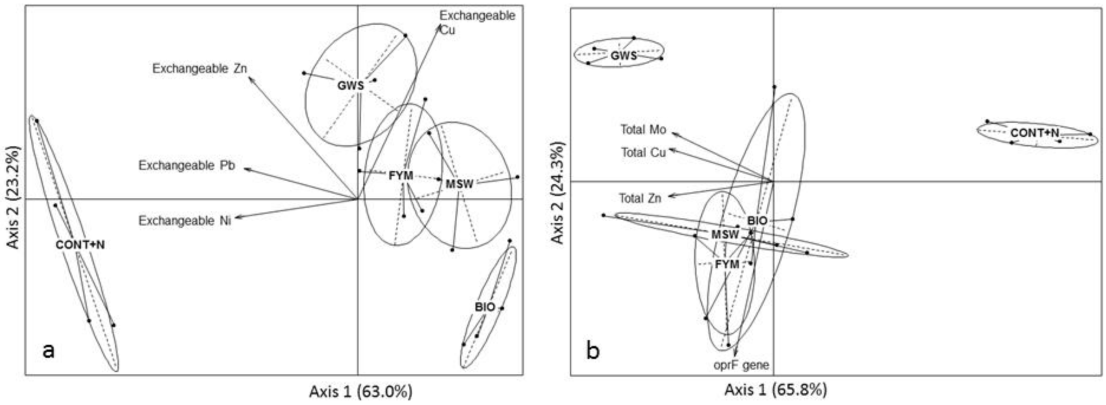

Figure 2: PCA with indicators included in the soil sanitary status available (a) and total (b).

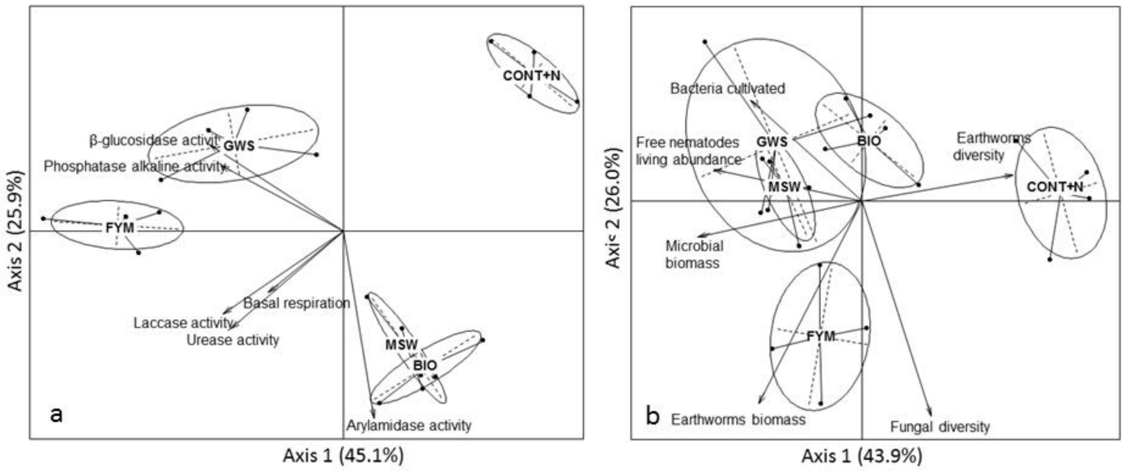

Figure 3: PCA with indicators included in the soil biological activity (a) and in soil biodiversity (b).

Soil physical properties: No differences between organic treatments appeared (detailed results not shown). OWP improve aggregate stability by increasing OM stocks in soil and because they stimulated the soil biological activities and soil biodiversity. Interactions existed between categories: for example, the stimulation of earthworms created soils macro pores by bioturbation and changed the water retention.

3.2 Effects of the OWP nature

The main driver of differentiation between OWP was their efficiency at increasing soil organic matter in relation with their organic matter stability (Table 1), then their input in nutrients (mainly N, P, K and Mg) but also their impact on soil pH that indirectly drives the mobility of trace elements. Thus the OWP that increased more soil organic matter improved more soil fertility (Figure 1). However even if organic treatments allowed reaching similar yields than with mineral fertilizers, the release of nitrogen into soil is different and is easier to control with mineral fertilizers. Indeed with mineral fertilizer, all N is available at the same time while N availability is progressive with OWP and laying on OWP and soil organic matter mineralization. Thus the release of mineral N may be out of steps with the needs of the crops, explaining why the proteins content was higher with mineral fertilizer than with OWP. On the other hand, the most biodegradable OWP further improved soil biodiversity such as MSW and FYM (Figure 3b). Some OWP were specifically characterized by larger P and B contents (GWS), K content (FYM), higher pH (BIO and FYM), thus may be used to enhance soil fertility with these specific objectives. There is potentially a risk of unbalance of nutrients or excess salinity increasing soil conductivity. An optimum value proposed for the ratio K/Mg is 1.375, equivalent to 1.66 K_2O/Mg, and was exceeded in the FYM treatment meaning excess of K and lack of Mg could occur for plants (Figure 1b). Finally, all OWP maintained soil pH around 7. Some treatments even increased pH and nearly reached the optimum pH value (7.5) for loamy soil (BIO and FYM) when soil pH tended to decrease with mineral fertilizer.

4. Conclusion and outlook

SQI calculated are represented in a radar (Figure 4). All OWP treatments were clearly differentiated from the control treatment receiving mineral fertilizer except for the crop productivity indice. Thus the main driver of differentiation was the increase in soil organic matter more or less important with the different OWP.

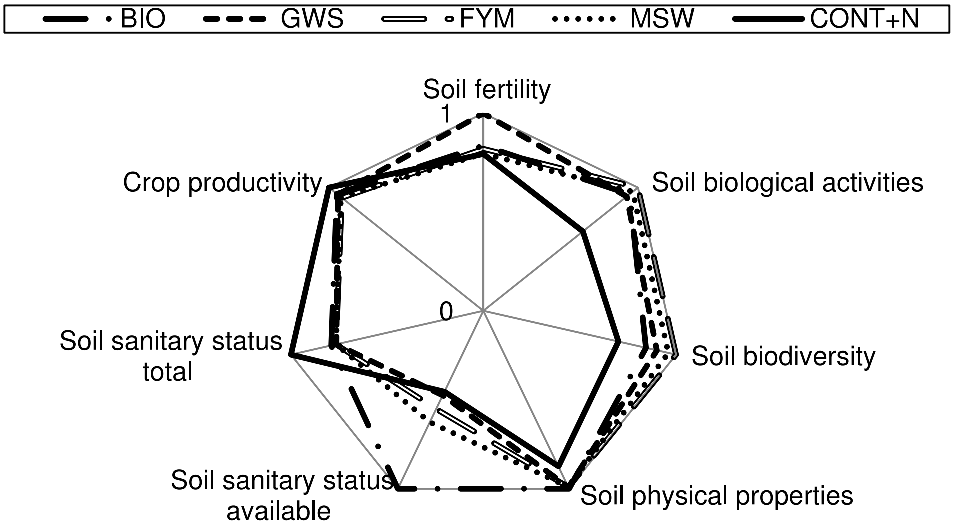

Figure 4: Radar representation of the soil quality indices. All indices were normalized (provided in proportion to the maximum).

The OM quality drove the responses of OWP application for each category of impacts. The organic treatments could also be differentiated from each other for some categories: soil fertility, soil biological activities and soil biodiversity. Compare to mineral fertilization, OWP application improved soil physical and soil biological quality and had the same crop productivity. OWP could then provide a better resilience to soils.

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References

[1] Houot, S. ; Clergeot, D. ; Michelin, J. ; Francou, C. ; Bourgeois, S. ; Caria, G. ; Ciesielski, H. (2002). Agronomic value and environmental impacts of urban composts used in agriculture. In: Insam, et al. (Eds.), Microbiology of Composting. Springer Verlag, Berlin, pp. 457–472.

[2] Lashermes, G. ; Nicolardot, B. ; Parnaudeau, V. ; Thuriès, L. ; Chaussod, R. ; Guillotin, M.L. ; Linères, M. ; Mary, B. ; Metzger, L. ; Morvan, T.; Tricaud, A. ; Villette, C.; Houot, S. (2009) Indicator of potential residual carbon in soils after exogenous organic matter application. Europ J Soil Sci 60: 297-310.

[3] Velasquez, E. ; Lavelle, P. ; Andrade, M. (2007) : GISQ, a multifunctional indicator of soil quality. Soil Biol. Biochem. 39 : 3066-3080

TA-O_24 Effect of manure-based phosphorus fertilizer and biochar on biomass yield of spring barley and faba bean in comparison to conventional fertilizer

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1. Objectives

This study aimed to evaluate the fertilizing performance of an innovative phosphorus fertilizer recovered from pig manure (referred to as “P-Salt”) on biomass yield and quality of summer barley and faba bean in comparison to conventional fertilizer. A further objective was to investigate whether additional application of biochar, which is considered a soil amendment, results in increased biomass yield.

2. Methodology

The P-Salt contains the nutrients precipitated from the liquid fraction of separated pig manure. It is basically a complex of struvite and calcium phosphate. Biochar is produced by pyrolysis of the separated solid fraction. Both products were tested in a greenhouse experiment with spring barley and faba bean on two soils, loess and sand. These soils were chosen due to their particularly low contents of nutrients and organic matter. In addition to phosphorus (P), P-Salt also contains nitrogen (N). Conventional synthetic fertilizers (ammonium nitrate and calcium dihydrogen phosphate) were applied in an according nutrient ratio as a reference.

All fertilizer treatments – manure-based and synthetic – were applied in three levels:
- deficient nutrient supply (50%),
- optimal nutrient supply (100%),
- oversupply (200%).

The 100% treatment hereby corresponded to a total N and P concentration of 240 and 150 mg per kg soil, respectively. The other application rates were calculated accordingly. Table 1 provides an overview of the performed treatments.

In this study, biochar was considered a soil improver and was applied in two concentrations (0,1% and 0,2% w/w) in combination with the 100% level of P-Salt. Untreated controls were included. The experiment was established with four replications. Pots were watered from the top with deionized water. Fresh and dry matter yield (DMY) were determined after six weeks. P concentration in plant samples was determined using microwave digestion followed by ICP spectrometer measurement. Soil samples taken at harvest day were analysed for plant-available P by CAL extraction.

Table 1: Overview of all treatments performed. (Results from treatments in parentheses not covered in this article)

Treatment	Fertilizer			Biochar
	50%	100%	200%	
P-Salt	x	x	X	
P-Salt + biochar		x		0.1%
		x		0.2%
Synthetic N + P	(x)	x	(x)	

Statistical analysis of the data was done using SAS software version 9.3 (SAS Institute Inc., Cary, NC, USA). Analyses of variance were performed using the SAS procedure PROC MIXED. Mean values were compared at $P \leq 0.05$.

3. Results and discussion

The influences of the factors 'Soil', 'Treatment' and their interaction 'Soil*Treatment' were highly significant (P < 0,0001) in both crops tested.

3.1 Fertilizing effect of P-Salt

P-Salt application resulted in significantly higher DMY of barley compared to the control in both soils (Figure 1A). The 200% dose increased DMY further than the optimal dose in sand only. The P concentration in treated plants was higher than in the untreated ones, yet it hardly varied between the different doses in clay. In sand, it even decreased with increasing P-Salt dose: the P concentration of barley fertilized with the 200% dose was significantly lower compared to the other doses. This probably indicates that barley plants were saturated with P.

Fertilization with 100% and 200% doses of P-Salt increased DMY of faba bean in both soils (Figure 1B). The 50% dose did not significantly affect DMY. The 200% dose led to a significant DMY increase compared to the 100% dose in clay only. The relatively high DMY of untreated bean may be explained by its high thousand-seed weight which provides more resources as opposed to barley. This is particularly advantageous during early crop growth and when the available growth period is limited, as in this experiment. The P concentration in bean increased evenly with the P-Salt dose.

The P concentration of both crops was generally higher in plants grown in sand compared to those grown in clay. This is certainly a consequence of the strong potential of clay to immobilize nutrients. A clear difference was noticed in the P concentration between barley and bean regardless of the soil type. P concentration in bean was directly proportional to the fertilizer dose, whereas this was not the case in barley. This can be explained by the bean's specific ability (as a legume) to make immobilized P readily available for uptake by releasing root exudates. Unlike legumes, cereals do not have this mechanism.

P-Salt exhibited a stronger fertilizing effect in barley compared to bean. This becomes obvious if the DMY of the control is compared with that of the pots treated with the 50% dose: the 50% dose did not increase DMY of bean, whereas its yield effect was significant in barley. This might also be a result of the bean's high thousand-seed weight. Furthermore, both crops were treated with the same fertilizer amounts. Bean generated more biomass than barley which is usually accompanied by a higher nutrient demand. Results might be different if bean had received more nutrients.

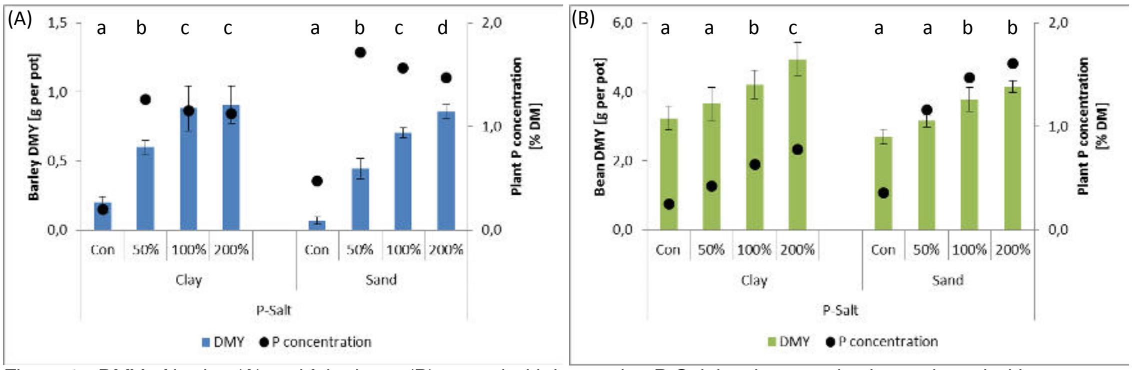

Figure 1: DMY of barley (A) and faba bean (B) treated with increasing P-Salt levels grown in clay and sand with corresponding P concentration in biomass. DMY values indicated with the same letter are not significantly different.

3.2 Effect of P-Salt and biochar in comparison to synthetic fertilizer

Barley

In clay, the combined application of 0,1% biochar and 100% P-Salt slightly increased DMY of barley compared to P-Salt only; however, this effect was not statistically significant (Figure 2A). A further increase in biochar concentration to 0,2% seemed to reverse this tendency again. In sand, both biochar concentrations resulted in a significant yield effect compared to P-Salt only. DMY in pots containing 0,1% biochar was even a little, but not significantly, higher than in those with 0,2% biochar. This leads to the assumption that nutrients, particularly nitrogen, were possibly fixed by biochar and thus not available for crop uptake. This assumption can be clearly confirmed by the N concentration in plants and soil which has been determined as well (not shown in this paper).

All P-Salt treatments – with and without biochar – still resulted in higher DMY than the synthetic fertilizer in clay. This comparison was not possible for barley grown in sand because no biomass could be harvested in the respective pots. Fertilization with ammonium nitrate is supposed to be one reason why these plants withered and died soon after germination. Concentration of plant-available N was probably too high for the sensitive seedlings and, thus, toxic. This could have been avoided by splitting of the N dose.

In both soils, the P concentration in biomass remained on the same level for all P-Salt treatments including the combination with biochar. In contrast, synthetic fertilizer considerably increased the P concentration compared to the treatments with P-Salt. The reason is certainly the higher availability of synthetic P right from the beginning. The synthetically fertilized plants possibly took up all required nutrients in the first weeks of the experiment. In contrast, the crops fertilized with P-Salt – where nutrients are only gradually released - were not able to compensate for that during the remaining time. Longer test duration might have shown different outcome.

Faba bean

Figure 2B compares DMY of bean fertilized with P-Salt, P-Salt and biochar in combination, and synthetic fertilizer. DMY of bean increased significantly following fertilization with P-Salt only and in combination with biochar compared to the control in both soils (Figure 2B). The biochar concentration did not have any influence.

The combined application of P-Salt and biochar showed a tendency to slightly higher yields than P-Salt only; however, this effect was not significant. In clay, DMY did not differ between the different treatments including the synthetic fertilizer. In sand, DMY results following the P-Salt and combined P-Salt/biochar treatments were comparable whereas DMY of control and synthetically fertilized pots both achieved a similar, yet lower level.

The P concentration followed the same pattern in both soils, with once again higher concentrations in plants grown in sand compared to those grown in clay. The highest P concentration was measured in plants treated with P-Salt only and P-Salt combined with 0,2% biochar. It was significantly lower in plants treated with P-Salt and 0,1% biochar and still lower in plants treated with synthetic fertilizer.

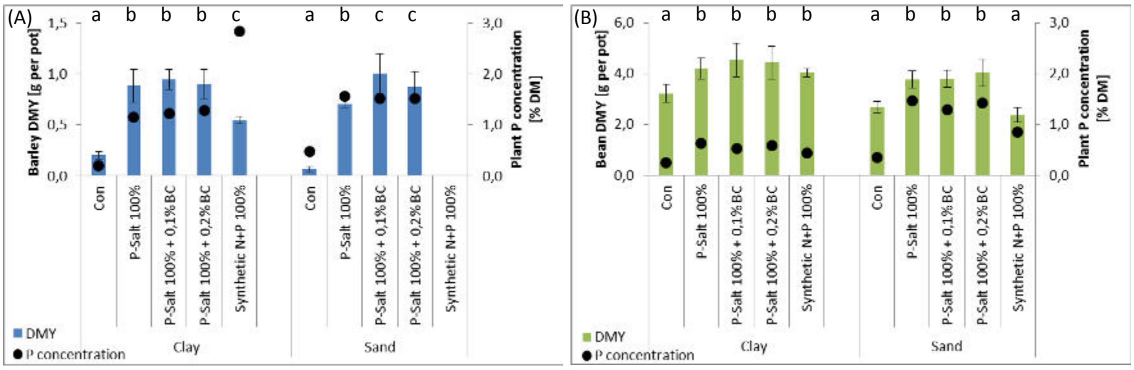

Figure 2: Comparison of DMY of barley (A) and faba bean (B) treated with P-Salt, biochar and synthetic fertilizer grown in clay and sand with corresponding P concentration in biomass. DMY values indicated with the same letter are not significantly different.

Biochar is frequently reported to immobilize nutrients – in particular nitrogen - and make them unavailable for plant uptake which can result in yield losses. This was found by [1] and many others. On the positive side, immobilization also prevents nutrients from leaching [2]. The addition of biochar to light or poor soils, e.g. sand, is known to positively influence their water retention capacity. This is confirmed by numerous publications as well as by own results. This effect was incidentally observed, yet not monitored in particular in this experiment. Sand pots with added biochar seemed to retain more water, whereas it ran faster through the pot into the saucer in sand pots without biochar. Plants in the latter pots sometimes showed more obvious symptoms of drought stress shortly before watering.

4. Conclusion and outlook

This study found the manure-based P-Salt to have a very good fertilizing effect. Fertilization with P-Salt increased barley and bean DMY significantly in both soils compared to the control. Moreover, it was able to keep up with synthetic fertilization in the tested crops and soils. The P-Salt treatments even resulted in higher DMY than the synthetic fertilizer in both crops and both soils except for bean grown in clay. Comparable results have been reported for struvite products before [3]. P concentration in the biomass was similar following all P-Salt treatments including the combination with biochar. However, P concentration was higher in plants treated with synthetic fertilizer than in plants treated with P-Salt in barley and vice versa in bean. The slow release of nutrients can prevent leaching and makes P-Salt a suitable fertilizer for light soils with high sand contents.
The combination of P-Salt and biochar resulted in significantly higher DMY than P-Salt only in barley grown in sand; otherwise this treatment showed only a trend to slightly higher yields. Thus, the influence of biochar needs to be further specified. The next step is testing of P-Salt in a field experiment with maize in summer 2015.
Generally, the recycling of nutrients from pig manure offers an attractive solution for dealing with the accumulation of manure which is particularly problematic in regions with high livestock densities and limited land availability. The high performance of this new fertilizer seen here is a promising precondition for continued research.

Acknowledgements

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References

[1] Alburquerque, J. A., Salazar, P., Barrón, V., Torrent, J., Carmen del Campillo, M., Gallardo, A., Villar, R. (2013): Enhanced wheat yield by biochar addition under different mineral fertilization levels. Agronomy for Sustainable Development 33, 475-484

[2] Atkinson, C. J., Fitzgerald, J. D., Hipps, N. A. (2010): Potential mechanisms for achieving agricultural benefits from biochar application to temperate soils: a review. Plant Soil 337, 1–18

[3] Liu, Y., Rahman, M., Kwag, J.-H., Kim, J.-H., Ra, C. (2011): Eco-friendly production of maize using struvite recovered from swine wastewater as sustainable fertilizer source. Asian-Australasian Journal of Animal Science 24 (12), 1699-1705

TA-O_25 Improving nitrogen fertilization effect from residues in spring and winter cereals

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1. Objectives

Apart from manure, there are other organic residues used as fertilizers in agriculture. The percentage of the nitrogen that is readily plant available varies widely between residues and is often not related to ammonium content as it is in most manures. In a recent study [1] with pot experiments as a reference, the carbon/nitrogen ratio was found to be a promising indicator for estimating the mineral fertilizer equivalent (MFE) of different residues. However, before this method can be adopted, it should be tested under field conditions. This is easily done in spring sown cereals, where all types of residues can be applied and incorporated before sowing. In winter sown cereals, other residue characteristics can be crucial for the effect, since the residues should be applied after crop emergence and soil incorporation can be difficult. For winter cereals, it is therefore not only a question of what MFE to expect, but also how and when it should be applied.
The three main objectives with this study were to:

- assess carbon/nitrogen-ratio (C/N-ratio) as a tool to predict MFE of different organic residues
- estimate second year nitrogen (N) effects of different residues
- test whether N effects on yield from different kinds of residues can be improved with incorporation or early application when applied to growing winter wheat

2. Methodology

2.1 Experiments in spring oats

A total of nine field experiments in spring oats were conducted at three locations (Halmstad, Lidköping and Uppsala) in Sweden during the years 2012-2014. The grain yields from fertilizing with eight different residues were compared with four levels of mineral N fertilizer (Table 1). The treatments were randomized into four blocks. All treatments were fertilized with P and K fertilizers according to local recommendations, to ensure that yield differences were due to N rather than P or K limitation. The second year crop was only moderately fertilized with N (30-40 kg N ha^{-1}), to study second year effects of the residues on yield and N offtake.

Table 1: Treatments in the field experiments in oats

Fertilizer/residue	Application rate (kg total N ha^{-1})
Ammonium nitrate	0
Ammonium nitrate	40
Ammonium nitrate	80
Ammonium nitrate	100
Pig slurry	100
Cattle slurry	100
Pelleted meat meal 1	80
Pelleted meat meal 2	80
Biogas residue 1	80
Biogas residue 2	80
Chicken manure	100
Vinasse	80

2.2 Experiments in winter wheat

A total of nine field experiments in winter wheat (Table 2) were conducted at three locations in Sweden (Halmstad, Lidköping and Västerås) during the years 2012-2014. The locations were chosen to represent the same climate conditions as for the oat experiments. Yield effects of fertilizing with three different residues with and without soil incorporation (treatments 6-11) were compared with four levels of mineral N fertilizer (treatments1-4). There was an additional mineral fertilizer N treatment with 60 kg N ha^{-1} with harrowing in growing crop in spring (treatment 5), to achieve similar crop damage as in treatments with residue incorporation. In some experiments there were also additional treatments with late autumn fertilization with chicken manure and pelleted meat meal (treatments 12-13). The three different residues were chosen to represent three common types of residues with different physical characteristics and therefore different potentials for being successively incorporated into soil.

Table 2: Treatments in the field experiments in winter wheat

Treatment	Fertilizer/residue	Residue incorporation	Time for application	Application rate (kg total N ha^{-1})
1	Ammonium nitrate			0
2	Ammonium nitrate		spring	60
3	Ammonium nitrate		spring	80
4	Ammonium nitrate		spring	120
5	Ammonium nitrate	spring harrowing	spring	60
6	Pelleted meat meal	no incorporation	spring	120
7	Pelleted meat meal	direct incorporation	spring	120
8	Biogas residue	no incorporation	spring	120
9	Biogas residue	shallow injection	spring	120
10	Chicken manure	no incorporation	spring	120
11	Chicken manure	harrowing	spring	120
12	Chicken manure		late autumn	120
13	Pelleted meat meal		late autumn	120

2.3 Data analysis

The grain yield in treatments with residues were compared to the yield response to mineral fertilizer N in order to calculate MFE, i.e. the fraction of total N in residues equally available to plants as inorganic N. For the oat experiments, MFE was plotted against C/N-ratio and the correlation was compared to that of the former pot experiment. The yields in residue amended treatments were compared to those with no and those with 40 kg N ha^{-1}, in order to estimate second year effects. The treatment with 40 kg N ha^{-1} was considered most relevant as reference without residue, since it provided similar amounts of mineral N to the crop as the residues in the first year.

3. Results and discussion

3.1 Mineral fertilizer equivalent in oat experiments

The oat experiments in Uppsala (2 experiments) showed very low N response and were thus excluded from the results. The MFE in the other trials (7 experiments) on was average 33% for cattle slurry, 43% for chicken manure, 53% for pig slurry, 62% for biogas residues, 68% for vinasse and 70% for pelleted meat meal. Chicken manure had similar MFE in winter wheat, whereas MFE of biogas residue and pelleted meat meal tended to be lower in winter wheat than in spring oats.

3.2 C/N-ratio to predict mineral fertilizer equivalent

The relation between C/N-ratio and MFE from compiled data from the oat experiments was very similar to the relation found the former pot experiment [1] and also to correlations found in other studies [2;3;4]. However, the correlation was weaker in the field (r^2=0.26) than in the pots (r^2=0.84) (Figure 1). Looking at one experiment at a time did improve the r^2-value to around 0.5, but also altered the regression coefficient. The regression line tended to be steeper in experiments with later sowing and shorter period for crop N uptake and more flat in experiments with early sowing.

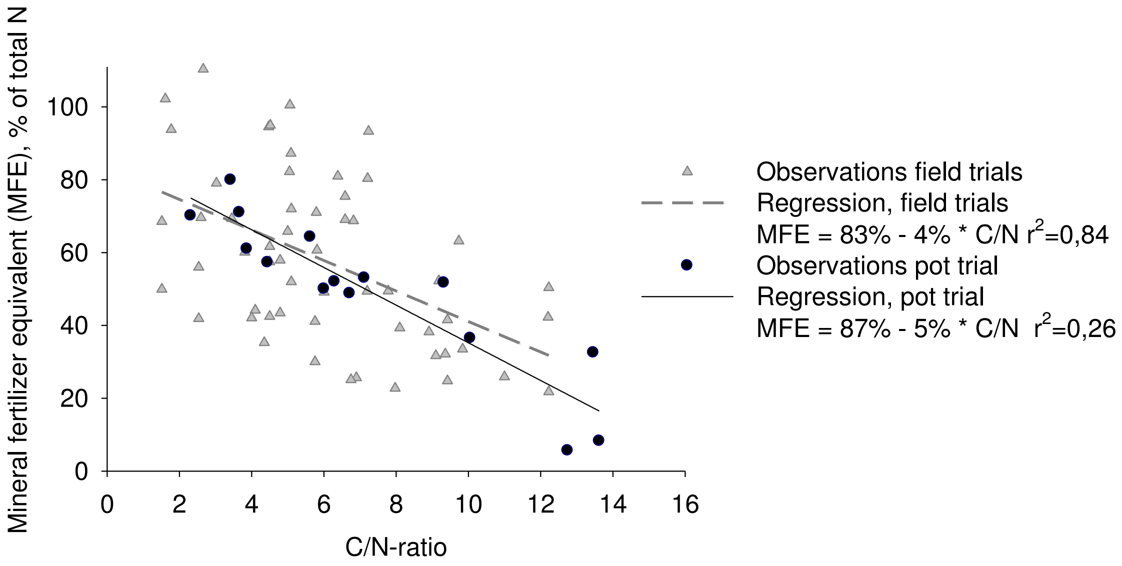

Figure 1: Average mineral fertilizer equivalent (MFE) for different categories of residues from trials in Halmstad (4n) and Lidköping (3 n).

3.4 Second year effects

The crop N offtake of the next crop (grown the year after application) increased by on average 3 kg N ha^{-1} compared to the treatment without N fertilization. That is about to 3% as much N as added with residues. However, compared to the treatments with mineral fertilizer, there was hardly any difference in N offtake.

3.5 Soil incorporation in growing winter wheat

Soil incorporation of residues to winter wheat led to significant yield increases in some experiments 2013 and 2014, but not in 2012 when rainfall was high. Incorporation of pelleted meat meal with a sowing machine gave increased yield by 940-1300 kg ha^{-1} in three different experiments, although the difference was only significant in one in which MFE increased from 32 to 63%. Direct incorporation of biogas residues increased yield by 940 kg ha^{-1} and MFE from 21 to 44% in one experiment on a clay soil, where incorporation of the other residues was not successful due to a hard clay surface and the different techniques used. Incorporation of chicken manure by harrowing increased yield by 740 kg ha^{-1}, and MFE from 30 to 42%, in one experiment, but it was not statistically significant.

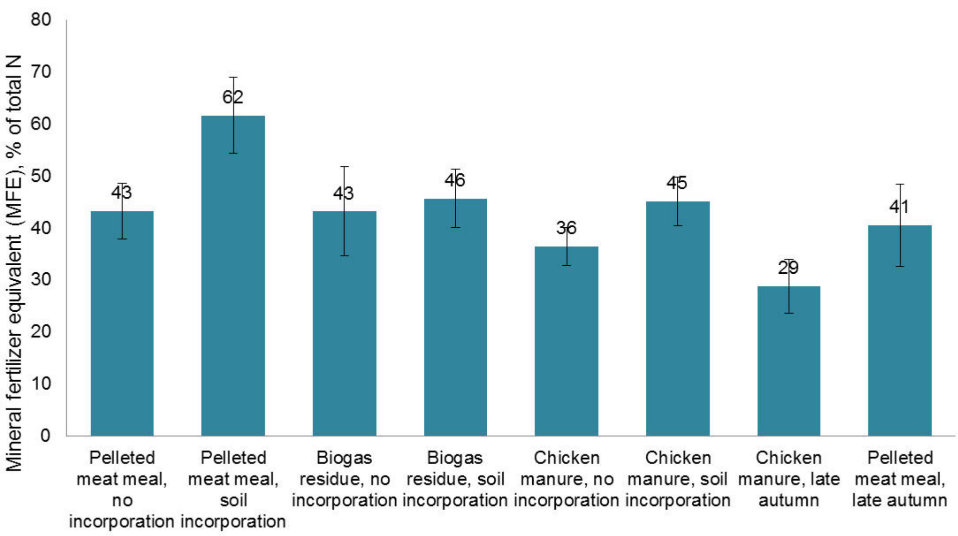

Figure 2: Average mineral fertilizer equivalent (MFE) for different residues applied to winter wheat, as affected by soil incorporation in spring and application in late autumn, in field trials at Halmstad (2n), Lidköping (2n) and Västerås (1n) 2013 and 2014.

3.6 Time of application

Application of chicken manure in late autumn/winter (February-November) reduced yield by 600 - 1550 kg ha^{-1} and MFE by 7-23% compared with spring application. Autumn application of meat meal pellets had no significant effect on yield compared to spring application without incorporation.

4. Conclusion and outlook

Mineral fertilizer equivalent of different residues are correlated to C/N-ratio also under field conditions. The relation from the field experiments is on average the same as in the pot experiments, but varies a bit depending on length of period for crop N uptake. The relation should be useful for fertilization guidelines. A residue with a certain C/N-ratio could be expected to on average for spring cereals have the corresponding MFE according to the relation presented. However, the variation between the field experiments suggests that this recommendation could be adjusted depending on how long time there is between field application and end of crop N uptake. The second year N effects of organic residues were small. Incorporation of residues in growing winter wheat can increase yields and MFE for all three types of residues, depending on conditions, but is most likely to be successful for meat meal pellets. The need for incorporation is greater under dry conditions, and the technology must be good enough to succeed with the incorporation on the soil in question. Autumn/winter application reduced yield and MFE, mainly for chicken manure, and cannot be recommended.

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References

[1] Delin, S.; Stenberg, B.; Nyberg, A.; Brohede, L. 2012. Potential methods for estimating nitrogen fertilizer value of organic residues. Soil Use and Management 28, 283–291.

[2] Gale, E.S., Sullivan, D.M., Cogger, C.G., Bary, A.I., Hemphill, D., Myhre, E.A. 2006. Estimating plant-available nitrogen releasefrom manures, composts, and specialty products. Journal of Environmental Quality, 35, 2321–2332.

[3] Sørensen, P. & Ferna´ ndez, J.A. 2003. Dietary effects on the composition of pig slurry and on the plant utilization of pig slurry nitrogen. Journal of Agricultural Science, 140, 343–355.

[4] Sørensen, P., Weisberg, M.R. & Lund, P. 2003. Dietary effects on the composition and plant utilization of nitrogen in dairy cattle manure. Journal of Agricultural Science, 141, 79–91.

TA-O_26 Recycled phosphorus fertilizers from urban residues tested in agricultural crop production

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1. Objectives

Global Phosphorus (P) resources are getting depleted and it is clear that recycling strategies of P will gain importance in the future. In the European Union, 11 million tons of sewage sludge dry mass is available every year and several techniques have been developed to recover P from waste water. A range of different recycled P-products was tested in pot and field experiments with *Trifolium pratense* L. and *Zea mays* L. for their P-availability in comparison to $Ca(H_2PO_4)_2$ and Phosphate Rock (PR).

2. Methodology

Two pot experiments were conducted in the greenhouse. The investigated recycled P-fertilizers were a struvite and a thermo-chemically treated sewage sludge ash (SSA), tested on clover. In a second experiment with maize, an additional struvite, differing in its production process, as well as a Calcium-Phosphate (Ca-P), slag-P, and coal, was tested. Control treatments in both experiments were unfertilized, fertilized with PR and $Ca(H_2PO_4)_2$. Fertilizers were applied to a target of 50 mg P kg^{-1} soil DM, other nutrients were provided sufficiently. The used substrate was a 2:1 mixture of a silty loam soil (CAL-P 2.2 mg P 100 g^{-1} soil DM, pH 7.2 ($CaCl_2$)), and silica sand. Pot size was 1.8 L. Clover was harvested four, six and eight month after sawing. Maize was harvested after eight weeks of cultivation. Biomass and P-content in plant tissue was determined.
A field experiment took place with maize at the University of Hohenheim, Stuttgart. Investigated recycled P-fertilizers were both struvites as in the pot experiments, Ca-P, coal and SSA. A bio waste compost, PR and unfertilized plots represented control treatments. Phosphorus was applied according to a target of 80 kg P ha^{-1}. 100 kg Nitrogen ha^{-1} was applied as hornmeal. The soil was a silty Loam with a CAL-P content of 3 mg P 100 g^{-1} DM and pH 6.6 ($CaCl_2$). Maize was harvested as silage maize, biomass and P-content in plant tissue was determined after wet digestion [1].

3. Results and discussion

3.1 Pot experiments

At first harvest date of clover, the struvite treatment reached a biomass and shoot P-uptake significantly higher than the SSA, the unfertilized and the PR treatment, and did not differ from the $Ca(H_2PO_4)_2$ treatment (Figure 1A). This supports the reported usually high availability of the P contained in struvite [2]. Shoot DM and P-uptake in the SSA treatment was significantly decreased and not different from the unfertilized control and the PR treatment, showing that the PR and the SSA were completely ineffective at given soil conditions. It has been reported [3] that the relative effectiveness of SSA, compared to a highly soluble P-fertilizer, decreases with increasing soil pH, which explains the results of the SSA treatment in this experiment, being in the same range as the unfertilized control. At second harvest date, struvite, SSA, unfertilized and PR did not differ anymore from each other in shoot DM (Figure 1B). Only in the $Ca(H_2PO_4)_2$ treatment, a significantly higher shoot DM could be obtained, compared to the PR treatment. There were no significant differences anymore in shoot DM between all treatments at third harvest date (data not shown). Shoot P-uptake of all treatments increased from the first to the third harvest, however, at third harvest date, differences between single treatments were levelled out and not anymore significant from each other (data not shown). Overall little differences between treatments, and the reduction in differences of DM and shoot P-uptake in clover towards the third harvest date indicates that clover was able to mobilize considerable amounts of P, even under the unfavourable neutral soil conditions. This assumption is supported by findings of the second pot experiment. Same recycled P-fertilizers led to distinct differences in DM and shoot P-uptake, as tested under identical conditions with maize (Figure 3 A, B). It can be assumed, that a proton release of symbiotically grown red clover roots led to a considerable P mobilization from soil and fertilizer [4].

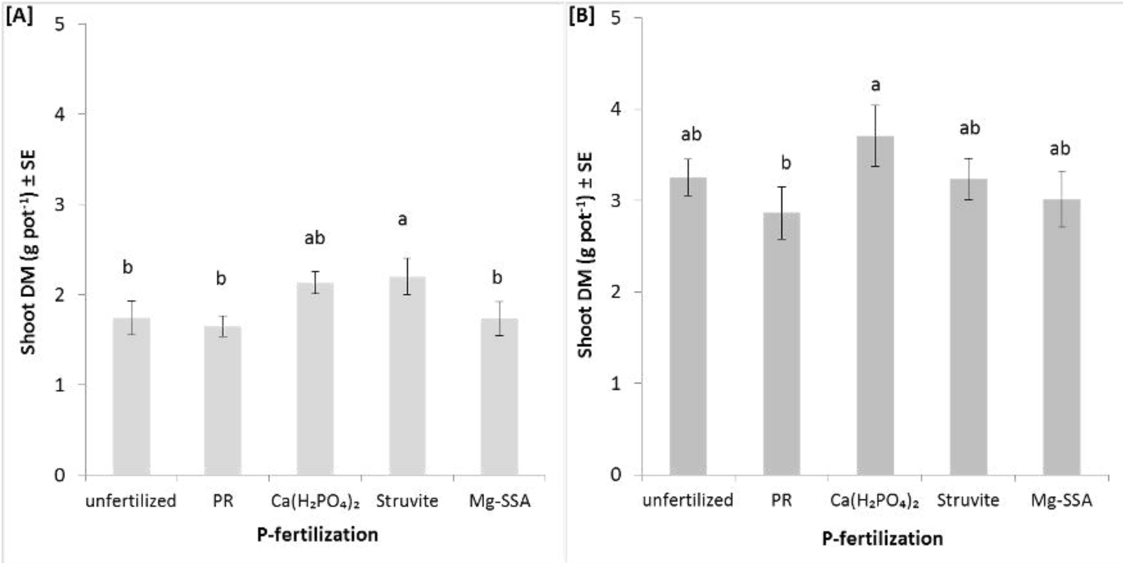

Figure 1: Mean shoot DM of *Trifolium pratense* per pot ± SE at first [A] and second [B] harvest date, in dependency of different P-fertilization. Different letters indicate significant differences ($p \leq 0.05$).

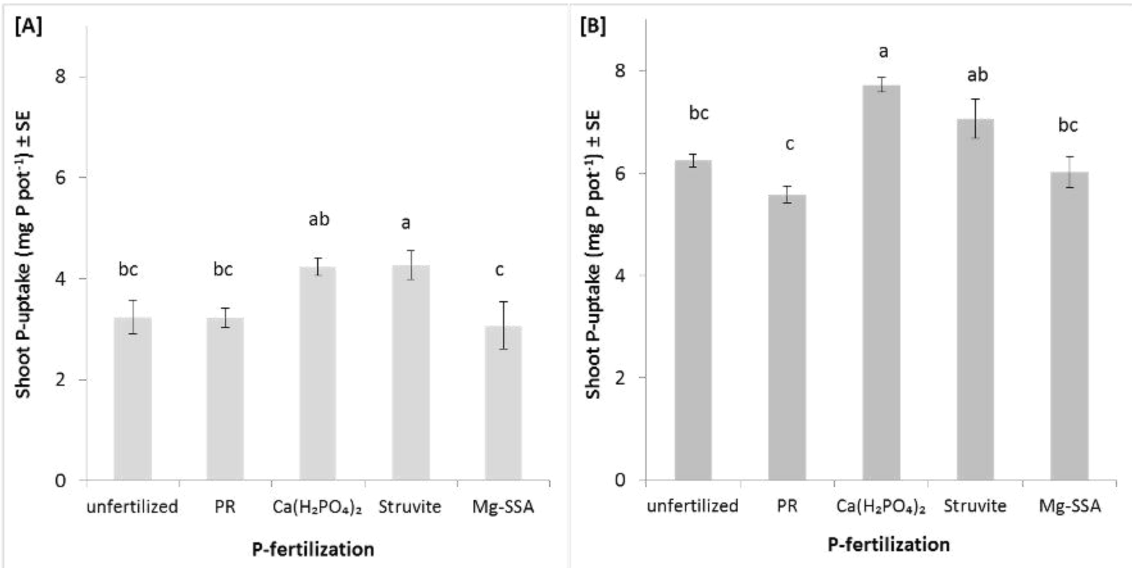

Figure 2: Mean shoot P-uptake of *Trifolium pratense* per pot ± SE at first [A] and second [B] harvest date, in dependency of different P-fertilization. Different letters indicate significant differences ($p \leq 0.05$).

In the experiment with maize, noticeable differences due to different recycled P-fertilizers could be shown in DM and shoot P-uptake. Generally, three treatment groups could be distinguished: The SSA and coal treatment performed in the same, lowest range of the unfertilized and PR treatment. Intermediate values of DM and P-uptake were measured in slag-P, Ca-P and one of the struvites, while a second struvite reached significantly higher values and was not different from the $Ca(H_2PO_4)_2$ treatment (Figure 3 A, B). Results matched investigations on maize and rye in pot experiments [5], showing struvites being as effective as highly soluble triple superphosphate fertilizers. However, by comparing both struvites in this experiment, it can be seen, that P-availability of struvites differs in dependency from their original material and production process. The Ca-P product is produced by crystallization from process water. In dependency of the P-load of the raw material, the end product can contain blends of tri-calcium phosphates, di-calcium phosphates and mono-calcium phosphates [6], which determine its P-availability. From these results, it is obvious that the Ca-P product contained a certain amount of easily soluble P-compounds, like e.g. mono-calcium phosphate. It has been reported [3] that P-availability of recovered products is mainly determined by their form of chemical bonding and degree of crystallization. Thus, from the results of the thermally recovered P-products SSA, coal and slag-P, it can be assumed that these contain sparingly availably P-forms, like brushite, stanfieldite and hydroxyl-apatite-like compounds [3].

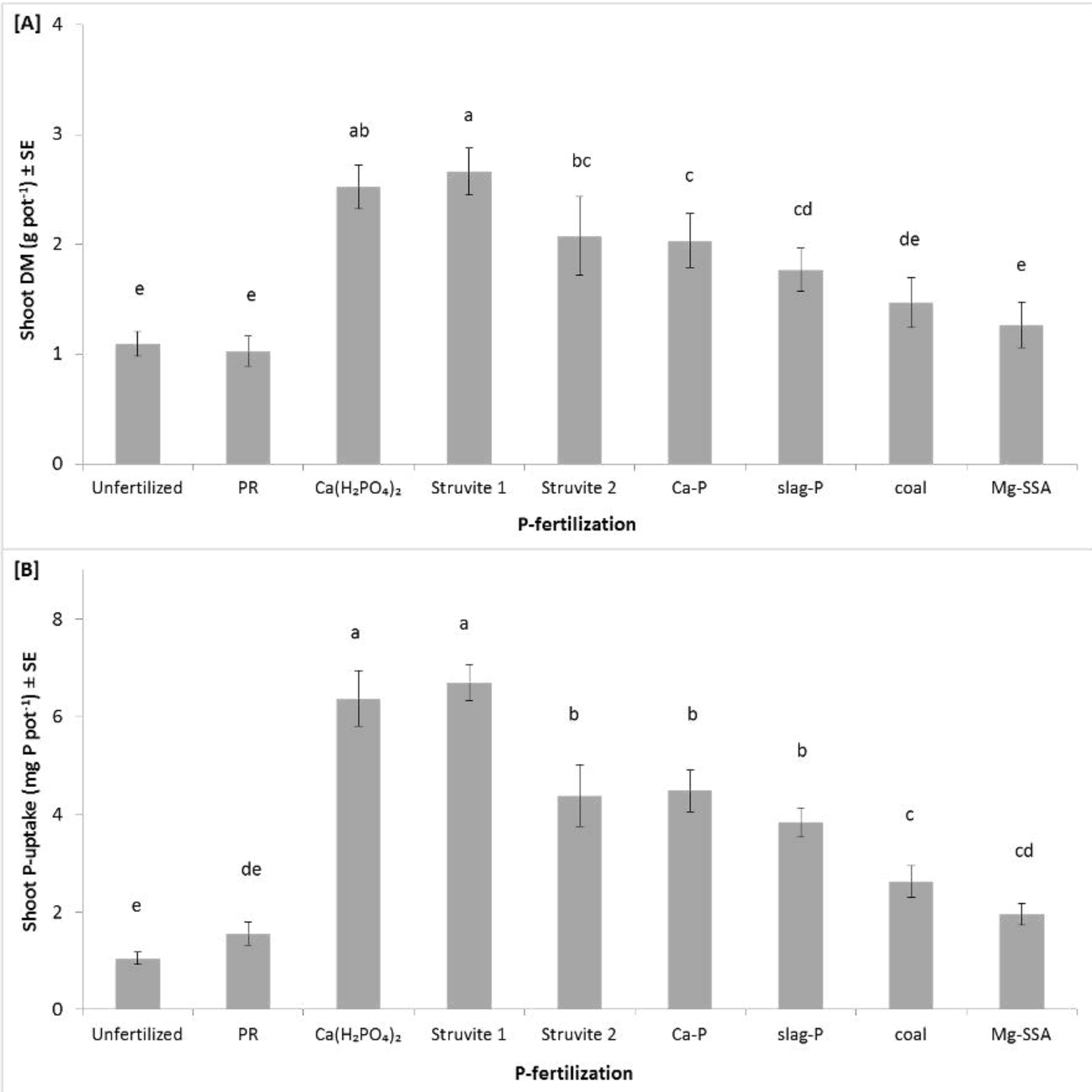

Figure 3: Mean shoot DM [A] and P-uptake [B] of *Zea mays* per pot ± SE in dependency of different P-fertilization. Different letters indicate significant differences ($p \leq 0.05$).

3.2 Field experiment

Aboveground biomass of the silage maize ranged between 118 dt DM ha^{-1} in the unfertilized and compost treatments, and 141 dt DM ha^{-1} in the struvite 1 treatment (Figure 4). Despite, that the initial CAL-P content on the field trial was relatively low (3 mg P 100 g^{-1} soil DM) no differences due to different P-fertilizers could be obtained. The field trial is located on a bio-dynamic research farm with high share of clover in their crop rotation and soils being rich in organic matter. One assumption is that there was a release of P from the soil, possibly from organic P compounds, which are not detected in a CAL-P measurement. Yet, for a concluding evaluation, it remains to evaluate P-content in plant tissue, which still is in analysis, as well as the results of the second year of field experiment in 2015.

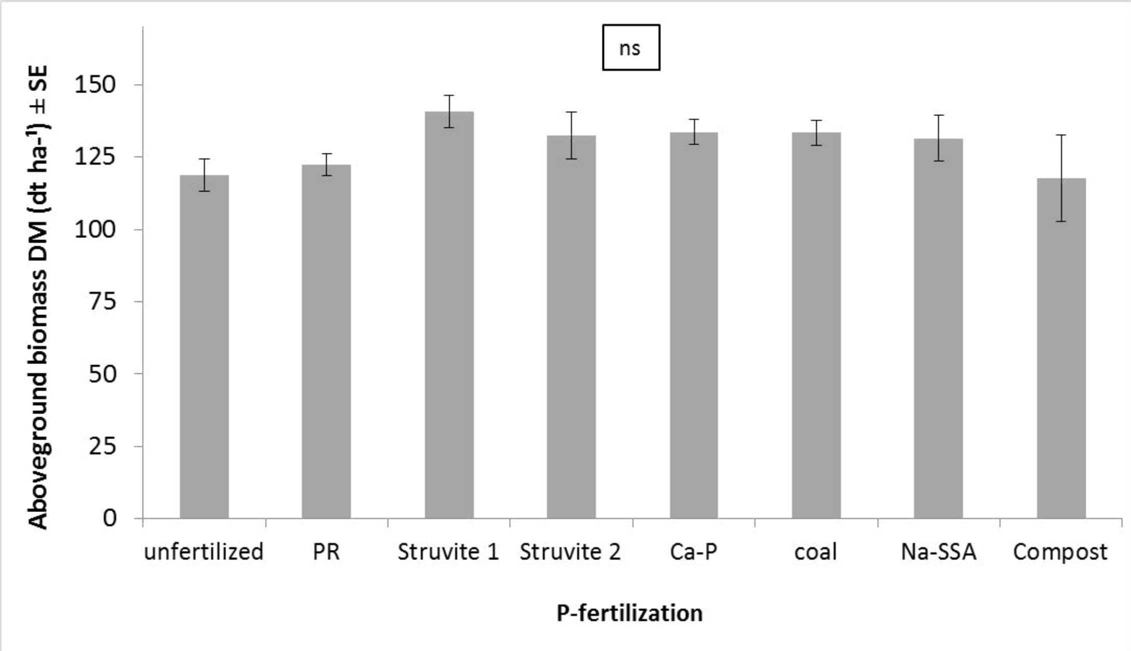

Figure 4: Mean aboveground biomass DM of *Zea mays* grown in a field experiment ± SE in dependency of different P fertilization. ns= not significant ($p \leq 0.05$).

4. Conclusion and outlook

Recycled P products from wastewater can be a potential alternative source for a specific P-supply of plants. It could be shown in pot experiments on both clover and maize, that struvites offer a P-availability comparable to that of highly soluble P-fertilizers on neutral soils. Thermally recovered products like SSA, coal and slag-P were less effective under neutral soil conditions. However, it became obvious that legumes like red clover are very efficient in mobilizing P also from less available sources under neutral soil conditions. Further experiments should be conducted to find soil conditions and plant species under which specific recovered P-products are as effective as could be seen on struvites in the pot experiments. Distinct results shown in the pot experiments could not yet be confirmed under field conditions in the first year of investigation.

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References

[1] VDLUFA (2011): Methodenbuch Band VII, Umweltanalytik, 4. Auflage, Methode 2.1.1. VDLUFA Verlag Darmstadt

[2] Cabeza, R.; Steingrobe, B.; Römer, W.; Claassen, N. (2011): Effectiveness of recycled P products as P fertilizers, as evaluated in pot experiments. Nutr Cycl Agroecosyst 91, 173-184

[3] Nanzer, S.; Oberson, A.; Berger, L.; Berset, E.; Hermann, L.; Frossard, E. (2014): The plant availability of phosphorus from thermo-chemically treated sewage sludge ashes as studied by 33P labeling techniques. Plant and Soil, 1-18

[4] Hauter, R.; Steffens, D. (1985): Einfluss einer mineralischen und symbiontischen Stickstoffernährung auf Protonenabgabe der Wurzeln, Phosphataufnahme und Wurzelentwicklung von Rotklee, Z. Pflanzenernaehr. Bodenk. 148, 633-646

[5] Römer, W. (2013): Phosphordüngewirkung neuer Phosphatrecyclingprodukte. Zeitschrift für Agrarpolitik und Landwirtschaft, BMELV 91, 1

[6] Wollmann, I.; Möller, K. (2014): Assessment of Alternative Phosphorus Fertilizers for Organic Farming: Sewage Precipitation Products. FiBL Factsheets, FiBL-Ordner Nr. 1665 (https://www.fibl.org/fileadmin/documents/shop/1665-sewage-precipitation-products.pdf)

TA-O_27 Sulphur availability from organic materials applied to winter wheat crops

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1. Objectives

To quantify sulphur (S) supply from organic materials applied to winter wheat crops, in order to improve current recommendations on the use of farm manures and biosolilds as sources of available S for arable crops.

2. Methodology

Field experiments were carried out at 3 sites in England cropped with winter wheat over 2 harvest years at each site (between 2009/10 and 2011/12). There were 7 organic material treatments at each site comprising autumn applied cattle farmyard manure (FYM), pig FYM, broiler litter and two biosolids products, and spring applied broiler litter and pig/cattle slurry. The autumn applied organic material treatments were applied to stubble prior to cultivation and drilling with winter wheat, and the spring applied organic material treatments were top-dressed to the growing crop. All organic material treatments were applied at a target application rate equivalent to 50 kg/ha total SO_3.

Samples of organic materials were taken at application and analysed for dry matter, total N (Kjeldahl digestion), available N (NH_4-N, NO_3-N and for broiler litter uric acid-N), total S (ICP-AES) and extractable SO_3 (0.016M KH_2PO_4 extract and analysis via ICP-AES).

Organic material treatments were compared with inorganic (water soluble) S fertiliser response treatments (0, 12.5, 25, 50 and 75 kg/ha SO_3) to determine the fertiliser S replacement value and hence S availability of the applied organic materials. Fertiliser S was applied in a single dose in early spring as potassium sulphate. There were 4 replicates of each treatment arranged in a randomized block design.

In order to ensure, as far as practically possible, that S was the only limiting nutrient, manufacture red fertiliser N was applied at recommended rates, taking into account supply of crop available N from the organic materials. Similarly, fertiliser P_2O_5 and K_2O were applied at recommended rates based on soil analysis.

Grain yields were determined at harvest, and grain samples were taken and analysed for dry matter and total S. Grain yields and grain SO_3 offtake was calculated. Where there was a yield response to applied S, a response curve was fitted to grain SO_3 offtake data. It was not possible to fit a response curve to yield data at any of the sites because the majority of the yield increase from fertiliser S occurred at the first S application rate. Fertiliser S replacement values of the organic material treatments were estimated by comparing S offtake from the organic material treatments with the fitted model to the fertiliser S response plots.

3. Results and discussion

3.1 Crop response to sulphur

The winter wheat crop responded to S at three of the six sites; at Frostenden (Suffolk) in 2009/10 and 2011/12 and at Woburn (Bedfordshire) in 2010/11.

The yield increase from the application of S fertiliser was 0.6 and 0.2 t/ha at Frostenden in 2009/10 and 2011/12 (P>0.05), respectively, and 1.2 t/ha at Woburn 2010/11 (P=0.06). Although these yield increases were not statistically significant, yields from all SO_3 application rates (12 to 75 kg/ha SO_3 fertiliser) were consistently higher than from the zero S control at these sites. The yield increase from applied inorganic S fertiliser occurred at the first S application rate (12 kg/ha SO_3 fertiliser), with no further trend for increasing yields with greater fertiliser S application rates (12 to 75 kg/ha SO_3).

At Frostenden 2009/10, grain S content and SO_3 offtake increased (P<0.05) with increasing S fertiliser application. Spring applications of broiler litter and cattle slurry increased grain SO_3 offtake compared to the zero S control treatment; this increase was significant (P<0.05) for the broiler litter, but not cattle slurry treatment. The difference in S response (yield and grain SO_3 offtake) between the autumn and spring applied organic material treatments was likely to be due to the loss of available S from the autumn applied organic materials via overwinter leaching. For the spring applied organic materials, yields and SO_3 offtakes were greater from the broiler litter than from cattle slurry treatment, reflecting the greater quantity of 'extractable' SO_3 supplied by the broiler litter (36 kg/ha) than cattle slurry (8 kg/ha).

At Frostenden in 2011/12, grain SO_3 offtake increased from 14 kg/ha SO_3 on the zero S control up to *c.*19 kg/ha SO_3 on the 25 to 75 kg/ha SO_3 fertiliser treatments. There was a significant increase in grain SO_3 offtake (P<0.05) for the autumn limed biosolids treatment and the spring broiler litter and pig slurry applications compared with the zero S control.

At Woburn in 2010/11, grain S content and grain SO_3 offtake increased (P<0.05) with fertiliser SO_3 application rate (Figure 1). Grain SO_3 offtake from the autumn applied cattle FYM was the same as from the zero S control, however there was an increase in grain SO_3 offtake from the other autumn applied organic materials, which was statistically significant (P<0.05) for the broiler litter and two biosolids treatments, but not for pig FYM (Figure 1). The lower yields and grain SO_3 offtake from the autumn applied cattle FYM reflects the lower proportion of total S in the 'extractable' form in the cattle FYM and hence lower application rate of 'extractable' SO_3 (3 kg/ha 'extractable' SO_3 applied) compared to the other autumn applied organic materials (12–38 kg/ha 'extractable' SO_3 applied). Grain SO_3 offtake from the spring applied broiler litter and cattle slurry treatments was comparable to the grain SO_3 offtake measured from the highest S fertiliser (75 kg/ha SO_3) rate (Figure 1). The greater yields and grain SO_3 offtakes from the spring compared to autumn applied organic materials (P<0.05), are consistent with the results from the Frostenden site and are likely to be due to loss of available S from the autumn applied organic materials via overwinter leaching.

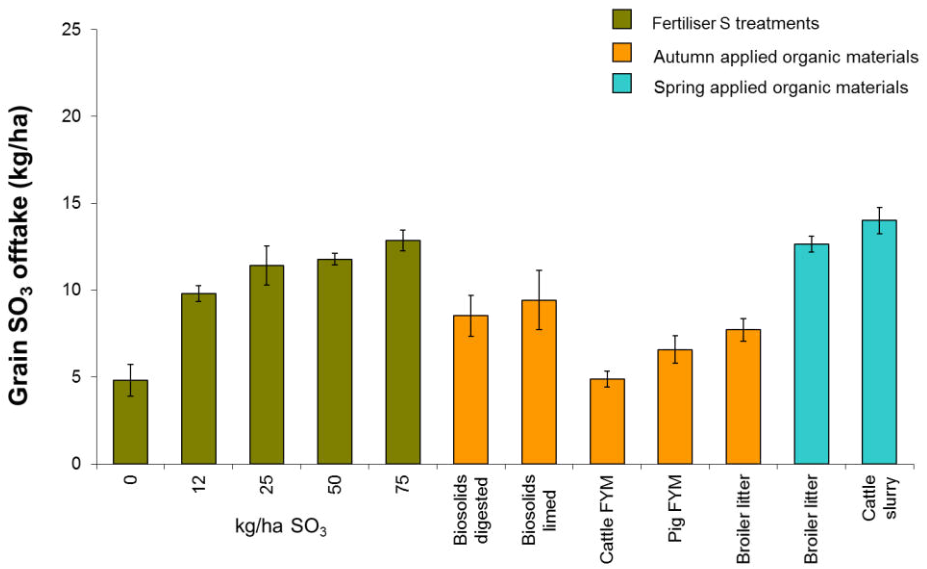

Figure 1: Grain SO_3 offtake at the Woburn field site in 2010/11

3.2 Fertiliser replacement value of applied organic materials

The spring applied cattle slurry at Frostenden 2009/10 had a SO_3 fertiliser replacement value of 7.5 kg/ha SO_3, representing an efficiency of 16% of total SO_3 applied. In 2011/12, the spring applied pig slurry had a SO_3 fertiliser replacement value of 26 kg/ha SO_3 (i.e. an efficiency of 27% of total SO_3 applied). At Woburn in 2010/11, the spring applied broiler litter had a SO_3 fertiliser replacement value of 70 kg/ha SO_3, representing an efficiency of 96% of total SO_3 applied (Table 1). It was not possible to calculate a SO_3 replacement value for the spring applied broiler litter in either year at Frostenden or the cattle slurry at Woburn as the grain SO_3 offtake was greater than the maximum SO_3 offtake from the fertiliser SO_3 response treatments.

At Frostenden in 2011/12, the autumn applied organic materials (excluding cattle FYM), had fertiliser SO_3 replacement values of between 7 and 20 kg/ha SO_3, representing efficiencies relative to fertiliser SO_3 of up to 29% of total SO_3 applied (Table 1). At Woburn in 2010/11, the autumn applied organic materials had fertiliser SO_3 replacement values of between 0 and 10 kg/ha SO_3, representing efficiencies relative to fertiliser SO_3 of up to 20% of total SO_3 applied (Table 1).

Applications of organic materials in the spring were the most effective at supplying SO_3 to the crop, as the SO_3 was applied when the crop was growing and was not subject to overwinter leaching loss.

Table 1: Fertiliser SO_3 replacement value of applied organic materials

Organic material	Sulphur applied		Fertiliser SO_3 replacement value	
	Total SO_3 kg/ha	Extractable SO_3 kg/ha	kg/ha	Efficiency % total SO_3 applied*
Frostenden 2009/10 - Spring applied organic materials				
Broiler litter	66	36	>max	>max
Cattle slurry	46	8	7.5	16
Frostenden 2011/12 - Autumn applied organic materials				
Biosolids – digested	55	9	7.4	14
Biosolids – limed	69	20	20.2	29
Cattle FYM	86	14	0	0
Pig FYM	77	20	7.5	10
Broiler litter	67	41	9.3	14
Frostenden 2011/12 - Spring applied organic materials				
Broiler litter	68	42	>max	>max
Pig slurry	99	21	26.3	27
Woburn 2010/12 - Autumn applied organic materials				
Biosolids – digested	48	16	9.6	20
Biosolids – limed	75	16	10.0	13
Cattle FYM	29	3	0.4	1
Pig FYM	60	12	4.0	7
Broiler litter	59	38	6.9	12
Woburn 2010/12 - Spring applied organic materials				
Broiler litter	73	50	69.7	96
Cattle slurry	68	41	>max	>max

* Fertiliser replacement value (kg/ha) as a percentage of total SO_3 applied in the organic material.
** Fertiliser replacement value (kg/ha) as a percentage of 'extractable' SO_3 applied in the organic material.

3.3 Sulphur availability from organic materials

For the spring organic material applications (broiler litter and slurry), there was a good relationship between the recovery of SO_3 in the grain (treatment minus control) and the amount of 'extractable' SO_3 applied in the organic materials (Figure 2). This suggests that, for spring applied organic materials, 'extractable' SO_3 is a good indicator of crop available SO_3. For the spring applied organic materials there was a linear relationship between the fertiliser SO_3 replacement value and quantity of extractable SO_3 applied in the organic materials, and therefore the availability of S from spring applications of organic materials can be assumed to be equivalent to the proportion of total SO_3 which is present in 'extractable' form. Based on the analysis of organic materials used in this project, 'extractable' SO_3 was 15% of total SO_3 for cattle FYM, 25% for pig FYM, 60% for broiler litter, 35% for slurry (cattle and pig) and 20% for biosolids.

'Extractable' SO_3 from autumn applications of organic materials may be lost via overwinter leaching, with the quantity lost depending on the amount of 'extractable' SO_3 applied, soil type and overwinter rainfall. For the autumn applications, there was no relationship between grain SO_3 recovery (or fertiliser replacement value) and the amount of 'extractable' SO_3 applied; this was most likely a reflection of overwinter SO_3 leaching losses. At the Frostenden (2011/12) and Woburn (2010/11) sites, where fertiliser SO_3 replacement values for autumn organic material applications could be calculated, SO_3 use efficiency was lower than for spring applied organic materials; ranging from 0 to 13% of total SO_3 for livestock manures (mean 5% total SO_3) and from 5 to 29% for biosolids (mean 14% total SO_3). Based on the data from this project, for autumn applied organic materials we suggest efficiency figures of 5–10% of total SO_3 for livestock manures and 10–20% of total SO_3 for biosolids.

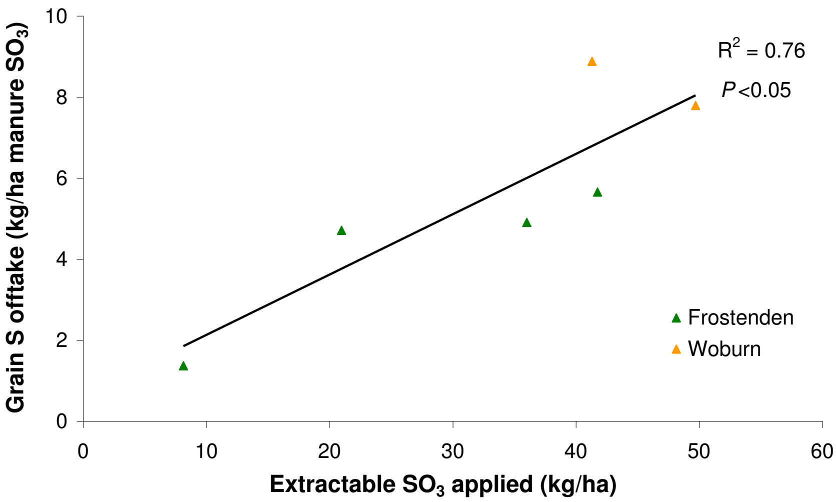

Figure 2: Relationship between the grain recovery of SO_3 from the spring applied organic materials and quantity of 'ex tractable' SO_3 applied

4. Conclusion and outlook

Organic materials can be a source of crop available S. However, application timing (autumn compared to spring) has an effect on crop available S supply, with crop S offtake consistently higher from the spring compared to autumn manure applications. Readily available S from organic manures applied in the autumn may be lost via over-winter leaching, with the quantity lost dependent upon soil type and the amount of overwinter rainfall.

The results from the field experiments showed that for the spring-applied organic manures, 'extractable' S was a good indicator of crop available S supply from the manures, and could generally be considered equivalent to inorganic fertiliser S. On this basis, the S use efficiency for spring organic manure applications can be assumed to be equivalent to the proportion of total S in the extractable form.

Results from this project showed a lower S use efficiency from autumn compared with spring-applied organic materials; typical autumn S use efficiencies in the range 5–10% of total SO_3 for livestock manures and 10–20% of total SO_3 for biosolids.

This work has led to a better understanding of the available S supply from organic materials, allowing guidance to be produced for farmers on the availability of S from applications of organic materials. This updated guidance was published by HGCA in an information sheet for farmers on 'Sulphur for cereals and oilseed rape'[1]. This is likely to improve farm profitability by reducing S applications to cereal crops receiving applications of organic materials.

Acknowledgements

Funding for this study from the HGCA, Anglian Water Ltd and Severn Trent Water Ltd is gratefully acknowledged.

References

[1] HGCA Information Sheet 28. Sulphur for cereals and oilseed rape. Available from http://www.hgca.com/media/357116/is28-sulphur-for-cereals-and-oilseed-rape.pdf

Thematic area TA – Quality fertilizers from residues (Poster presentations)

Christiane Lüdtke: Art from Tetrapak

TA-P_01 Apparent nitrogen recovery in Italian ryegrass (*Lolium perenne*, L.) from the solid fraction of two digestates

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1. Objectives

Plant available N released after the incorporation into the soil of digestate solid fractions (DSF) is a basic knowledge in order to draw up affordable fertilization plans. The objective of this work was to estimate, with a greenhouse pot experiment, the amount of N recovered in Italian ryegrass (*Lolium perenne*, L.) after application of two different DSFs.

2. Methodology

The soil used in the experiment had the following characteristics: sandy-loam texture, pH_W 6.5, Tot. N and Tot. C 0.97 and 7.81 g kg^{-1}, Olsen-P 10 mg kg^{-1}, and exchangeable K 55 mg kg^{-1}. Before the beginning of the experiment, the bulk soil was fertilised with an equivalent amount of 23 kg ha^{-1} of P (as superphosphate) and 161 kg ha^{-1} of K (as potassium sulphate). The soil was moist-sieved at 4 mm and then mixed with washed neutral sand in a ratio of 2 parts of soil and 1 part of sand on a weight basis. Each pot was filled with about 170 g of vermiculite as drainage layer and with 2.8 kg of the soil-sand mixture (hereafter named soil solely). The experiment considered the following treatments: 1) unfertilized soil (CON); and soil fertilised with 2) ammonium sulphate (AS); 3) DSF obtained from a cattle livestock farm (CLF); 4) DSF obtained from a slaughterhouse (SH). The two DSFs (Table 1) were sampled and hand grind at 4 mm, in order to increase their homogeneity. Fertilisers were applied at a rate of 100 and 340 kg ha^{-1} for AS and the two DSFs, respectively.

Table 1: Characteristics of the digestate solid fractions (DSF) used in the greenhouse experiment.

DSF	DM (%)	C (% DM)	Total N (% DM)	NH_4-N (% DM)	C/N
DSF from cattle livestock farm (CLF)	32.0	45.3	1.99	0.30	23
DSF from slaughterhouse (SH)	21.1	43.9	6.22	1.18	7

DM-dry matter

All materials were thoroughly mixed with the soil; thereafter, pots were gently pressed in order to reach a bulk density of about 1.3 g cm^{-3} and water was added to attain a soil water content (WC) corresponding to 50% of the water holding capacity (WHC). Pots were sown with Italian ryegrass, cv. Pamir, and, after emergence, 35 plants per pot were left (Figure 1). Greenhouse conditions were kept constant during all the experiment, with 16 h of lighting (150-180 W m^{-2}) and 8 h of dark per day, and 22-30 and 16-22°C during lighting and dark, respectively. Soil WC was kept between 25 and 50% of WHC by weighting of pots. Above ground biomass (AGB) was cut after 45, 66 and 123 days after seeding while roots biomass was determined at the end of the experiment. Plant samples were oven dried at 105°C, grind at 0.2 mm and analysed for their total N content using an elemental analyser. Apparent nitrogen recovery (ANR) in AGB and roots was calculated as the ratio between the N uptake measured in the fertilised treatments (net to N uptake measured in CON) and the applied N. Cumulated AGB and ANR in AGB were calculated summing the values measured in the three ryegrass cuts. Analysis of variance (ANOVA) was performed separately for AGB, root biomass, ANR in AGB and ANR in roots using the SPSS procedure UNIANOVA (SPSS Version 22.0.0). Mean separation was conducted with the HSD Tukey test ($P < 0.05$). The treatment was considered a fixed factor, while the block was random. Homogeneity of variances was evaluated using the Levene test ($P < 0.05$). Within the text, significant effects of fertiliser application are reported with a P value lower than 0.05.

Figure 1: Pots sown with Italian ryegrass in the greenhouse experiment.

3. Results and discussion

3.1 Above and below ground biomass

Results of cumulated AGB are reported in Figures 2 and 3. Total AGB in CON (3.5 g pot^{-1}) was lower compared to that of fertilised treatments (4.9, 5.7 and 7.1 g pot^{-1} in AS, CLF and SH, respectively). The three fertilisers gave different cumulated AGBs, in the order AS < CLF < SH.
At the end of the experiment, root biomass (Figure 3) accounted for 37-39% of total plant biomass. Differences in root biomass among treatments followed the same trend observed for AGB, with higher biomass in the soil fertilised with the two DSFs compared to the unfertilized soil.

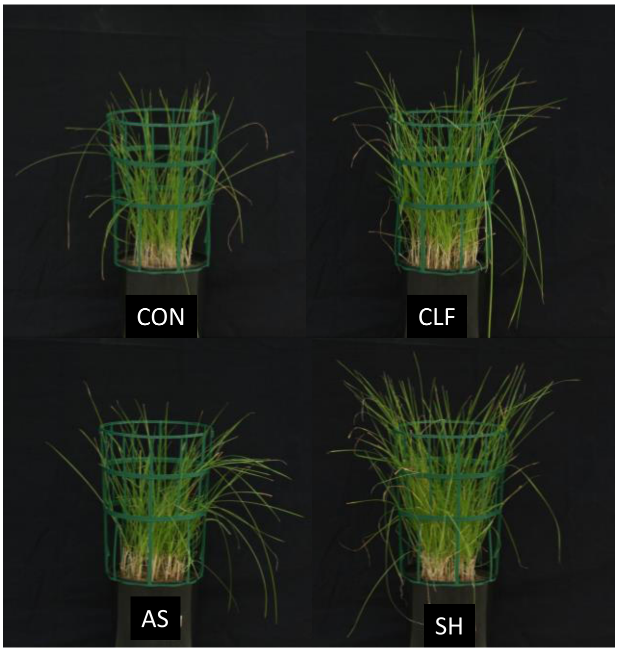

Figure 2: Above ground biomass of Italian ryegrass in unfertised soil (CON) and soil fertilised with ammonium sulphate (AS), the digestate solid fraction from cattle livestock farm (CLF), and the digestate solid fraction from slaughter house (SH).

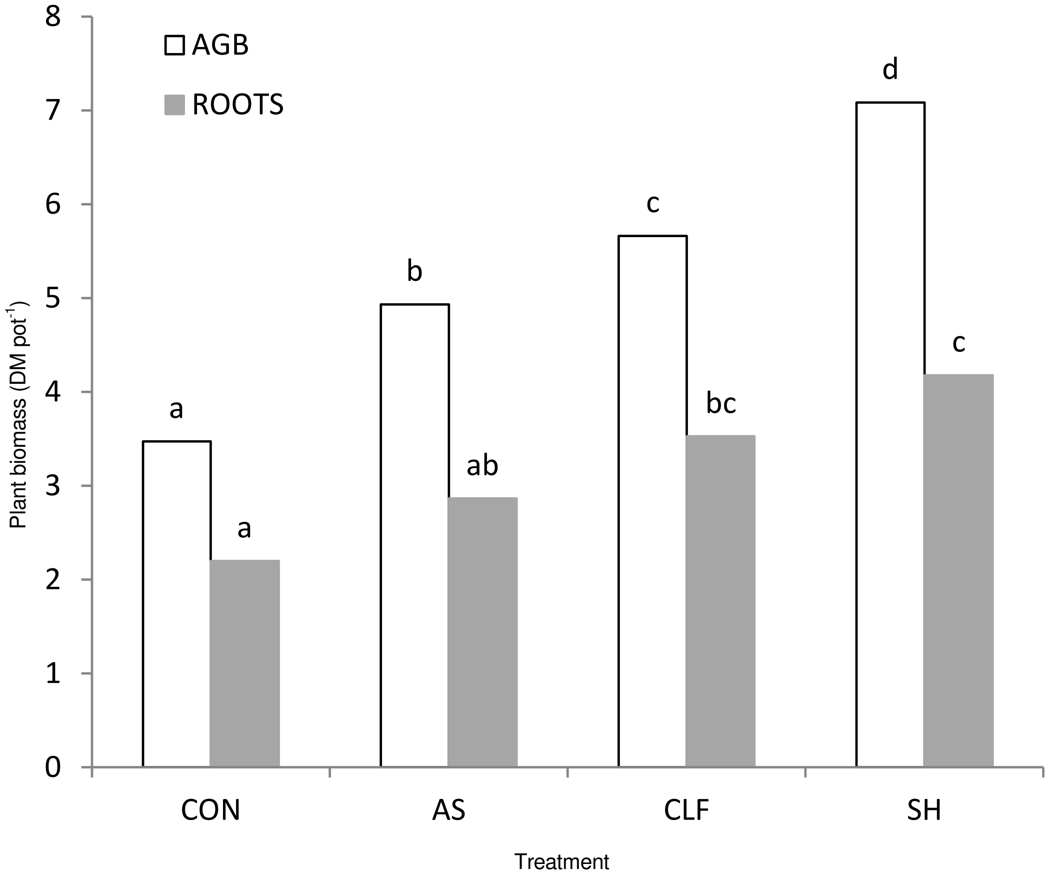

Figure 3: Cumulated above ground biomass (AGB) and root biomass of Italian ryegrass. Letters indicate significant differences ($P < 0.05$) among treatments within above and below ground biomass (HSD Tukey test). CON: unfertilized soil; AS: soil fertilised with ammonium sulphate; CLF: soil fertilised with the digestate solid fraction from cattle livestock farm; SH: soil fertilised with the digestate solid fraction from slaughterhouse.

3.2 Apparent N recovery

Results of cumulated ANR in AGB and roots are reported in Figure 4. Cumulated recovery of N in AGB was higher in AS (72%) compared to CLF (13%) and SH (37%). Similar differences were found at the 1st ryegrass cut, when ANR in AGB was 49%, 21% and 3% in AS, SH and CLF, respectively. In the 2nd and 3rd cut about 20% of applied N was recovered in AS and SH treatments, while a lower percentage (10%) was recovered in CLF. Recovery of applied N in roots was not significantly different among treatments and averaged 7% of the applied N. Results of ANR are in good agreement with the availability of N from the three fertilisers. In the 123 days of Italian ryegrass growth, SH gave a net release of N during decomposition, most likely due to the low C to N ratio and due to the lack of fibrous components (it originated form a digested slaughtering waste). Conversely, the higher C to N ratio and presence of fibers in CLF probably resulted in a net N immobilisation for at least in the first stages of decomposition, and thus in a reduced availability of N for the Italian ryegrass.

In a field trial (Cavalli et al., 2014) in which CLF was applied to silage maize, ANR in the first year of the experiment was very low (1%), while in the second year 17% of the applied N was recovered in maize AGB. These results confirm the low availability of N for crops after the application of CLF-like DSFs.

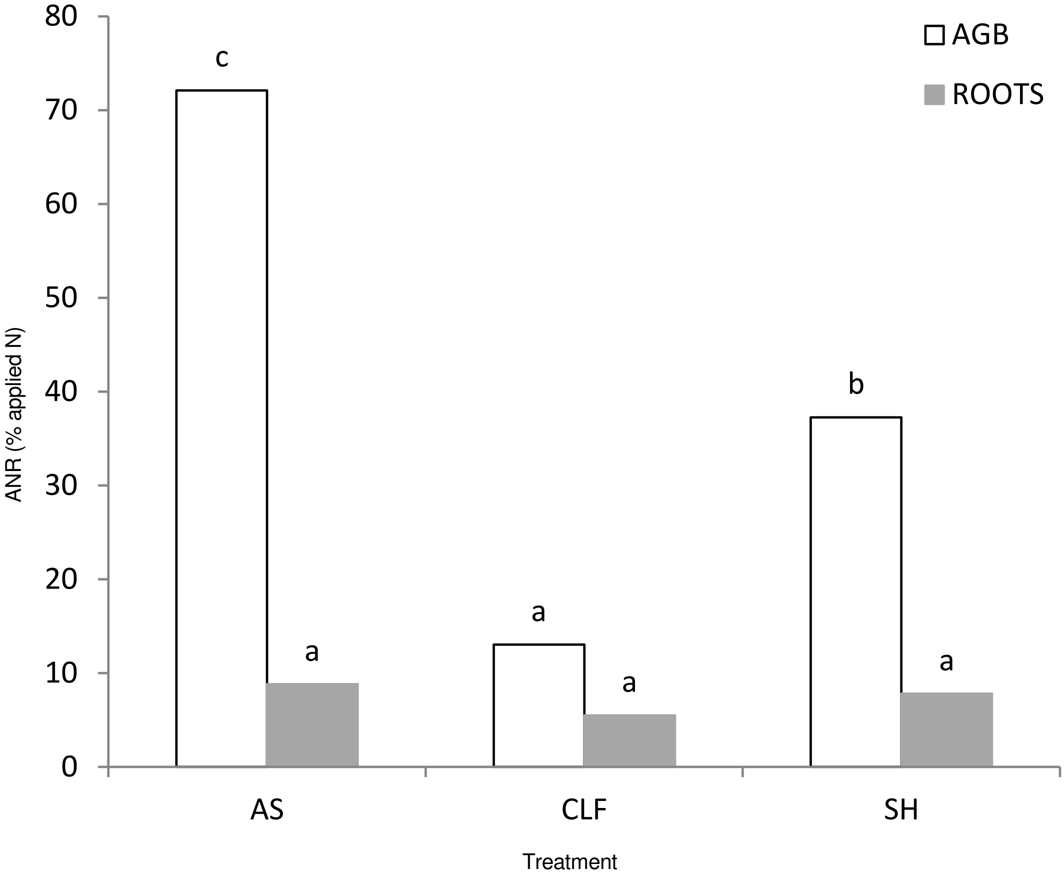

Figure 4: Cumulated apparent N recovery (ANR) in above ground biomass (AGB) and roots of Italian ryegrass. Letters indicate significant differences ($P < 0.05$) among treatments within above and below ground biomass (HSD Tukey test). AS: soil fertilised with ammonium sulphate; CLF: soil fertilised with the digestate solid fraction from cattle livestock farm; SH: soil fertilised with digestate solid fraction from slaughterhouse.

4. Conclusion and outlook

The results of this pot experiment confirmed the high variability in the short-term availability of N from DSFs of different origins. Solid fraction of digested cattle slurry (like CLF), with high C to N ratio and low NH_4-N content, can immobilise consistent amounts of N during decomposition into the soil. Opposite, solid fraction from non-fibrous materials, with low C to N ratio, is quickly mineralised, releasing relevant amount of N that is available for crop growth. For DSFs such as CLF, it could be proper to integrate plant N fertilisation with mineral fertilisers. Moreover, for these DSFs it is expected a slow release of N in the long period, as a consequence of the remineralisation of previously immobilized N and the mineralisation of the more resistant fractions of organic matter. Both these processes give rise to N residual effects that, in the long time can consistently enhance ANRs of these materials.

Acknowledgements

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References

[1] Cavalli, D.; Cabassi, G.; Borrelli, L.; Fuccella, R.; Degano, L.; Bechini, L.; Marino, P. (2014): Nitrogen fertiliser value of digested dairy cow slurry, its liquid and solid fractions, and of dairy cow slurry, Italian J Agron 9, 71–78.

TA-P_02 Optimal placement of pelleted organic fertilizer in spring oat

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1. Objectives

For fertilization with granulated mineral fertilizers, there are guidelines about on where to place the fertilizer in relation to the seed row to gain maximum fertilizer efficiency [1]. Organic fertilizers are due to their physical properties usually more difficult to apply with accurate precision. However, when pelleted, it is possible. Apart from increasing availability of nutrients to the crop, placement close to the crop row may also help reduce weed pressure [2], by fertilizing the crop rather than the weeds. This is of special importance in organic production where chemical weed control is banned and where pelleted organic fertilizers are most commonly used. The objective with this study is to investigate the nitrogen fertilization effect on yield and weed abundance in spring oats depending on placement of pelleted organic fertilizer at different soil depths and distances from crop row. This paper presents results from studies performed during the first year of the project.

2. Methodology

2.1 Field experiments

The effects on crop nitrogen uptake, grain yield and weed abundance of placing pelleted meat bone meal (MBM) at different soil depths and at different distances from the seed row was tested in two field experiments in Sweden (58°N,13°E). One was in a field with silty clay and the other in a field with sandy loam. The treatments (Table 1) were randomized within four blocks. Each plot was only 70 cm long and 100 cm wide and sown and fertilized by hand with a crop row distance of 25 cm. The seeds were sown at 4 cm depth and the pellets were placed at two or three depths (1, 4 and 8 cm on the sandy soil and 1 and 4 cm depth on the clay soil) and at three distances (0, 4 and 12.5 cm) from row. There were also control treatments without N fertilization, with mineral N fertilizer and with surface broadcasting with shallow (0-1 cm) incorporation. All fertilized treatments received 60 kg total N ha^{-1}.

Table 1: Treatments in the field experiments performed in oats during 2014. MBM = meat bone meal.

Treatment	Fertilizer	Soil incorporation	Distance from row
1	-	-	-
2	Ammonium nitrate	1 cm	4 cm
3	Pelleted MBM	8 cm	0 cm
4	Pelleted MBM	8 cm	4 cm
5	Pelleted MBM	8 cm	12.5 cm
6	Pelleted MBM	4 cm	0 cm
7	Pelleted MBM	4 cm	4 cm
8	Pelleted MBM	4 cm	12.5 cm
9	Pelleted MBM	1 cm	0 cm
10	Pelleted MBM	1 cm	4 cm
11	Pelleted MBM	1 cm	12.5 cm
12	Pelleted MBM	0-1 cm	Broadcasting

Weeds were counted in beginning and end of June and were then harvested to measure dry matter yield. The oat was harvested at ripening, by cutting the straw at soil surface in a net area of 50 cm x 50 cm within each plot. Plant samples were threshed in the laboratory and grains and straw were measured separately and analyzed for N contents.

2.2 Statistical analyses

The yields in treatments 3-11 were compared using General Linear Model two factor analysis in Minitab software. Number of weeds in all treatments were compared using mixed model with the factors block, treatment, date and treatment x date in SAS software. The weed biomass in all treatments were compared using mixed model with the factors block, treatment and treatment x block in SAS software.

3. Results and discussion

3.1 Yield effects in the field experiment on clay soil

In the field experiment on the clay soil, grain yield in the unfertilized treatment was 1200 kg ha^{-1} and in the treatment with broadcasting 1600 kg ha^{-1}. There was a tendency for higher grain yield the closer to crop row (p=0.058) and the deeper in the soil (p=0.054) the pellets was placed with no interaction between depth and distance from row (p=0.45) (Figure 1a). Grain yield increased on average with 300 kg ha^{-1} when incorporated to 4 cm compared to 1 cm (p=0.054). Placement of pellet 0 cm from crop row gave 470 kg ha^{-1} higher yield than placement 12.5 cm from crop row (p=0.051).

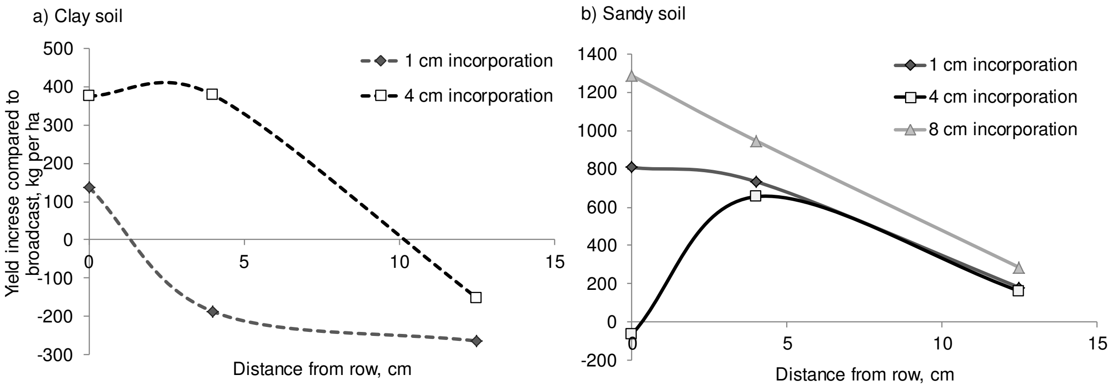

Figure 1: Yield increase from placement of pelleted fertilizer at different depths and distances from crop row compared to broadcasting in field experiments performed on (a) clay soil and (b) sandy soil.

3.2 Yield effects in the field experiment on sandy soil

In the field experiment on the sandy soil, grain yield in the unfertilized treatment was 3700 kg ha^{-1} and in the treatment with broadcasting 4100 kg ha^{-1}. There was a tendency to differences in yield depending on both incorporation depth (p=0.07) and distance to crop row (p=0.06) without interaction between depth and distance from row (p=0.30) (Figure 1 b). Grain yield increased by on average 500 kg ha^{-1} if applied at 8 compared to 4 cm depth (p=0.02), partly because placing pellet together with seeds were unfavourable in this trial. The difference in yield between incorporation at 1 and 8 cm depth was smaller (230 kg ha^{-1}) and not significant (p=0.068). Placement 4 cm from the crop row increased yield with on average 485 kg ha^{-1} compared to placement 12.5 cm from crop row.

3.3 Effects on weeds

The weed density was high in the field experiment on the sandy soil in the beginning of the growing season, but was significantly reduced from 2 to 24 June (Table 2). There were no significant differences between treatments in number of weed plants or weed biomass on 24 June, probably because of the vigorous crop competing well with weeds. In the field experiment on the clay soil, the crop was sparser. Here there were a significantly higher number of weed plants in treatments 10-12 (Table 2), with more shallow incorporation in combination with some distance to crop row. The weed biomass was significantly higher in treatments 8 and 12. In other words, the weeds tended to be more frequent in treatments with either broadcasting or placement far from the crop row. However, the weed density was still rather low and did probably not affect the crop yield.

Table 2: Weed density in different treatments (see Table 1) in the experiments at clay soil and sandy soil.

Treatment	Clay soil Number of weeds plants m^{-2} 02-jun	Clay soil Number of weeds plants m^{-2} 24-jun	Clay soil Weed bio-mass kg DM ha^{-1} 24-jun	Sandy soil Number of weeds plants m^{-2} 02-jun	Sandy soil Number of weeds plants m^{-2} 24-jun	Sandy soil Weed biomass kg DM ha^{-1} 24-jun
1	168	172	152	452	224	232
2	180	212	240	376	148	188
3				436	192	200
4				424	216	232
5				272	132	228
6	168	140	240	388	224	300
7	168	172	188	376	164	204
8	156	168	364	336	160	288
9	208	208	176	396	176	164
10	192	264	232	448	200	184
11	196	228	260	396	220	328
12	220	316	308	472	224	412

3.4 Aboveground plant nitrogen

In the field experiment on the clay soil, the above ground crop nitrogen content amounted to 35 kg N ha^{-1} on average in the treatments with pelleted fertilizers, with around 31 kg N ha^{-1} in the lower yielding treatments and 38 kg N ha^{-1} in the higher yielding treatments (Figure 2a). Above ground weed nitrogen on the clay soil was 2-6 kg N ha^{-1}. In the field experiment on the sandy soil, the above ground crop nitrogen content amounted to 100 kg N ha^{-1} on average in the treatments with pelleted fertilizers, with around 93 kg N ha^{-1} in the lower yielding treatments, 105 kg N ha^{-1} in the intermediate yielding treatments and 115 kg N ha^{-1} in the highest yielding treatment (Figure 2b). Above ground weed nitrogen on the sandy soil was 3-7 kg N ha^{-1}.

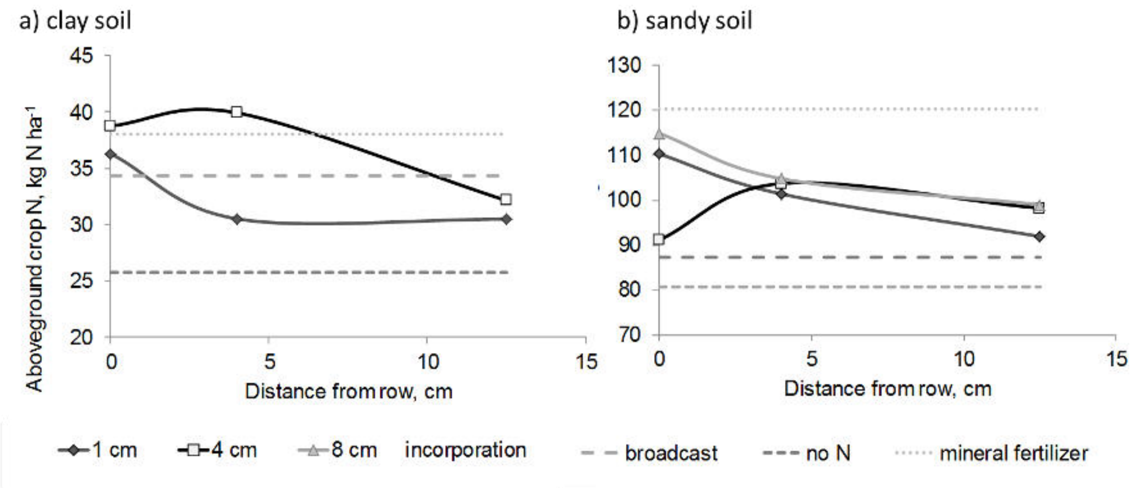

Figure 2: Aboveground crop nitrogen in treatments with different placement of pelleted fertilizer at different depths and distances from crop row compared to broadcasting, no fertilization and mineral fertilization in field experiments performed on (a) clay soil and (b) sandy soil.

4. Conclusion and outlook

It is too early to draw conclusions from only one year of results. However, the results indicate that placement of organic fertilizer close to the crop row and incorporation deeper in the soil can improve yield and reduce weed biomass compared to broadcasting and fertilizing shallowly between rows.

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References

[1] Rasmussen, K.; Rasmussen, J.; Petersen, J. 1996. Effects of fertiliser placement on weeds in weed harrowed spring barley. Acta Agriculturae Scandinavica Section B: Plant and Soil Science 46, 192–196.

[2] Huhtapalo, Å. 1982. Scandinavian principles for fertilizer placement. Utilization of fertilizer-N. In: Proceedings of the 9th conference of the international soil tillage research organization, ISTRO, 669-674.

TA-P_03 Reuse potential of urine as a source of plant micronutrients

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1. Objectives

Soil micronutrient deficiencies are widespread in India (20), South America (5), Canada (10), Thailand (14), Pakistan (3) and Nepal (2). It is also a problem in the USA, Tropical Latin America and to major climatic zones of tropical Africa (22). Important agricultural soils of the world are 49%, 31%, 15%, 14%, 10% and 3% deficient in Zinc (Zn), Boron (B), Molybdenum (Mo), Copper (Cu), Manganese (Mn) and Iron (Fe) respectively (19). Soil micronutrient concentration is highly affected by the type of soil parent materials, soil pH, organic matter content, soil texture and mineralogy and other physical, chemical, and biological conditions in the rhizosphere (5, 22, & 1). Additionally, the vertical distribution of nutrients in the soil can be affected by weathering, atmospheric deposition, leaching and biological cycling (9). The main objective of this study was to investigate the potential of human urine as a source of plant micronutrients. Specifically, the study wanted to find out the micronutrient uptake of maize *(Zea mays*) with urine application during vegetative growth stage. Data on uptake will tell us how much of each nutrient is contributed by the treatments.

2. Methodology

A pot experiment with maize was conducted under greenhouse condition at the Institute of Wastewater Management and Water Protection, Hamburg University of Technology, Germany. Each plant was grown in a 5-liter plastic pot. Growing medium was composed of 73% woodchips, 18% soil and 9% sand. The use of woodchips in this experiment is a demonstration of a poor soil with low retention capacity for water and nutrients. Urine was compared with a synthetic (chemical) commercial granular fertilizer containing 16% Nitrogen (N), 7% Phosphorous (P), 13% Potassium (K), 2% Magnesium (Mg), 6% Sulfur (S), 0.02% Boron (B), 0.01% Zinc (Zn), 0.03% Manganese (Mn), 0.008% Molybdenum (Mo), 0.028% Copper (Cu) and 0.29% Iron (Fe). Only water was applied to the control. Urine was diluted to 3:1 ratio (water and urine respectively). In the fertilizer treatment, 13 grams/plant was applied and irrigated with water. Irrigation schedule with water and the urine treatment was the same for all pots (including volume). Volume was increased gradually overtime. Each treatment was replicated 5 times. Light was provided for 12 hours/day. Twenty-six (26) days after sowing (DAS), plant height was measured from the base of the plant shoot up to the base of the tassel. After measuring plant height, the plants were cut at the base of the shoot, placed individually in paper bags and dried in the oven for drying. After drying, each plant sample was milled and analyzed for plant nutrient contents using Inductively Coupled Plasma Spectrometry at the Institute of Plant Nutrition, Leibniz University of Hannover, Germany. Analysis of urine revealed N contents 0.01 - 1.80 mg/L nitrate and 0.02 – 0.95 µg/L ammonium.

3. Results and discussion

3.1 Plant biomass

Control plants had stunted growth with the lowest average dry weight and plant height (Table 1). The leaves were thin with reddish-purple color (Figure 1) and plant growth did not increase further after 20 DAS. In contrast, urine treated plants had green healthy vigorous leaves (Figure 1). Plants applied with urine had almost the same physical appearance with the fertilizer treated plants (Figure 1) and in both treatments, plants had shown similar performance in growth and development during the vegetative growth stage as observed in the experiment. Vegetative growth phase is a period of high nutrient uptake and the maize plant stores nutrients needed for its entire life cycle. Difference in biomass accumulation (dry weight) between urine and fertilizer treated plants was only very slight (Table 1).

Table 1: Plant height of maize (cm) and dry weight (g) at 26 days after sawing (DAS) for different nutrient supplementation variants.

Parameter	Control	Urine	Fertilizer
Plant height	20.76	56.80	79.30
Dry weight	0.67	35.08	40.36

3.2 Uptake of plant nutrients

Plant shoot analysis revealed macro- and micro-nutrient uptake including other detected elements (Table 2, 3 & 4). Generally, micronutrients (B, Zn, Fe, Mn & Cu) are needed by plants only in minute quantities while macronutrients (N, P, K, Ca & Mg) are required in high amounts (6 & 3). Regardless of plant requirement, they are all considered essential and play equally important functions in plant physiological processes (8 & 5). B is essential for plant cell wall formation, Cu catalyzes several plant processes, Fe promotes chlorophyll formation, Mn plays a direct role in photosynthesis and Zn aids plant growth hormones and enzyme system.

CONTROL	URINE	URINE	FERTILIZER

Figure 1: Plant growth and development at 26 DAS.

All micronutrients in the urine treatment were higher than the control (Table 2) and such results confirmed the contents of the nutrients in the urine which were readily available for plant uptake. It is also very interesting to note the very high Mn content in the urine compared to the control and fertilizer treatments (Table 2). Significant differences were found among treatments for B, Zn, Mn and Fe at 5% level of significance. Plant micronutrient contents in urine were also reported in some studies (11 & 15) but works on agricultural investigation are very limited. Maize accumulated minimum amounts of B, Cu, Mn and Zn in the initial stages of development and at physiological maturity, the above ground parts accumulate nutrients as follows: Zn>Mn>Cu>B (4). The total amount required to produce one ton of corn is: 900mg B; 19000-20000mg Cu; 42000-46000mg Mn; 100000-194000mg Zn (4).

Table 2: Plant shoot uptake (mg/kg dry matter) of micronutrients at 26 DAS.

Treatment	B	Zn	Mn	Fe	Cu
Control	21.01	31.61	73.86	15.51	6.11
Urine	30.04	56.59	136.16	41.95	8.64
Fertilizer	52.35	62.83	77.46	55.48	8.69

It is interesting to note that urine treated plants had the lowest P content but had the highest contents of Ca and Mg (Table 3) compared to the control and fertilizer treatments. There is a significant difference among treatments for Ca and Mg at 5% level of significance. Comparing P contents of control and the urine treatment (Table 3), urine seemed to have a negative effect on uptake of available P from the medium.

Table 3: Plant shoot uptake (mg/kg dry matter) of macronutrients at 26 DAS.

Treatment	P	K	Ca	Mg
Control	4315.66	17838.79	1846.44	1113.86
Urine	2583.58	16928.95	3545.48	2135.27
Fertilizer	4851.88	18634.41	2593.81	1479.17

Contents of other elements such as Barium (Ba), Cobalt (Co), Nickel (Ni), Sodium (Na), Silicon (Si), Strontium (Sr) and Aluminum (Al) were also detected in urine (Table 4). Highly significant differences were found among treatments for Sr and Al at 1% and 5% level of significance whereas no significant differences were found for the other elements. Ni is actually one of the essential plant micronutrients (12). It is a constituent of urease, and small quantities of Ni are essential for some plant species (18).

Table 4: Plant shoot uptake (mg/kg dry matter) of other detected elements at 26 DAS.

Treatment	Ba	Co	Ni	Na	Si	Sr	Al
Control	20.26	0.87	2.58	3,018.33	276.45	0.22	31.73
Urine	52.33	0.95	3.68	2,487.44	434.15	4.67	41.52
Fertilizer	64.87	0.89	3.10	4,027.20	577.52	2.97	116.70

The kind of plant nutrients and other elements detected were all the same (except for the amount) for both the control and the urine treated plants (Table 2, 3 & 4). This means that the huge difference in growth and development between the both treatments must be due to N in the urine. Urine has been proven to be a good source of N for plants (17, 7 & 16). N demanding plants like maize normally show a clear and immediate respond to urine application (13). N is a vital component of all plant proteins and is one of the main chemical elements required for photosynthesis and plant growth. This proves that the deficiency of any single essential nutrient usually results to specific nutritional ailments, exhibition of unmistaken signs of hidden hunger and significant reduction of crops' growth and yield (1 & 3) which were all manifested by the control plants. In this experiment, N was the limiting nutrient.

4. Conclusion and outlook

Urine is as good as commercial synthetic fertilizer in providing macro- and micro- plant nutrients particularly necessary for the vegetative growth stage. Crops can still be produced even in naturally poor and infertile soils considering urine as an important agricultural input. Using urine as fertilizer can empower small farmers in marginal areas by overcoming food insecurity situation due to insufficient harvest. In large scale crop production, reusing urine can definitely reduce dependence on expensive chemical fertilizers. Several works have already addressed the hygienic concerns in the reuse of urine. Storage of urine is a simple technique that elevates urine pH to the alkaline level creating an unfavorable condition for the survival of pathogenic bacteria.

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References

[1] Akinrinde, E. A. (2006): Issues of optimum nutrient supply for sustainable crop production in tropical developing countries, Pakistan Journal of Nutrition 5 (4): 387-397, Asian Network for Scientific Information, ISSN 1680-5194

[2] Andersen, P. (2007): A review of micronutrient problems in the cultivated soil of nepal - An issue with implications for agriculture and human health, Mountain research and development, Vol 27, No 4, Nov 2007: 331-335

[3] Askari, A.; Siddiqui, I. H.; Yasmin, A.; Qadiruddin, M.; Jafri, R.; Zaidi, S. H. (1995): Studies on the essential trace elements on the growth and yield of two solanaceous plants, Journal of Islamic Academy of Sciences 8:1, 9-14, 1995

[4] Borges, I. D.; Von Pinho, R. G.; Pereira, J. L. (2009): Micronutrients accumulation at different maize development stages, Ciênc. agrotec., Lavras, v. 33, n. 4, p. 1018-1025, jul./ago. 2009

[5] Fageria, N.; Stone, L. (2008): Micronutrient deficiency problems in south america, Micronutrient deficiencies in global crop production B.J Alloway (ed), Chapter 10, Springer Science + Business Media B.V.

[6] FAO. (1983): Micronutrients, Fertilizer and plant nutrition bulletin 7, Food and Agriculture Organization of the United Nations

[7] Gensch, R.; Miso, A.; Itchon, G. (2011): Urine as Liquid Fertilizer in Agricultural Production in the Philippines - a practical field guide, Xavier University Press, ISBN 978-971-9094-11-1

[8] Graham, R. D. (2008): Micronutrient deficiencies in crops and their global significance, Micronutrient deficiencies in global crop production B.J Alloway (ed), Chapter 2, Springer Science + Business Media B.V.

[9] Jobbagy, E. G.; Jackson, R. B. (2001): The distribution of soil nutrients with depth: Global patterns and the imprint of plants, Biogeochemistry 53: 51-77, Kluwer Academic Publishers

[10] Karamanos, R. E. Micronutrients - Determining crop responses http://www.umanitoba.ca/faculties/afs/MAC_proceedings/proceedings/2000/mac2000-05.pdf (accessed: June 3, 2015)

[11] Kirchmann, H.; Petterson, S. (1995): Human urine - Chemical composition and fertilizer use efficiency, Fertilizer Research 40: 149-154

[12] Merchant, S. (2010): The elements of plant micronutrients, Future Perspectives in Plant Biology, Plant Physiology, October 2010, Vol. 154, pp. 512-515, American Society of Plant Biologists

[13] Morgan, P.; SEI (2004) The usefulness of urine chapter 10 http://www.ecosanres.org/pdf_files/PM_Report/Chapter_10_The_usefulness_of_urine_a.pdf (accessed: June 3, 2015)

[14] Osotsapar, Y. (1999): Micronutrients in Crop Production in Thailand http://www.fao.org/prods/gap/database/gap/files/1278_MICRONUTRIENTS_THAILAND.PDF (accessed: June 3, 2015)

[15] Palmquist, H.; Jönsson, H. (2004): Urine, faeces, greywater, greywater and biodegradable solid waste as potential fertilizers, Ecosan - closing the loop, Proceedings of the 2nd International Symposium on Ecological Sanitation, Incorporating the 1st IWA Specialist Group Conference on Sustainable Sanitation, 7th - 11th April, Luebeck, Germany: 587-594

[16] Richert, A.; Gensch, R.; Jönsson, H.; Stenström, T.; Dagerskog, L. (2010): Practical guidance on the use of urine in crop production, Stockholm Environment Institute, EcoSanRes Series, 2010-1, ISBN 978-91-86125-21-9

[17] Schönning, C.; Stenström, T.A. (2004): Guidelines on the Safe Use of Urine and Faeces in Ecological Sanitation Systems. (EcoSanRes Publication Series). Stockholm: Stockholm Environment Institute (SEI)

[18] Seregin, I.V.; Kozhevnikova, A.D. (2006): Physiological role of Nickel and its toxic effects on higher plants, Fiziologiya Rastenii, 2006, Vol. 53, N. 2, pp. 285-308, ISSN 1021-4437

[19] Sillanpää, M. (1990). Micronutrient assessment at country level: An international study. Soils Bulletin No. 63, FAO, Rome, 208 pp.

20] Singh, M.V. Emerging micronutrient deficiencies in soils and crops in india and strategies for meeting challenges http://zinc-crops.ionainteractive.com/ZnCrops2007/PDF/2007_zinccrops2007_mv_singh_abstract.pdf (accessed: June 3, 2015)

[21] UNEP (2014) Assessing Global Land Use: Balancing Consumption with Sustainable Supply. A Report of the Working Group on Land and Soils of the International Resource Panel. Bringezu S., Schütz H., Pengue W., O´Brien M., Garcia F., Sims R., Howarth R., Kauppi L., Swilling M., and Herrick J. ISBN: 978-92-807-3330-3

[22] White, J. G.; Zasoski, R. J. (1998): Mapping soil micronutrients, Field Crops research 60 (1999) 11-26, ELSIVIER

TA-P_04 Effect of fertilizer source on honey bees number in black cumin plant

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1. Abstract

Loss of biodiversity is may be occur by different problems such as high and unbalanced application of chemical fertilizers for crop production, that it cause to disorder in agroecosystem functions. The objective of this study was to evaluate honey bees number in black cumin (*Nigella sativa* L.) plant treated with chemical fertilizer, organic manure and their combination, in semi-arid conditions, Iran. The experiment was conducted in randomized complete block with four replications at research farm of Shahrekord University in 2013. Treatments were consisted of three sources of fertilizer (Chemical fertilizer, broiler litter, and integrated fertilizer 50:50. Honey bees number per plot (2.5 × 2.25 m) for black cumin plants was counted in flowering stage (during eight days). The results showed that effect fertilizer source was significant for honey bees number per plot in the first, second and seventh days. Honey bees number per plot were in the following decreasing order; in the first day, integrated= organic manure > chemical fertilizer, in the second day, integrated= organic manure > chemical fertilizer and in the seventh day, integrated≥ organic manure ≥ chemical fertilizer. In the first, second and seventh days, increase honey bees number per plot in integrated and organic treatments compared to chemical fertilizer were 145 and 100 %, 140 and 120 %, 83 and 33%, respectively. In general concluded that the application of organic manure in protecting of pollinators insects can play better than conventional systems. Therefore, more researches is needed to investigate the composition of produced nectar of plant flowers in organic conditions.

2. Introduction

Black cumin (*Nigella sativa*) (family Ranunuculaceae) is emerging as a miracle herb with a rich historical and religious background since many researches revealed its wide spectrum of pharmacological usage (khare, 2004). This plant is found in various countries bordering the Mediterranean Sea, Pakistan, India, and Iran (Jansen, 1981; and Hosseinzadeh and Parvardeh, 2004).

Black cumin is an annual flowering plant which grows to 20-90 cm tall, with finely divided leaves, the leaf segments narrowly linear to threadlike. The flowers are delicate, and usually colored white, yellow, pink, pale blue or pale purple, with 5-10 petals. The fruit is a large and inflated capsule composed of 3-7 follicles, each containing numerous seeds (Coreja, 2003; Warrien and Nambiar, 2004). This plant because of the attractive and colorful flowers, pollinated by insects that honey bee plays an important role.

There are fixed oils (about 30%) and volatile oils (0.5 -1.5%) in seeds of black cumin; this plant is also a rich source of unsaturated fatty acids, amino acids and proteins, carbohydrates, quinones (such as thymoquinone, nigellone, and thymohydroquinone), alkaloids and terpenoids, carvacrol, t-anetholet, crude fiber, and minerals such as calcium, iron, sodium, and potassium (Hosseinzadeh and Parvardeh, 2004; Gali Muhtasib *et al*, 2006). Most of the thorapelutic propertioss of this plant are due to the presence of they theymoquininone (TQ) which is a majar active chemical component of the essential oil. Black cumin seed are also used in food like flavoring additive in the breads and pickles because of it has very low level of toxicity (Al-Ali, 2008).

Black cumin has been extensively studied for its biological activitises and therapeutic potential and shown to prossess wide spectrum of activities viz. as diuretic, antihypertensive, antidiabetic, anticancer and immunomodulatory, analgesic, antimicrobial, anthelmintics, analgesics and anti-inflammatory, spasmolytic bronchodilator, gesroprotective, hepatoprotective, renal protective and axtionidant properties. The seeds of black cumin are widely used in the treatment of various diseases like bronchitis, asthma, diarrheal, rheumatism and skin disorders. It is also used as liver tonic, digestive, anti-diarheal, appetite stimulant emmenagogue, to increase milk production in nursing mothers to fight parasitic infections and to support immune system (Goreja, 2003; Khaled, 2009; Assayed, 2010; Boskabadi *et al*, 2010; Abdol- Zaher, 2011; Abel-Salem, 2012).

Application of fertilizers is one of the major environmental issues in agricultural production. Energy used for the production of chemical fertilizers accounts for around 1.2% of total global energy consumption (IFA, 2009). Nitrogen is provided in the form of synthetic chemical fertilizer (such as urea). Such chemical fertilizers pose a health hazard and microbial population problem in soil besides beings quite expensive and making the cost of production high. In such a situation the organic manure play a major role (Chandrasekar *et al*, 2005).

The use of organic manure are promoted to improve crop yield is without doubt (Mweta *et al*., 2007; Makumba and Akinnifesi, 2008; Kang *et al*., 2009). The application of organic manures (solitary or combined application with chemical fertilizer) provide more advantages over mineral fertilizers because of improvements in soil structure, aggregate stability, soil nutrient exchange capacity, water holding capacity, soil bulk density, microbial biomass and activity, and crop yields (Haynes and Naidu, 1998; Barzegar *et al*., 2002; Manna *et al*., 2007). Researchers reason of benefits organic manure known resulting from the more match the nitrogen accessible nitrogen and needs plant (Alizadeh *et al*., 12). On the other hand organic manure can be increase height, essential oil, and etc., followed increase aggregation honey bees that may be effective in increasing and protection diversity, but in this case has not been studied in the world. Therefore, the purpose of this study is effect application of different fertilizer (chemical, integrated and organic manure) on honey bees number of black cumin.

3. Materials and methods

The experiment was conducted in randomized complete block with three replications at research farm of Shahrekord University in 2013. The site lies at longitude 50° 49, and latitude 32° 21 and the altitude of the area is 2050 m above sea level. It has a cold dry climate with the mean minimum, and maximum air temperatures of 2.9, and 34.8°C, respectively. The characteristics of soil and organic manure is shown in Table 1).

Treatments were consisted of three sources of fertilizer (Chemical fertilizer, broiler litter, and integrated fertilizer 50:50. All treatments were applied before sowing. Black cumin was planted manually in 17 may 2013. Experiment plots were seeded with Sistan landrace with 25 cm row to row distance and 4 cm between plants. Seeds were sown 2 cm depth. Weeds were removed by hand. After planting, irrigation was applied as required during the growing season.

Honey bees number per plot (2.5 × 2.25 m) for black cumin plants was counted in flowering stage (during eight days).

Tabel 1: Chemical analysis of soil and organic manure experiment

Property	Organic manure	Soil
Texture	**-**	**Cly-loam**
EC (dS/m)	6.23	1.01
pH	7.91	7.96
N (g/kg)	21.1	0.82
P (g/kg)	3.8	0.0108
K (g/kg)	18.6	0.391
Mn (mg/kg)	47.11	8.73
Zn (mg/kg)	23.14	0.68
Fe (mg/kg)	256	8.09
Cu (mg/kg)	21.15	0.91

4. Result and Discussion

Effect of fertilizer source on honey bees number are presented in Table 2. Effect fertilizer source was significant for honey bees number per plot in the first, second and seventh days (Table 2). While the effect of fertilizer source on honey bees number per plot in the third, foruth, fifth, sixth, and eighth day was not significant (Table 2).
Honey bees number per plot were in the following decreasing order; in the first day, integrated= organic manure > chemical fertilizer, in the second day, integrated= organic manure > chemical fertilizer and in the seventh day, integrated≥ organic manure ≥ chemical fertilizer (Table 3).

Table 2: Effects of fertilizer source on honey bees number of black cumin.

S.O.V	Df	Day 1	Day 2	Day 3	Day 4	Day 5	Day 6	Day 7	Day 8
Replication	2	10.1^{ns}	11.1^{ns}	8.3^{ns}	5.44^{ns}	2.7^{ns}	7.1^{ns}	2.7^{ns}	11.1^{ns}
Fertilizer source	2	73.4^{*}	119.4^{**}	25^{ns}	2.1^{ns}	2.7^{ns}	42.1^{ns}	52.7^{*}	19.4^{ns}
Error	4	8.4	2.7	8.3	8.7	11.1	12.1	6.9	11.1
CV (%)	-	23.9	10.7	17.3	16.9	18.7	21.9	18.9	19.3

ns, * and ** indicate are non-significant, significant at the 5 and 1% probability level, respectively.

In the first, second and seventh days, increase honey bees number per plot in integrated and organic treatments compared to chemical fertilizer were 145 and 100 %, 140 and 120 %, 83 and

Table 3: Effect fertilizer source on honey bee number of black cumin during the flowering stage

Day 1	Day 2	Day 3	Day 4	Day 5	Day 6	Day 7	Day 8	Treatment
6.66b	8.33b	20a	17.33a	16.67a	15a	10b	16.66a	Chemical fertilizer
16.33a	20a	15a	16.66a	18.33a	20a	18.33a	20a	Integrated fertilizer
13.33a	18.33a	15a	18.33a	18.33a	12.66a	13.33ab	15a	Broiler litter

Mean with the same letter are not significantly different.

33%, respectively. Nevertheless highest honey bees number per plot in the third, fourth, fifth, sixth, and eighth day was observed in chemical fertilizer, organic manure, organic manure and integrated, integrated and integrated sources, respectively. The source integrated amendment (organic manure+ chemical fertilizer) has the greatest effect on honey bee number diversity and maintenance.

More presence of honey bee in organic systems, may be due to increase the height and essential oil of medicine black cumin plants. As in this study, the maximum plant height and essential oil contain (*Black cumin*) were observed in the Integrated system (data not presented). In this regard Abou El-Magd et al., (2008) reported the application organic manure compared with no fertilizer increased significantly plant height of *(Foeniculum vulgare* mill). Salehi (2013) reported the maximum essential oil contain of *N. sativa* Was obtained in Integrated system (organic manure + chemical fertilizer).
In general it can be concluded that the application of organic manure in protecting of pollinators insects can play better than conventional systems. Therefore, more researches is needed to investigate the composition of produced nectar of plant flowers in organic conditions.

Figure 1: Images of plants

References

[1] Abdel-Zaher, A.O., Abdel-Rahman, M.S., and Elwasei, F.M. (2011): Protective effect of Nigella sativa oil against tramadol-induced tolerance and dependence in mice: role of nitric oxide and oxidative stress **.**Neurotoxicology, 32(6), 725-733.

[2] Abel-Salam, B.K. (2012): Immunomodulatory effects of black seeds and garlic on alloxan-induced diabetes diabetes in albino rat. Allergol Immunopathol (Madr), 40(6),336-340.

[3] Abou El-Magd, M.M., Zaki, M.F., and Abou-Hussein, S.D. (2008): Effect of organic manure and different levels of saline irrigation water on growth, green yield and chemical content of sweet fennel. Australian Journal of Basic and Applied Sciences, 2(1),90-98.

[4] Al-Ali, A., Alkhawajah, A.A., Randhawa, M.A., Shaikh, N.A. (2008): Oral and intraperitoneal LD50 of thymoquinone, an active principle of Nigella sativa, in mice and rats. J Ayub Med Coll Abbottabad, 20(2), 25-27.

[5] Alizadeh, P., Fallah, S., and Raiesi, F. (2012): Potential N mineralization and availability to irrigated maize in a calcareous soil amended with organic manures and urea under field conditions. International Journal of Plant Production, 6(4),493-512.

[6] Assayed, M.E.(2010): Radioprotective effects of black seed (Nigella sativa) oil against hemopoietic damage and immunosuppression in gamma-irradiated rats. Immunopharmacology and Immunotoxicology, 32 (2), 284-296.

[7] Barzegar, A.R., Yousefi, A., Daryashenas, A. (2002): The effect of addition of different amounts and types of organic materials on soil physical properties and yield of wheat. Plant and Soil, 47(2), 295-301.

[8] Boskabady M.H., Mohsenpoor, N., Takaloo, L. (2010): Antiasthmatic effect of Nigella sativa in airways of asthmatic patients. Phytomedicine, 17(10), 707-713.

[9] Chandrasekar, B.R., Ambrose, G., and Jayabalan, N. (2005): Influence of biofertilizers and nitrogen source level on the growth and yield of *Echinochloa frumentacea* (Roxb) Link. Journal of Agricultural Technology, 1(2), 223-234.

[10] Gali-Muhtasib, H., El-Najjar, N., Schneider-Stock, R. (2006): The medicinal potential of black seed (*Nigella sativa*) and its components. Advances in Phytomedicine, 2,133-153.

[11] Goreja WG.2003. Black seed: nature's miracle remedy. New York, NY 7 Amazing Herbs Press.

[12] Haynes, R.J., Naidu, R. (1998): Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. *Nutrient Cycling in Agroecosystems,* 51 (2), 123-137.

[13] Hosseinzadeh, H.,Parvardeh, S. (2004): Anticonvulsant effects of thymoquinone, the major constituent of Nigella sativa seeds, in mice. Phytomedicine, 11(1),56-64.

[14] IFA (International Fertilizer Association). Energy efficiency and CO2 emissions in ammonia production. 2008–2009 summary report; September 2009.

[15] Jansen, P.C.M. (1981): Spices, Condiments and Medicinal Plants in Ethiopia, Their Taxonomy and Agricultural Significance. Addis Ababa: Center for Agricultural Publishing and Documentation.

[16] Kang, H., Shannon, D.A., Prior, S.A., Arriaga, F.J. (2009): Hedgerow pruning effect on light interception, water relations and yield in Alley-cropped maize. *Journal of Sustainable Agriculture,* 31(4) 115–137.

[17] Khaled, A.A.S. (2009): Gastroprotective effects of Nigella Sativa oil on the formation of stress gastritis in hypothyroidal rats. International Journal of Physiology Pathophysiology Pharmacology, 1(2), 143-149.

[18] Khare, C.P. (2004): Encyclopedia of Indian medicinal plants. *Springes-Verlag Berlin Heidelberg, New York*.

[19] Makumba, W., Akinnifesi, F.K., Janssen, B. (2007): Long-term impact of a Gliricidiamaize intercropping system on carbon sequestration in southern Malawi. *Agriculture, ecosystems & environment,* 118(1), 237–243

[20] Manna, M.C., Swarup, A., Wanjari, R.H., Mishra, B., Shahi, D.K. (2007): Long-term fertilization, manure and liming effects on soil organic matter and crop yields. *Soil and Tillage Research,* 94(2), 397-409.

[21] Mweta, D.E., Akinnifesi, F.K., Saka, J.D.K. (2007): Green manure from prunings and mineral fertilizer affect phosphorus adsorption and uptake by maize crop in a Gliricidia-maize intercropping. *Scientific Research and Essays*, 2, 446–453.

[22] Salehi, A. (2013): Effect of soility and interated application of cattle manure and urea fertilizer on soil CO2 flux, growht and yield of black cumin (*Nigeilla sativa* L.). MSc. thesis of Agroecology, Agriculture College, Shahrekord University.

[23] Warrier, P.K., Nambiar, V.P.K., Ramankutty (2004): Indian medicinal plants-a compendium of 500 species. Chennai: Orient Longman Pvt Ltd; p. 139-142.

TA-P_05 Agronomic assessment of the fertilizing capacity of agroindustrial waste composts in Chimborazo province (Ecuador)

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1. Objectives

The aim of this experiment was to determine the effects of different agroindustrial waste composts on the soil properties and on plant yield and some marketable parameters of two successive horticultural crops.

2. Methodology

2.1 Materials and experimental design

The field experiment was conducted at the Tunshi Research Station of the Polytechnic School of Chimborazo (Riobamba, Ecuador) (01°45′0″S, 78°30′38″W and elevation 2,829 m above sea level). The climate of this region is temperate, with an average cumulative annual precipitation of 475 mm and an average annual temperature of 13.8 °C. The main characteristics of the soil were: loam texture, pH 7.2, electrical conductivity (EC) (1:5) 0.44 dS m^{-1}, organic matter (OM) 1.04%, total nitrogen (N_t) 500 mg kg^{-1}, available-P (P_{ava}) 13.9 mg kg^{-1} and available-K (K_{ava}) 241 mg kg^{-1}.

Six treatments, in a completely randomised design with three replicates per treatment, were set up in experimental plots of 6 m^2 each, with 0.5 m distance between plots. The treatments were: control without amendment (C); mineral fertiliser (175, 60 and 200 kg ha^{-1} of N, P_2O_5 and K_2O, respectively) (M); poultry manure (PM) (9.7 t ha^{-1}), a traditional amendment used as a reference in this study; compost comprising flower waste (vegetable waste from rose postharvest, made up of stems, leaves and non-marketable flowers), poultry manure and sawdust (C1) (9.3 t ha^{-1}); compost comprising broccoli waste, poultry manure and sawdust (C2) (28 t ha^{-1}), and compost comprising tomato waste (vegetable waste from postharvest, consisting of stems, leaves and non-marketable tomatoes), poultry manure and sawdust (C3) (16.7 t ha^{-1}). These composts were prepared in the following proportions, on a fresh weight basis: 50% vegetable waste + 15% poultry manure + 35% sawdust. The main characteristics of these organic materials are shown in Table 1. The amendment application rate was adjusted to supply 175 kg ha^{-1} of nitrogen. The organic materials were applied uniformly by hand and immediately incorporated to a soil depth of 25 cm, by light roto-tilling. The unamended plots and those receiving inorganic fertiliser were also tilled. After the incorporation of these treatments, two successive crops of broccoli (*Brassica oleracea* cv. Avenger) and lettuce (*Lactuca sativa L.* cv. capitata) were carried out. Seedlings of uniform size of both horticultural plants were selected and planted in each plot, leading to a plant density of 35,000 and 40,000 plants ha^{-1} for broccoli and lettuce, respectively. Periodic irrigation with water was given during the growing seasons and treatments of insecticide were applied when necessary.

Table 1: Characteristics of the organic materials used in the experiment.

Parameter*	PM	C1	C2	C3
pH	8.2	8.2	8.4	8.9
Electrical conductivity (dS m^{-1})	8.08	2.99	2.50	3.52
Organic matter (%)	38.0	54.6	24.1	31.9
Total organic C (%)	23.7	31.7	16.3	28.6
Total N (%)	2.33	2.80	0.92	1.60
Total organic C/Total N	10.2	11.3	17.7	17.9
P (g kg^{-1})	14.6	10.2	4.82	6.01
K (g kg^{-1})	38.4	20.5	12.8	19.7

*Values on a dry matter basis.
PM: poultry manure; C1: compost of flower waste +poultry manure + sawdust; C2: compost of broccoli waste +poultry manure + sawdust; C3: compost of tomato waste +poultry manure + sawdust

The timing of the experiment was as follows (Table 2):

Table 2: The timing of the experiment.

Date	Operation
15 April 2014	Addition of soil amendments and of the mineral fertiliser and first soil sampling (S1)
21 April 2014	Transplanting of broccoli seedlings
29 July 2014	Harvesting of broccoli and soil sampling (S2)
31 July 2014	Light rototilling for the homogeneous mixing of soil with the roots of broccoli, transplanting of lettuce seedlings and soil sampling (S3)
13 October 2014	Harvesting of lettuce and soil sampling (S4)

The cropping period from transplanting to the end of harvesting was about 99 and 74 days for broccoli and lettuce, respectively.

2.2 Analytical and statistical methods

All plants of each of the treatment plots were harvested and weighed and counted to determine the total plant yield and the vegetable weight. The marketable parameters in both horticultural crops were determined by the following methods:

- Method 1: The diameter of broccoli and lettuce was measured using calipers.
- Method 2: The degree of compactness of broccoli and lettuce was calculated by dividing the vegetable diameter between its weight.
- Method 3: The colour of broccoli was evaluated according to the following criteria: 1 (yellow) to 5 (green).

All the soil samples were produced by mixing six sub-samples, from six sites of each plot, taken at 0-25 cm depth. Each soil sample was sieved to 2 mm, after removal of vegetation and bigger roots and stones, and the granulometric fraction was air-dried and used for analysis. All the analytical determinations in the manure, compost and soil were determined according to the methods described by Paredes et al. [1]. All analyses were made in triplicate.
The data were tested for statistically significant differences using one-way analysis of variance (ANOVA), considering the type of treatment. Comparison of the means of these parameters was performed by the Tukey-b test at $P < 0.05$. Data analysis was carried out using the SPSS v. 22.0 statistical software packages.

3. Results and discussion

3.1 Effect of the treatments on physico-chemical soil properties

The soil initial pH was only affected by the mineral fertiliser (Table 3). This treatment reduced the soil pH probably due to the soil acidification caused by the nitrogen fertiliser. Makinde et al. [2] also observed a soil acidification after the application of NPK in two soil types in Nigeria (Orthic Luvisol and Dystric Fuvisol). However, the treatments with compost increased the soil pH at S2 and S3, this result being also found by Guénón and Gros [3] in a study of burned soil recovery by the addition of compost of municipal sewage sludge + pin barks + greenwastes with different maturity degree. At the end of the experimental period, great differences in the pH values were not found due to the treatment.

Organically and inorganically amended plots had higher EC values than the control at S1 (Table 3). This parameter diminished in all soils during the experimental period, possibly due to nutrient uptake by the crop, ion leaching and inorganic nitrogen immobilisation. Bustamante et al. [4] also observed a decrease of soil EC throughout a long-term experiment investigating the effects of the addition of agroindustrial composts and sheep manure to a calcareous vineyard soil. However, only M, PM and C1 soils reached EC final values statistically similar to those of the control soil. Soil EC values were clearly higher in the soils with C2 and C3 treatments, possibly due to their high application doses, as a consequence of their low nitrogen content.

PM and C3 addition to soil did not result in any significant initial effect on OM with respect to the control and mineral fertiliser, while the all organic fertilisers resulted in statistically significant increases in this parameter at S2 (Table 3). This fact indicated that the organic matter provided with the studied amendments was not easily degradable in the short term. Similar results were obtained by other authors in different studies about the effects of organic waste or compost on soil properties after a cropping period [1, 5]. Before and after lettuce cropping, no differences were found in the organic matter concentration of the soil due to the treatment effect. This fact could be due to the mixture of soil with the roots of broccoli carried out in all plots.

The soils treated with compost or poultry manure had the highest concentration of total N (N_t) after the addition of these organic amendments and after harvesting broccoli (S1 and S2) (Table 3). Increases of the nitrogen concentrations in the soil due to organic fertilisation were observed also by Paredes et al. [1], in a study of the effects of different olive mill wastewater sludge compost doses on soil properties. Later, only plots treated with C3 showed significantly higher N_t concentrations than the other treatments (S3 and S4). At S3, the C and M soils presented an increase in the N_t content possibly due to the organic matter incorporated with the mixture of soil with the roots of broccoli carried out in all plots.

Table 3: Physico-chemical characteristics of the soil according to the fertiliser treatments and samplings (dry weight basis). Sampling time S1: after the first application of organic materials and before broccoli planting, S2: after broccoli cropping, S3: before lettuce planting and S4: after lettuce cultivation.

Parameter	Sampling	C	M	PM	C1	C2	C3	ANOVA
pH	S1	7.2 b	6.7 a	7.4 b	7.3 b	7.3 b	7.2 b	***
	S2	7.3 ab	7.1 a	7.3 ab	7.4 bc	7.5 bc	7.7 c	**
	S3	7.4 abc	7.1 a	7.2 ab	7.5 bc	7.4 abc	7.7 c	**
	S4	7.3 ab	7.1 a	7.2 ab	7.5 ab	7.7 b	7.5 ab	*
EC (dS m^{-1})	S1	0.44 a	2.87 e	1.96 d	1.03 b	2.26 d	1.54 c	***
	S2	0.28 a	0.26 a	0.44b	0.41 ab	0.63 c	0.63 c	***
	S3	0.36 ab	0.28 a	0.49 bcd	0.41 abc	0.54 cd	0.60 d	***
	S4	0.31 a	0.30 a	0.35 ab	0.45 abc	0.56 c	0.50 bc	**
OM (%)	S1	1.04 a	1.08 a	1.35 ab	1.45 b	2.39 c	1.22 ab	***
	S2	0.40 a	0.37 a	0.88 b	0.82 b	0.79 b	0.77 b	***
	S3	0.92	0.80	0.83	0.89	1.03	0.80	NS
	S4	0.97	0.93	0.96	0.87	1.11	1.11	NS
N_t (mg kg^{-1})	S1	500 a	633 ab	700 b	733 b	767 b	700 b	**
	S2	199 a	100 a	447 b	401 b	497 b	700 c	***
	S3	462 a	402 a	415 a	445 a	514 a	700 b	**
	S4	453 a	433 a	458 a	467 a	567 ab	700 b	**
P_{ava} (mg kg^{-1})	S1	13.9 a	38.8 b	105.1 d	99.6 cd	81.2 c	55.7 b	***
	S2	10.7 a	15.6 a	59.8 c	34.8 b	62.4 c	69.1 c	***
	S3	10.2 a	15.7 a	68.7 c	40.9 b	52.1 bc	65.6 c	***
	S4	10.1 a	16.3 a	50.7 bc	43.5 b	63.6 c	64.5 c	***
K_{ava} (mg kg^{-1})	S1	241 a	484 b	420 b	392 b	360 b	368 b	**
	S2	262 a	340 abc	325 abc	310 ab	432 c	414 bc	**
	S3	203 a	300 bc	288 b	334 bc	378 c	382 c	***
	S4	192 a	293 b	215 ab	271 ab	309 b	264 ab	*

C: control; M: mineral fertiliser; PM: poultry manure; C1: compost of flower waste +poultry manure + sawdust; C2: compost of broccoli waste +poultry manure + sawdust; C3: compost of tomato waste +poultry manure + sawdust; EC: electrical conductivity; OM: organic matter; N_t: total nitrogen; P_{ava}: available-P; K_{ava}: available-K.

***, **, * and NS: significant at $P < 0.001$, 0.01, 0.05 and not significant, respectively.

Values in row followed by the same letter are not statistically different according to the Tukey's *b* test at $P < 0.05$.

As table 3 shows, at S1, the addition of mineral and organic fertilisers increased available P (P_{ava}) contents in the soil. However, only the amended plots with PM or compost had higher concentrations of P_{ava} than the control in the rest of the samplings, showing that these organic amendments had a higher P fertilising capacity than treatment M. This available P evolution suggests that the gradual mineralisation of organic P compensates the mineral P content, which is gradually lost through absorption by the crop [5].

M and amended plots had higher concentrations of available K (K_{ava}) than the control in all samplings, although the differences were not statistically significant for some samplings (Table 3). The soils with C2 and C3 treatments presented the highest K_{ava} contents at S2 and S3. At the end of the experimental period, great differences in the available-K concentrations were not found due to the treatment.

3.2 Effect of treatments on crop quality and production

The marketable yields of broccoli obtained with M, PM, C1 and C2 treatments were higher than for the compost 3 and control (Table 4); whereas, the non-marketable production (the leafs of broccoli plants) was higher with PM, C1 and C2 treatments. Non statistic differences were found in the values of curd weight, diameter colour and compactness degree of broccoli due to the treatment. Production results for lettuce and vegetable weight values were higher in the soil with organic amendments (Table 5). This fact indicated that the studied organic fertilisers had a higher residual fertilising effect than the mineral fertiliser. Great differences in other marketable parameters of lettuce were not found due to the treatment. Therefore, these organic amendments can enhance soil fertility and productivity, improving the plant nutrient status for potentially limiting nutrients such as N, P and K as well as for several micronutrients. These positive effects of the organic fertilisers on the yield of the horticultural crops were also observed by Alburquerque et al. [6] in an experiment to evaluate the fertilising capacity of a digestate using watermelon and cauliflower crops.

Table 4: The main effects of the tested fertiliser treatments on marketable parameters of broccoli (fresh weight basis).

Treatment	Marketable production ($Mg\ ha^{-1}$)	Non-marketable production ($Mg\ ha^{-1}$)	Curd weight ($kg\ piece^{-1}$)	Diameter (cm)	Colour	Compactness degree ($cm\ kg^{-1}$)
C	11.3 a	12.0 a	0.38	13.3	4.9	36.5
M	19.6 bc	15.8 a	0.53	16.6	4.8	31.5
PM	20.0 bc	21.1 b	0.55	17.5	4.7	32.7
C1	18.6 b	24.5 b	0.55	13.8	5.0	25.0
C2	24.5 c	23.7 b	0.63	15.7	5.0	25.4
C3	15.5 ab	14.4 a	0.48	15.4	5.0	33.2
ANOVA	***	***	NS	NS	NS	NS

C: control; M: mineral fertiliser; PM: poultry manure; C1: compost of flower waste +poultry manure + sawdust; C2: compost of broccoli waste +poultry manure + sawdust; C3: compost of tomato waste +poultry manure + sawdust.
** and NS: significant at $P < 0.01$ and not significant, respectively.
Values in column followed by the same letter are not statistically different according to the Tukey's *b* test at $P < 0.05$.

Table 5: The main effects of the tested fertiliser treatments on marketable parameters of lettuce (fresh weight basis).

Treatment	Production ($Mg\ ha^{-1}$)	Vegetable weight ($kg\ piece^{-1}$)	Diameter (cm)	Compactness degree ($cm\ kg^{-1}$)
C	10.5 a	0.30 a	7.7 a	25.7 b
M	13.5 ab	0.41 ab	9.6 ab	23.8 ab
PM	16.6 bc	0.48 b	10.6 b	22.2 ab
C1	16.1 bc	0.47 b	10.1 ab	21.3 ab
C2	19.8 c	0.57 b	10.9 b	19.6 a
C3	18.5 bc	0.54 b	10.5 b	19.6 a
ANOVA	**	**	*	*

C: control; M: mineral fertiliser; PM: poultry manure; C1: compost of flower waste +poultry manure + sawdust; C2: compost of broccoli waste +poultry manure + sawdust; C3: compost of tomato waste +poultry manure + sawdust.
***, ** and * : significant at $P < 0.001$, 0.01 and 0.05, respectively.
Values in column followed by the same letter are not statistically different according to the Tukey's *b* test at $P < 0.05$.

4. Conclusion and outlook

From the data obtained, it can be concluded that the application of the agroindustrial waste composts to soil produced positive effects on soil fertility and these composts did not lead to phytotoxic effects on the broccoli and lettuce plants.

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References

[1] Paredes, C.; Cegarra, J.; Bernal, M.P.; Roig, A. (2005): Influence of olive mill wastewater in composting and impact of the compost on Swiss chard crop and soil properties, Environment International 31, 305-312.

[2] Makinde, E.A.; Eniola, H.T.; Fagbola O. (2009): Effect of organic, organomineral and NPK fertilizers on soil pH, organic matter and micronutrient content in two soil types in Nigeria, Research on Crops 10, 77-85.

[3] Guénon, R.; Gros, R. (2015): Increasing the maturity of compost used affects the soil chemical properties and the stability of microbial activity along a mediterranean post-fire chronosequence, European Journal of Soil Biology 66, 1-10.

[4] Bustamante, M.A.; Said-Pullicino, D.; Paredes, C.; Moral, R. (2011): Application of winery and distillery waste composts to a Jumilla (SE Spain) vineyard: Effects on the characteristics of a calcareous sandy-loam soil, Agriculture, Ecosystems and Environment 140, 80-87.

[5] Medina, E.; Paredes, C.; Bustamante, M.A.; Moral, R.; Moreno-Caselles, J. (2012): Relationships between soil physico-chemical, chemical and biological properties in a soil amended with spent mushroom substrate, Geoderma 173-174, 152-161.

[6] Alburquerque, J.A.; de la Fuente, C.; Campoy, M.; Carrasco, L.; Nájera, I.; Baixauli, C.; Caravaca, F.; Roldán, A.; Cegarra, J.; Bernal, M.P. (2012): Agricultural use of digestate for horticultural crop production and improvement of soil properties, European Journal of Agronomy 43, 119-128.

TA-P_06 The influence of anaerobic digestion on the concentration of antibiotics, heavy metals and on phosphorous-solubility of digestates

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1. Objectives

Digestates are a nutrient rich source, especially with view to the finite resource phosphorus (P). Manure and sewage sludge are often used as co-substrates in biogas plants, thus digestates may be contaminated with antibiotics [1] and heavy metals [2]. Different substrates and digestates were analysed to assess the effect of digestion on the content of selected antibiotics, heavy metals and P-solubility.

2. Methodology

Several methods were used to analyse the different parameters in the substrates and digestates. The P-solubility was investigated by extracting substrates and digestates with CAL (calcium-ammonia-lactate) and water followed by the colorimetric determination of the P content. The antibiotics were quantified after liquid extraction and purification with solid phase extraction (SPE). Heavy metals were dissociated by *aqua regia* and determined by ICP-MS.

- The substrates and digestates were sampled in a commercial biogas plant. The anaerobic digestion of the substrates (corn and poultry manure) was conducted at 43 °C in two steps in a fermenter and a following post digester, from which the digestate was sampled.
- CAL/water extraction ([3],[4]adapted): The equivalent of 0.5 g dry matter of the fresh, homogenized material was extracted with 100 mL water or CAL solution and was shaken for 30 (water) or 60 minutes (CAL). The P content was determined as antimony-phosphorus-molybdenum-complex by colorimetric determination [5] in 4 replicates.
- *Aqua regia* ([6],adapted): 1 g dry matter was dissociated with 25 mL *aqua regia* and boiled under reflux for 2 hours. The concentration of heavy metals was determined by ICP-MS.
- Extraction of antibiotics/SPE ([7], adapted): Antibiotics were extracted by a citric acid buffer at pH 4.7 and by methanol citric acid (80/20, v/v, pH 3.5). After liquid-liquid extraction with heptane for the reduction of liposoluble pollutants, the extract was adjusted to pH 3 with formic acid and diluted with water. Afterwards, a SPE was performed, with a strong anion exchange phase to filter out matrix components like humic acids and a reversed phase to bind the antibiotics. After washing the reversed phase cartridge with water, the antibiotics were eluted in 2 mL methanol. The quantification of selected tetracyclines, fluorquinolones and sulfonamides was done by standard addition employing LC-MS.

3. Results and discussion

3.1 P-solubility

The total P content and the P solubility of the digestate are key parameters for the calculation of reasonable application rates to prevent excess fertilisation as well as eutrophication.
The P-solubility of substrates and digestates of a biogas plant was analysed in order to determine if the solubility changes during anaerobic digestion. Samples were taken from the basic substrates (corn, sugar beet and chicken manure), through the process steps (fermenter, post digester) and from the end product (digestates). The sampled biogas plant used a hydraulic press to separate digestates in liquid and solid phase at the end of the process. For this reason 3 different types of digestates were analysed, the digestate before separation and the liquid and solid phase after separation.

As shown in Figure 1 the substrates corn and sugar beet have low contents of water-soluble P compared to the samples of the fermenter, post digester and digestates. The chicken manure shows slightly higher contents than the digestate, but only accounts for one third of the amount of the input materials (Table 1). Mixing of the materials during the process is no explanation for an increase in the water-soluble P content. An increase was observed between substrates and the first fermentation step, while there was only a minor increase during the process steps towards the digestate. The liquid digestate showed slightly more water-soluble P than the solid digestate, because the liquid phase partly extracts the solid phase. The data suggests that during the anaerobic digestion process the P water-solubility of the substrates is increased.

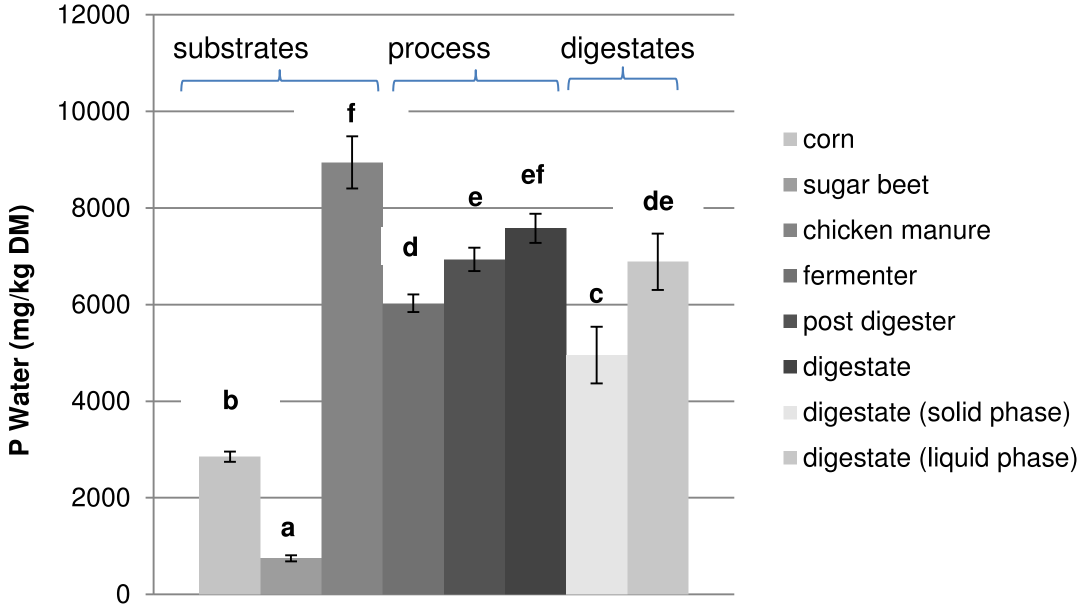

Figure 4:P water solubility of original substrates and digestates of a biogas plant (DM-dry matter, n=4 (different letters denote significant differences between groups as tested by Tukey post-hoc test ($P<0.05$))

Table 7: Composition of the substrate mix fed into the fermenter

Substrate	% of the input mixture
Corn	56.8
Chicken manure	29.5
Sugar beet	13.6

Additionally to the water extraction a CAL extraction was performed, which extracts bound P fractions which are plant available. Figure 2 shows the CAL-soluble P content of the samples, which is distinctly lower in the substrates than in the digestates. In contrast to the water extractable P, the CAL-soluble P increases significantly during the fermentation process from the fermenter towards the digestate.

The anaerobic digestion seems to lead to an increased formation of water-soluble P-forms early in the fermentation process, while the P-forms which are not soluble in water but in CAL, occur during the process steps to the digestate. A higher plant availability of P improves the efficiency of this type of organic fertiliser.

The water-soluble P contents of the original substrates are higher than the CAL-soluble P concentration. This finding has been unexpected as, the acidic CAL solution is supposed to additionally dissolve P fractions which are not water-soluble. One explanation for this phenomenon might be an interference of the antimony-phosphorus-molybdenum-complex during the colorimetrical analysis especially in samples rich in organic matter resulting in cloudy water extracts. In contrast to the substrate, the samples taken during the fermentation process and the end product show, as expected, higher CAL-soluble than water-soluble P contents.

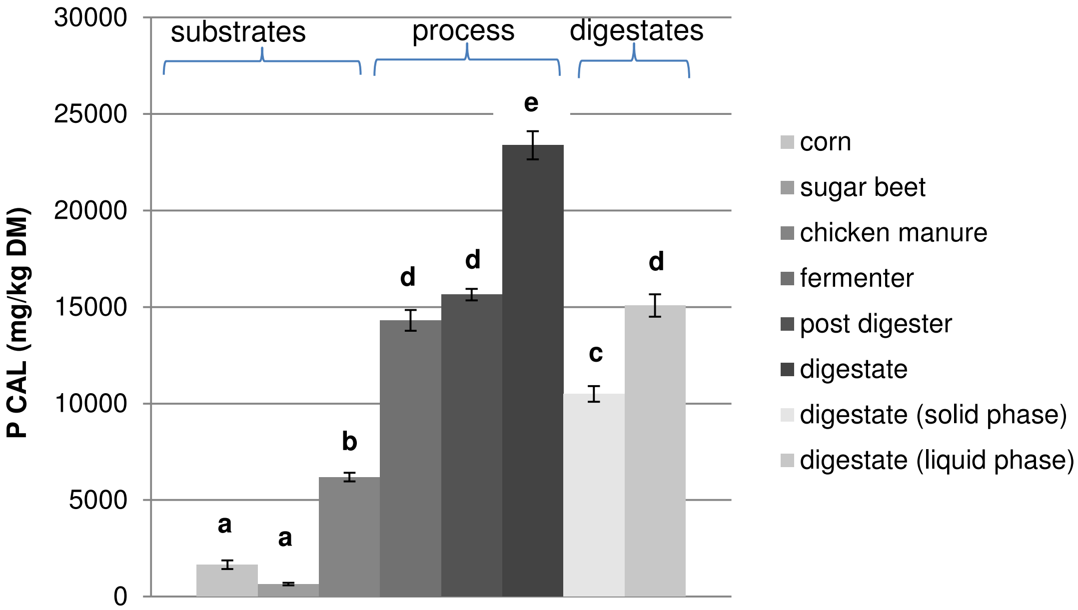

Figure 5: P CAL solubility of fresh substrates and digestates of a biogas plant, DM-dry matter, n=4 (different letters denote significant differences between groups as tested by Tukey post-hoc test (P<0.05))

3.2 Heavy metals

The heavy metal content (As, Pb, Cd, Cr, Ni,Tl) were far below the threshold values set in the German Fertiliser Ordinance (DüMV) [8]. It is not expected that there is a risk for accumulation of heavy metals in the soil when digestates are applied to the field.

3.3 Antibiotics in substrates and digestates of biogas plants

The number of biogas plants in Germany has been steadily increasing during the last decade to about 8,000 biogas plants. These produce large amounts of digestates that are used as organic fertilisers. Besides energy plants such as corn, manure from poultry, pig or cattle are used as co-substrates. In intensive livestock farming, antibiotics are regularly used and residues or metabolites of the antibiotics are excreted and end up in the animal manure [9]. Some antibiotics, especially tetracyclines, fluorquinolones and sulfonamides, are not degraded during the digestion process and can still be found in the digestate [1]. Residues of antibiotics that are applied to the soil via manure or digestate application can affect the soil fauna [10] and enhance the formation of microbial resistance to antibiotics [11].

To determine the influence of digestion on the degradation of antibiotics, compounds of three classes, tetracyclines, fluorquinolones and sulfonamides, were analysed in both original substrates and digestate by the employed method (Table 2). Fluorquinolones were found in 4 of the 6 tested samples, in one sample even two antibiotics were detected at once. The concentrations are in the same order of magnitude as observed by Ratsak and co-workers [1]. In contrast, Sulfonamides and tetracyclines were not detected in this batch of samples, while in literature the occurrence of these antibiotics in digestates was proven (e.g. [1]). The studied biogas plant was sampled twice in a time span of 4 months. While chicken and turkey manure was used as co-substrate on the first sampling date, only chicken manure was used as co-substrate on the second sampling date. Since the retention time of the substrate in a continuously working biogas plant is about 4-6 weeks and new substrate is steadily mixed with partially digested substrate in the fermenter, it is not possible to derive degradation rates of antibiotics by comparing input and output materials.

Preliminary results from 5 further biogas plants show that tetracyclines, fluorquinolones and sulphonamides can be detected in the digestates indicating that there is no complete degradation during the fermentation process. Each digestate was at least contaminated by one antibiotic, but up to five different antibiotics were also found in one digestate. 5 of 6 analysed manures contained antibiotics, and up to five different antibiotics were detected in one sample of turkey manure.

Table 8: Content of 8 antibiotics in substrates* and digestates (µg/kg DM) of a biogas plant on two sampling dates (July and December 2014). (Double determination, standard deviation in brackets)

Antibiotic	Chicken manure (July 2014)	Turkey manure (July 2014)	Digestate (July 2014)	Chicken manure (December 2014)	Fermenter (December 2014)	Digestate (December 2014)
Sulfadiazine	-	-	-	-	-	-
Sulfamethazine	-	-	-	-	-	-
Ciprofloxacin	-	202.9 (15.1)	-	-	-	-
Enrofloxacin	-	-	159.8 (67.9)	-	185.1 (2.2)	140.0 (11.6)
Difloxacin	-	-	-	-	17.8 (4.2)	-
Tetracycline	-	-	-	-	-	-
Oxytetracycline	-	-	-	-	-	-
Chlorotetracycline	-	-	-	-	-	-

* the substrate corn was not analysed

4. Conclusion and outlook

Anaerobic digestion does not have a negative influence on the P-availability of digestates in comparison to the substrates. On the contrary, CAL and water-soluble P even increases during the digestion process indicating that the end product of the digestion process contains higher amounts of plant available P-forms than the substrates. Furthermore, the heavy metal concentration of the analysed digestates does not exceed the limit values of the German Fertiliser Ordinance which also suggests that digestates are suitable fertilisers. However, the detectability of veterinary antibiotics in digestates indicates that further processing of these materials is required to provide a safe nutrient source. Further studies are necessary to evaluate the fate of different antibiotics and metabolites during the digestion process and to evaluate the risks related to the application of such materials in agriculture.

Acknowledgements

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References

[1] Ratsak C.; Guhl B.; Zühlke, S.; Delschen T. (2013) Veterinary antibiotic residues in manure and digestates in Northrhein-Westfalia, Environmental Sciences Europe, 25 (7), 1-22
[2] Zirkler D.; Peters A.; Kaupenjohann M. (2014) Elemental composition of biogas residues: Variability and alteration during anaerobic digestion. Biomass and Bioenergy 67, 89-98
[3] Schüller, H. (1969) Die CAL-Methode, eine neue Methode zur Bestimmung des pflanzenverfügbaren Phosphates in Böden. Zeitschrift für Pflanzenernährung und Bodenkunde 123: 48–63
[4] Van der Paauw, F., Sissingh, H. (1971) Een verbeterede methode van fosfaatextractie van grond met water: het Pw-getal; with a summary: An improved method of water extraction for the assessment of availability of soil phosphate: PW value. Verslagen van landbouwkundige onderzoekingen, Vol. 749
[5] John. M. K. (1970). Colorimetric determination of phosphorus in soil and plant materials with ascorbic acid. Soil Science 109: 214-220.
[6] VDLUFA Methodenbuch-II, 1. Erg. ,(2008) VDLUFA-Verlag, Darmstadt Kap. 5.1.1.1, 1-4
[7] Jacobsen, A. M., & Halling-Sorensen, B. (2006). Multi-component analysis of tetracyclines, sulfonamides and tylosin in swine manure by liquid chromatography-tandem mass spectrometry. Anal Bioanal Chem, 384(5), 1164-1174
[8] Düngemittelverordnung vom 5. Dezember (2012) (BGBl. I S. 2482), Anlage 2, Absatz 1.4 Schadstoffe
[9] Lamshoft, M., Sukul, P., Zuhlke, S., & Spiteller, M. (2007). Metabolism of 14C-labelled and non-labelled sulfadiazine after administration to pigs. Anal Bioanal Chem, 388(8), 1733-1745
[10]Ding, C., & He, J. (2010). Effect of antibiotics in the environment on microbial populations. Appl Microbiol Biotechnol, 87(3), 925-941
[11] Holzel, C. S., Harms, K. S., Kuchenhoff, H., Kunz, A., Muller, C., Meyer, K., Bauer, J. (2010). Phenotypic and genotypic bacterial antimicrobial resistance in liquid pig manure is variously associated with contents of tetracyclines and sulfonamides. J Appl Microbiol, 108(5), 1642-1656.

TA-P_07 WAVALUE – A new process to produce commercial fertilizers from digestate generated at biogas plants

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1. Objectives

The aim of this project (Wavalue) is to develop and tested a new technology to formulate and granulate organic wastes, specifically biogas plants digestate evaluating, performance, costs , Life cycle analyses . With the WAVALUE project, we are trying to introduce a process to transform digestate, or the by-products generated during the treatment processes, into high-value commercial fertilizers. These fertilizers are granulated, totally spherical with sizes between 1and 4 mm in diameter and have a N:P:K value that has been tailored to meet market demand.

2. Prototype description

The process for transforming digestate into commercial fertilizers comprises two basic steps:

1) Nutrient composition balancing: Liquid digestate is first mixed with other high nutrient organic wastes and/or commercial fertilizer chemicals in order to fortify and balance the N:P:K value to commercially viable levels recognized in the industry.

2) Drying and granulation: The liquid or slurry mixture from 1) above is then introduced into a Spouted Bed- type dryer/granulator where completely round granules of various sizes are produced. The dry granular product appears to be physically similar to commercial mineral fertilizers but differs in that it could contain between 20 and 100% organic materials.

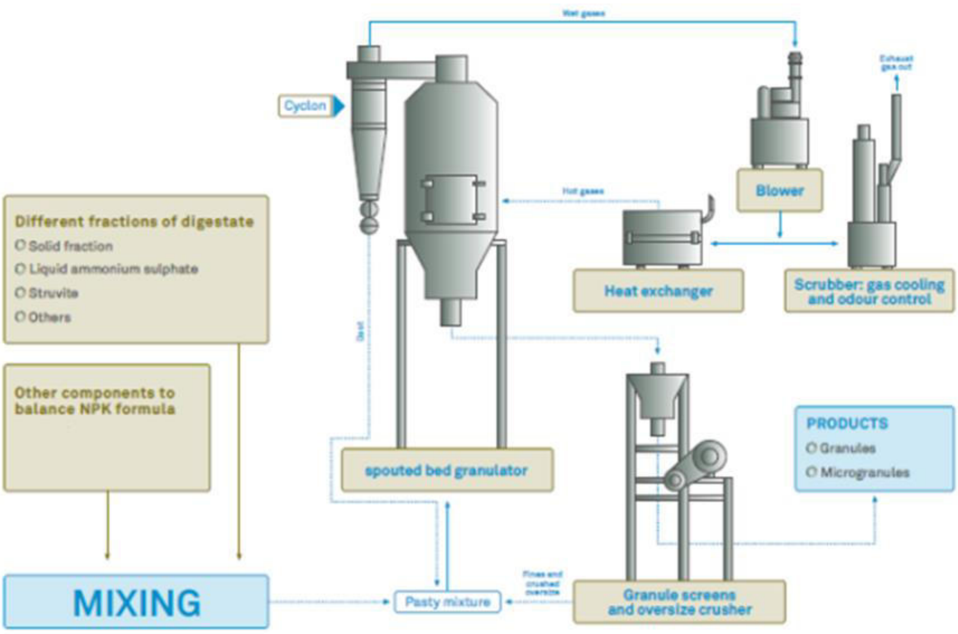

Figure 1: Simple process diagram of the WAVALUE PROCESS

2.1 Dosing and mixing:

The objective of this stage is to achieve a mixture with an adjusted N:P:K ratio, as well as the adjustment of the rheology and chemical properties required for good granulation.

The mixture includes the following fractions obtained when treating digestate: solid fraction from solid/liquid separation, liquid ammonium sulphate from ammonia stripping processes, and/or struvite obtained during the removal of phosphorus P. These concentrated fractions contain the nutrients present in digestate. The solid fraction of raw digestate is usually thickened or dewatered to reduce the energy costs of the dryer.

One or more of the fractions, could be mixed with different mineral fertilizers, depending on the desired N:P: K ratio in the final product.

The above materials are pumped into a mixing reactor into which different liquid fertilizer chemicals are dosed. Reactions are carefully controlled by programmed dosing pumps to ensure stoichiometric reactions thereby avoiding un-reacted chemicals in the final products. This results in a liquid or pasty material suitable for granulation in the Spouted Bed dryer.

2.2 Spouted Bed Drying:

The Spouted Bed Dryer is a fluid bed type dryer, but it has a vertical configuration with a conic bottom. Hot air is introduced in the conical section of the chamber which has been filled with fine material or “seed”. In the pilot unit this hot air is derived from a propane burner but in real life waste heat recovered from prime movers will be used. Hot aire flows vertically through the granules, producing strong circulating movements.

Fortified slurry previously prepared is then gradually introduced into the granular bed where it is quickly dispersed onto the surface of the fast moving seed material. The granules grow layer by layer from tiny balls into larger products. New seed material, created by attrition in the drying bed, forms the basis for new granules which grow into the desired size before being extracted from the bed via a pneumatic conveying mechanism.

Hot air is introduced in the bottom at about 300 ºC and exits the upper end of the vertical “free board” area of the drying chamber at about 100ºC. This hot air or gas serves to remove the moisture from the original pasty mixture as super heated steam.

This spouted fluid bed drying process enables the production of uniformly round hard dust free granules from the liquid fortified slurry produced from the digestate. Fines from the process is recovered in a cyclone and reused as seed material while the humid gasses are scrubbed to remove the evaporated moisture. The spherical granules are hard because of the layering process introduced by the spouting action.

Figure 2: General view of the granulation plant in NEIKER – TECNALIA, in Arkaute.

2.3 Screening of granules:

As mentioned above, the granules are continuously extracted from the drying chamber as they are produced and pneumatically conveyed to the product screening station where the different sizes of granules are classified:

Product sizes are: 2-4 mm, 1-2 mm and 0.5-1 mm (micro-granules). Oversize granules pass through a hammer mill from where the fines are recycled back into the feed slurry.

2. 4 Gas circuit:

Hot gases at between 100 and 110 ºC are drawn from the upper vertical portion of the drying chamber and after passing through a cyclone to separate the dust generated in the granulation process, continue on to a condensing scrubber used to remove the evaporated moisture. From there the scrubbed gasses are reheated back to 300 °C and re-introduced into the dryer. Dust from the cyclone is also recycled in the feed slurry.
This closed circuit configuration of gas recycling, minimizes gas and odour emissions into the atmosphere since the volatile fraction of the gases are oxidized when passing through the high temperature end of the burner chamber inside the heat exchanger.

3. Results and discussion

3.1 Developed Fertilizer products

Developed fertilizer products include a wide range of NPK values, from relatively low (for example 9-2-2 with 70% organic matter) when digestate is the main component of the mixture, to high NPK values (11-15-11 with 20% organic matter) when combined with mineral fertilizers.
Because of the reactions that take place between organic matter and added mineral nutrients, those fertilizers have slow release behavior. That makes them specially suitable for gardens, sport courses or horticulture.

A special type of fertilizer that can be produced are the microgranules. Microgranules are granules between 0,5 – 1,5 mm diameter. Because of their small size, they can be applied together with the seed, under the same labor. That puts the fertilizer in a “ultralocalizated” position, allowing a total availability of the fertilizer for the plant, at first moments of germination. Then, the fertilizer dose can be strongly reduced, without losing any production.

Figure 3: Different fertilizer products developed, the color is different because of the different organic matter content

3.2 Life cycle Analysis

Two life cycle analysis have been conducted, considering the whole chain and all the processes implied on fertilizers production using conventional processes for mineral fertilizers production based or on the other hand, considering organic-mineral fertilizers based on sewage sludge using Wavalue prototype. The compared fertilizers have been: one of the fertilizers developed in this project (12-12-12), and an equivalent dose of a very popular and similar mineral fertilizer (15-15-15). The result of the comparison is that the fertilizer produced by the WAVALUE process, produces 50% less impact on the global warming (IPCC 2007 – kg CO_2 eq index). Reducing also significantly the impacts in all the rest of usually used indexes in life cycle analysis. The employed methodology is based on Yara report (2010)

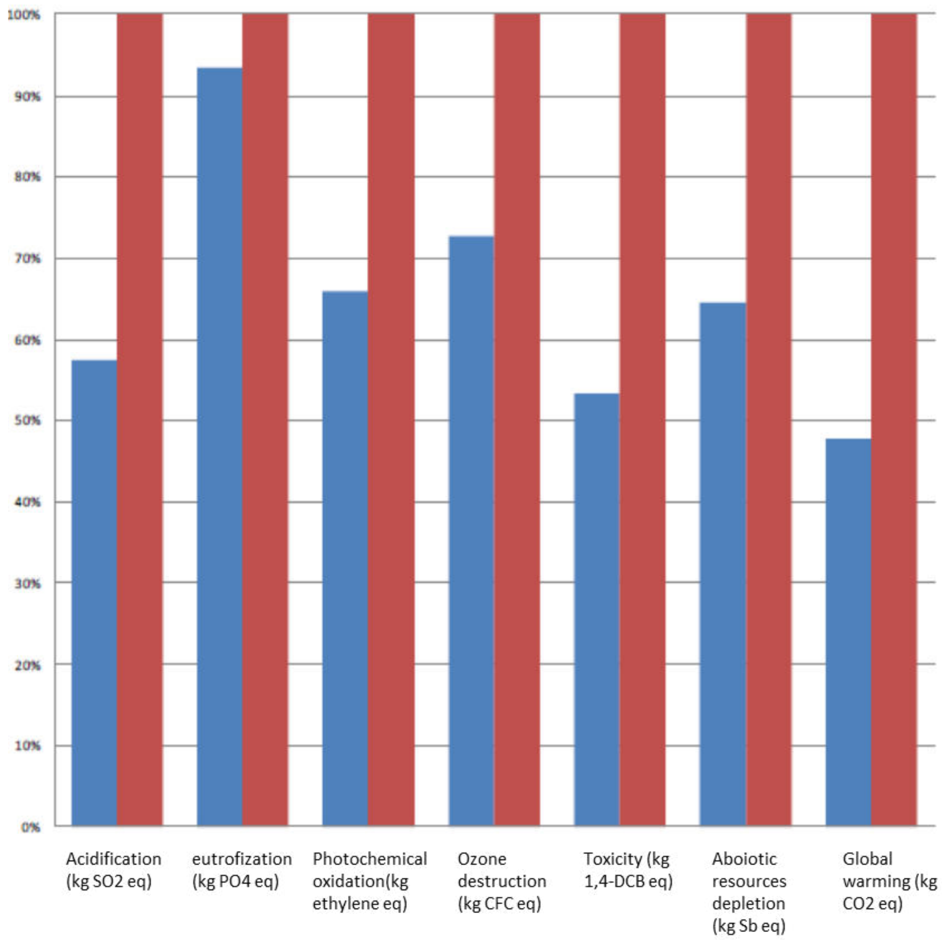

Figure. 4: Percentage of different impact indexes caused by WAVALUE fertilizer (blue) in comparison with mineral fertilizer (red).

3.3 Economic features

The economic success of a digestate granulation project, obviously, depends critically on the selling price of the final fertilizer product. During WAVALUE project, studying the commercialization ways and prices is been a key activity. The conclusion is that those granules, depending on the composition and the granule size, can be sold between 100 and 500 euros per ton, from manufacturer to distributor. Those prices make possible attractive returns on investment in many cases.

Another important feature are the synergies generated between biogas production, digestate treatment and fertilizer production. Features like the use of waste heat, saving logistic costs to spread digestate, saving the need of wide agricultural fields to spread digestate … can have a big impact on the economic result of a biogas project. For that reason, in the WAVALUE project, different calculation tools have been developed, to establish the economic consequences that can have choosing one or other project option, individually for each possible conditions.

Acknowledgements

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References

[1] Yara. 2010 Methodology for calculating the carbon footprint of AN-based fertilizers (2010), www.yara.com

TA-P_08 Effects of thermal drying on phosphorus availability from sewage sludge

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1. Objectives

Sewage sludge represents one of the largest pools of phosphorus (P) in urban waste streams [1]. Thermal drying of sewage sludge implies sanitation [2] and improves practical handling options of the sludge; however, it may also affect its value as a P fertiliser. The main objective of this study was to assess whether thermal drying of sewage sludge, as well as drying temperature, affects plant P availability.

2. Methodology

Sludge P availability were assessed through an incubation experiment and a pot experiment, including triple super phosphate (TSP) as a reference. Sludges used in the experiments (see details in table 1) originated from Randers Spildevand A/S, a municipal wastewater treatment plant in Denmark. Wastewater P removal at the plant combines enhanced biological and chemical (Fe dosing) removal. Sludge treatment involves anaerobic digestion, belt press dewatering and thermally drying at temperature up to 200°C. Sampled sludges were anaerobically digested and dewatered sludge (DWS) and thermally dried sludge (DRS). Subsamples of DWS were dried in a laboratory oven at different temperatures to obtain DWS 70, DWS 130, DWS 170 and DWS 250. All materials were ground to a particle size of <2 mm before application to soil. The used soil was a sandy loam with a total P content of 673 mg P kg^{-1}.

- Incubation experiment: All sludges and TSP were included in a 197 days soil incubation experiment (80 mg P kg^{-1} soil) with sampling over time for DGT (diffusive gradients in thin films) analyses as an indicator of P availability.
- Pot experiment: DWS, DRS and TSP were included in a pot experiment with evaluation of dry matter yield and P uptake in spring barley after 6 weeks growth in a greenhouse. To enhance P effects, the soil was mixed with quartz sand at ratio 1:1 and other nutrients than P were supplied in excess.

Table 1: Overview and details of the used sludges and TSP: dry matter (DM) %, concentrations of P, N and NH_4-N, and water extractable P (WEP).

		DM %	pH	Total P	Total N (g kg^{-1} DM)	NH_4-N	WEP (% of total P)
DWS	Dewatered sludge from Randers	19	8.4	38	57	9.0	1.26 ± 0.01
DRS	Dried sludge from Randers	95	7.3	39	47	3.4	2.00 ± 0.03
DWS 70	DWS dried in a laboratory oven at 70°C	>99	6.9	n.d.	45	6.1	1.23 ± 0.04
DWS 130	DWS dried in a laboratory oven at 130°C	>99	6.6	n.d.	45	4.6	1.14 ± 0.01
DWS 190	DWS dried in a laboratory oven at 190°C	>99	5.8	n.d.	45	1.1	1.69 ± 0.01
DWS 250	DWS dried in a laboratory oven at 250°C	>99	6.5	n.d.	45	1.2	0.93 ± 0.03
TSP	Triple super phosphate	100	3.7	174	n.d.	n.d.	62.66 ± 5.55

3. Results and discussion

3.1 Incubation experiment

Whereas P availability from the TSP-amended soil clearly decreased over time, the sludge-amended soils remained at the same level or increased slightly (figure 1a). However, the ranking of P availability for the different amendments remained the same throughout the experiment (average C_{DGT} in µg l^{-1} is shown in brackets): TSP (186) > DWS (66)> DRS (42) > control soil with no P amendment (19). C_{DGT} is the time average concentration of P at the interface of the DGT and the soil calculated according to [7]. No differences in C_{DGT} on soils amended with the four laboratory-dried sludges (measured on days 7 and 197) were found (figure 1b). Average C_{DGT} of these soils corresponded to 73 % of that of the soil amended with DRS.

The immediate P availability, as indicated by C_{DGT} one day after application of fertilisers, was 4 times higher for TSP than for DWS. However, after 197 days, this difference had levelled out and P availability from the TSP supplied soil was only 1.5 times higher than for the DWS applied soil. The sharp initial decrease for the TSP amended soil is most likely due to adsorption to soil particles. The initial decrease for DWS is much smaller and can be a result of both adsorption of the most soluble P fractions as well as immobilisation since also organic matter was added with the sludge. For both sludges, the incubation experiment showed an increase in P availability over time, from 14 days to 197 days, potentially a result of mineralisation of organic matter associated P.

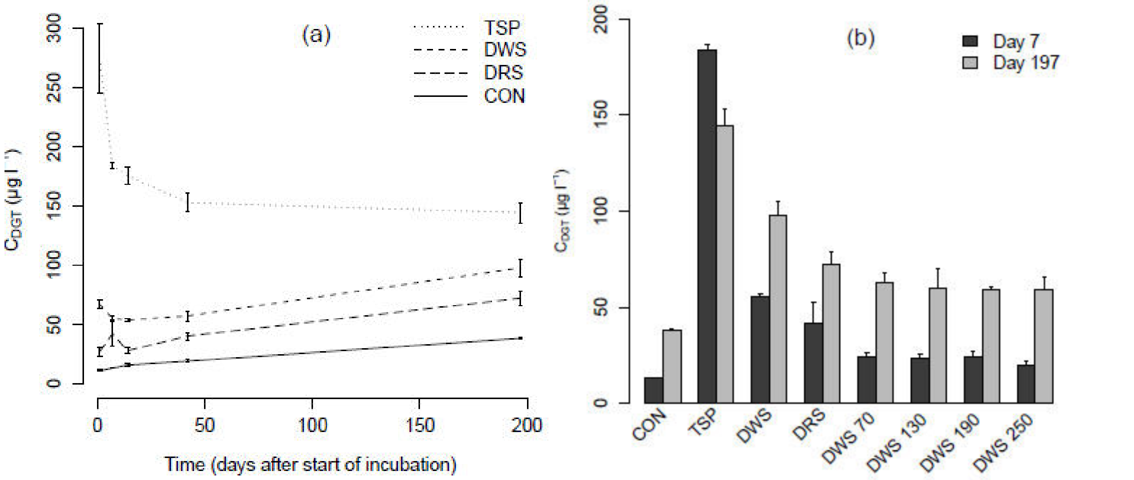

Figure 1: Development in C_{DGT} over time for TSP, DWS, DRS and CON (a) and C_{DGT} on day 7 and day 197 for all treat ments (b) in the incubation experiment. Error bars show standard error of the mean (n=3).

3.2 Pot experiment

Applications of TSP, DWS and DRS increased dry matter yields, P concentrations and P uptake in spring barley after 6 weeks of growth compared to no application of fertiliser (CON) (see Figure 2). Significant effects of the treatment were found for P concentrations and P uptake (p<0.001), but not for dry matter yields (p=0.14). The apparent P use efficiency (PUE) (calculated as the difference in P uptake between the fertilised and the unfertilised plant, divided by the amount of P applied with the fertiliser) was 10, 6 and 3 % for TSP, DWS and DRS, respectively.

The pot experiment confirmed the results found in the incubation experiment, and it furthermore emphasized that dried sludge has a poorer P fertiliser value than non-dried sludge. In line with the findings from the incubation experiment, TSP was superior to both sludges in its ability to support P uptake. However, whereas TSP had tripled C_{DGT} compared to DWS at day 42, PUE after 42 days of growth was only 1.7 times higher for TSP compared to DWS (Figure 1a).

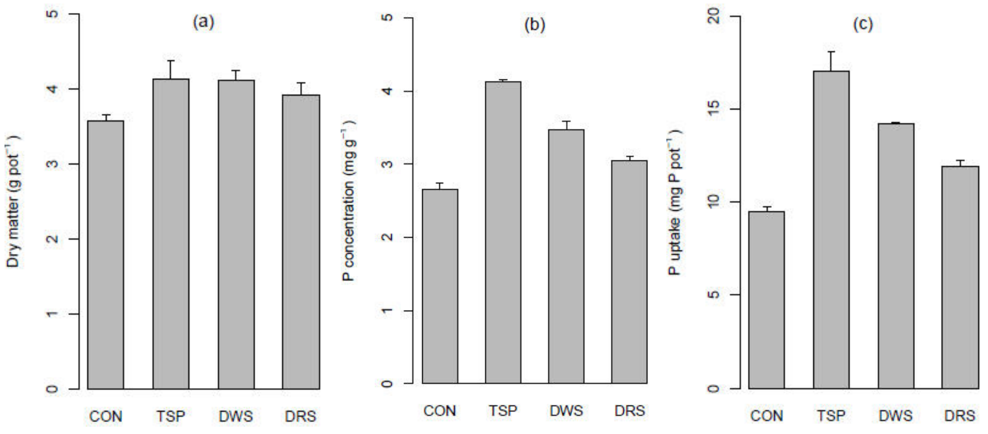

Figure 2: Dry matter yields (a), P concentrations (b) and P uptake (c) for 6 weeks old spring barley plants. Error bars show standard error of the mean (n=3).

3.3 Effects of thermal drying

Thermal drying of the sludge reduced P availability from the sludge after application to soil. This was evident from all results from the incubation experiment which showed an average decrease of 37 % for C_{DGT} (five time points). Also the pot experiment indicated a decline in P availability due to drying as shown by lower dry matter yields, lower P concentrations and a 50% lower PUE. These findings are in line with a previous study which for a range of different sludges, found lower bicarbonate extractable P after soil application of dried sludge compared to non-dried sludge [3].In contrast, other studies have found increases in the most labile P fractions of sewage sludge or manures after drying [4,5,6]. Such increases may be due to hydrolysis of microbial cells and release of microbial P due to the high temperatures [4]. This could also be the reason for the higher amount of water extractable P in DRS compared to DWS (Table 1). After application to soil this trend was no longer apparent, and instead we saw that P from DWS was more available than P from DRS. This indicates that other processes during drying are determining for the actual plant P availability. Such processes could be the formation of more stable calcium phosphates, e.g. apatite, or iron phosphates during drying [4]. The sludge used in our experiment had a high content of Fe (58 g kg^{-1} DM) due to addition of Fe salts during the wastewater treatment. Thus, Fe phosphate compounds present in the sewage sludge may have been affected by the drying and have influenced the final P availability. In fact, a previous study observed that whereas high Fe contents did not influence P availability from “fresh” sewage sludge, P availability from dried sewage sludge was affected negatively by high Fe contents [3].

The temperature during drying did not affect P availability from the sludge after addition to soil (Figure 1b). Differences in water extractable P between the sludge dried at different temperatures (see table 1), however, suggest that temperature does affect the inherent characteristics of the sludges. But since these were not apparent after addition to soil, we believe that changing drying temperatures between 70 and 250°C does not substantially affect sludge P availability.

In general, we saw that the laboratory-dried sludges had lower P availabilities after addition to soil than the on-plant dried sludge (Figure 1b). Reasons for this could include different drying processes and conditions. Whereas, the laboratory-dried sludges were dried at still conditions, on-plant drying involved continuous movement during the drying resulting in very brittle sludge pellets. This could have resulted in different physical properties of the sludges. However, we attempted to diminish this by grinding the samples to the same particle size. More likely, the reason for the differences in P availability between the laboratory-dried sludges and the on-plant dried sludge could be ascribed to the different final water contents of the dried sludges. The laboratory-dried sludges were dried to a DM-content >99%, whereas the on-plant dried sludge still contained around 5% of water. This lower drying intensity of the on-plant dried sludge may also have resulted in less precipitation of stable P and Fe phosphate compounds.

Choice of sewage sludge treatment is a compromise. Important arguments for choosing drying as a final treatment option are sanitation, reduction in volume and the achievement of a more stable and easily manageable product. However, our results show that drying also reduces P availability from the sewage sludge. Furthermore, drying also implies a loss of nitrogen (see table 1) due to evaporation of ammonia. Thus, from a plant nutritional point of view drying of sewage sludge is not an optimal treatment option.

4. Conclusion and outlook

The results clearly demonstrated that thermal drying reduces the plant P availability from sewage sludge. This was obvious from both the incubation experiment and the pot experiment. No evidence was found for the temperature during the drying process to have a major effect. Overall, our results suggest that thermal drying of sewage sludge is not a good treatment option if aiming at optimising plant P availability.

Acknowledgements

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References

[1] Oenema, O, Chardon, W., Ehlert, P., van Dirk, K., Schoumans, O., Rulkens, W. (2012): Phosphorus fertilisers from by-products and wastes, Proceedings 717, International fertilizer society, Leek, UK, pp. 1-56.
[2] Moritsuka, N.; Matsuoka, K.; Matsumoto, S.; Masunaga, T.; Matsui, K., & Wakatsuki, T. (2006): Effects of the application of heated sewage sludge on soil nutrient supply to plants. Soil Science and Plant Nutrition, 52(4), 528-539.
[3] Smith S.R., Triner N.G., Knight J.J. (2002): Phosphorus Release and Fertiliser Value of Enhanced-Treated and Nutrient-Removal Biosolids. Water Environ J 16:127–134.
[4] Ajiboye B., Akinremi O.O., Racz G.J. (2004): Laboratory characterization of phosphorus in fresh and oven-dried organic amendments. J Environ Qual 33:1062–9.
[5] He Z, Cade-Menun BJ, Toor GS, et al. (2007) Comparison of phosphorus forms in wet and dried animal manures by solution phosphorus-31 nuclear magnetic resonance spectroscopy and enzymatic hydrolysis. J Environ Qual 36:1086–95.
[6] Li R, Yin J, Wang W, et al. (2014) Transformation of phosphorus during drying and roasting of sewage sludge. Waste Manag 34:1211–6.
[7] Davison, W., & Zhang, H. (2012) Progress in understanding the use of diffusive gradients in thin films (DGT) - back to basics. Environmental Chemistry, 9(1), 1.

TA-P_09 Recycling vegetable waste, sewage sludge, wood ash and sawdust by composting

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1. Objectives

The objectives of the research were:

- to recycle vegetable waste with sewage sludge, sawdust and wood ash by composting
- to test and utilize compost as: (a) bio-fertilizer; (b) low cost adsorbent substrate in removal of heavy metal (Cu^{2+}) from polluted water.

2. Methodology

The research consists of two steps: (1) obtaining of the composts, and (2) testing and optimization the composts as low-costs adsorbents to remove the toxic Cu^{2+} from polluted water.

- Step 1: 4 types of composts (with different ratios between raw materials) were synthesized and characterized at laboratory scale. The composting process was monitored (15 weeks) by weekly investigating of several physical and chemical parameters: pH, electrical conductivity (EC), content on monocarbohydrates (%G), content on amino acids (%AA), and the C/N ratio. The compost maturity and stability was estimated using FTIR spectroscopy and germination tests, performed 7 days;
- Step 2: The C3 compost was tested as adsorbents of Cu^{2+} from synthetic wastewater. Amounts of (0.5–3 g) BNR2 were added to 100 mL of $Cu(NO_3)_2$ 0.1 N solution (Scharlau). The mixtures were stirred 1÷120min, at room temperature (20-23^0 C); then the substrates were removed by vacuum filtration. Both the initial solutions and the supernatants were analyzed by AAS (Analytik Jena, ZEEnit 700), at λ_{Cu} = 324.75 nm to evaluate the initial (c_i) and the equilibrium concentrations (c_e) There was no physical or chemical treatment applied to the compost prior to adsorption.

3. Results and discussion

3.1 Synthesis and characterization compost as bio-fertiliser

Four composts were prepared: C1: 100% vegetables; C2: 80% vegetables, 10% sawdust, 10% wood ash; C3: 70% vegetables, 20% sewage sludge, 10% sawdust; C4: 70% vegetables, 10% sewage sludge, 10% sawdust, 10% wood ash); the composts were obtained and characterized in the Laboratories of the Centre Renewable Energy Systems and Recycling from the Transilvania University of Brasov, Romania. The data presented in Table 1 for the composts at the end of the composting process (the values of pH, EC, C/N ratio), respectively in Figure 1 (germination index of the composts) are in agreement with the recommended conditions of stability and maturity for bio-fertilizers: pH: 6÷9; EC: 2,0÷3,5 mS/cm; C/N ratio: 20÷35; PSG>60%; GI>85%. The FTIR spectra confirmed the biodegradation of complex compounds such as proteins, poly carbohydrates, etc. into simpler, biological active compounds, like carboxylic acids, alcohols, phenols, amines and their salts, amides etc. proving the viability of the process based on wastes.

Table 1: Characteristics of the compost samples

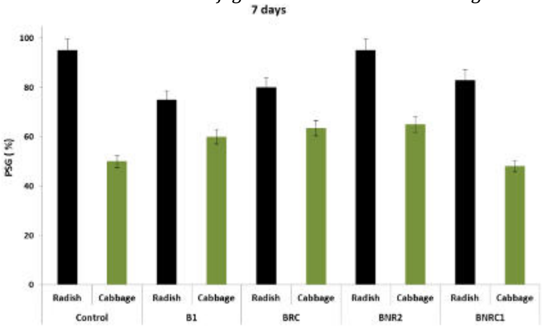
Compost (Notation)	pH	EC mS/cm	% AA	% G	C/N
C1(B1)	8.43	2.10	0.14	0.099	20.45
C2(BRC)	8.50	3.40	0.15	0.450	22.56
C3(BNR2)	8.12	2.47	0.14	0.171	27.14
C4(BNRC1)	7.29	2.31	0.3	0.117	31.78

The germination test represents the most important parameter to evaluate the nutritional quality, toxicity and maturation degree of the compost. The germination indexes GI (for 7 days) combine relative seed germination and relative root growth for soil cultivated with *Raphanus sativus* and

Brassica oleracea, comparable with the control sample (GI = 100%). It is considered that a mature compost, without fitotoxines, has GI = 80-85% or PSG>60% (calculated with Eqs. 1 and 2). As shown in Figure 1 (A and B) all the compost samples had good GI for both radish and cabbage seeds, but the best results were obtained for compost C3 obtained from biomass, sewage sludge and sawdust which have enriched content in nutrients and biocatalysts enzymes.

$$PSG\ (\%) = \frac{number\ of\ seeds\ germinated\ in\ compost\ sample}{number\ of\ seeds\ germinated\ in\ control\ sample}\ x\ 100 \qquad \text{Eq.1}$$

$$GI\ (\%) = \frac{number\ of\ germinated\ seeds\ x\ average\ root\ length\ of\ compost\ sample}{number\ of\ germinated\ seeds\ x\ average\ root\ length\ of\ control\ sample}\ x\ 100 \qquad \text{Eq.2}$$



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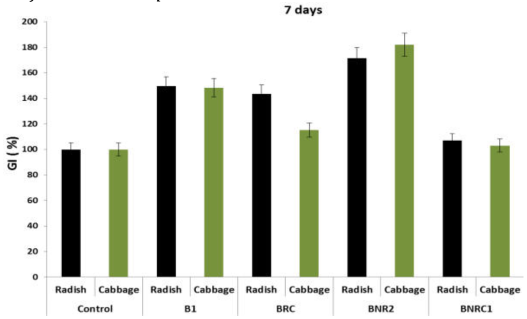


B

Figure 1: Germination test for the compost samples: (A) Percent of seeds germinated (B) Germination index

3.2 Adsorption studies

The C3 compost was used for Cu^{2+} removal from wastewater. The adsorbent compost structure and morphology were investigated, before and after copper adsorption, by FTIR - Fourier transform infrared spectroscopy, AFM - Atomic force microscopy and SEM - Scanning electron microscopy techniques. The heterogeneity of the adsorbent C3 determines a physical, multilayer, cooperative adsorption, on heterogeneous compost surface.

The optimum parameters for the adsorption process: contact time and mass of substrate: solution volume ratio were experimentally optimised based on the process efficiency. The results were 20 min for the equilibration duration and an optimal value of 1.5 g compost for 100mL pollutant solution.

The experiments were run at the natural pH (5~6), when copper ions are hydrated $[Cu(H_2O)_{n=4-6}]^{2+}$, supporting possible interactions with the partially dissociated –OH and –COOH reactive functional groups on the C3 surface.

Adsorption isotherms: In the previously optimized condition, adsorption studies at specific concentrations in the range of 0…300mg/L were done and the substrate adsorption capacity was evaluated by calculating the adsorption coefficient, q (mass of adsorbed pollutant/mass of compost, mg/g). Based on these data, the adsorption isotherms ($q=f(c_e)$) were plotted in the linear form, according to the Langmuir and Freundlich models (eq. 3 and 4).. The results are given in Figure 2.

The linear form of the Langmuir isotherm, is:

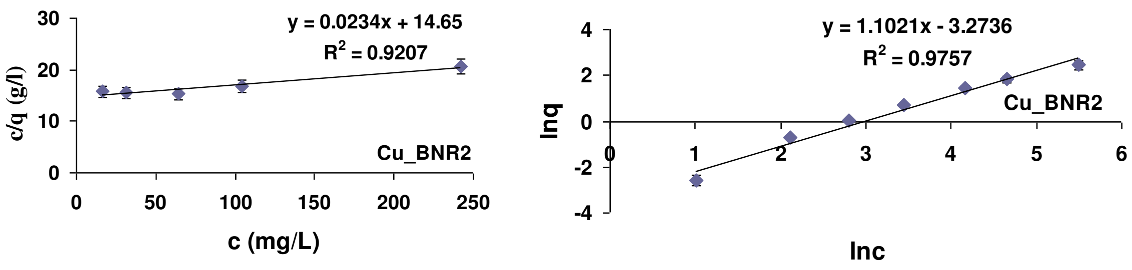
$$\frac{c_e}{q} = \frac{c_e}{q_{max}} + \frac{1}{q_{max} x\ a} \qquad \text{Eq. 3}$$

where: c_e is the equilibrium concentration (mg/L); q_{max} is the maximum adsorption coefficient (the maximum amount adsorbed per specified amount of adsorbent, mg/g); a is the equilibrium constant.

The linear form of the Freundlich isotherm is:

$$\log q = \frac{1}{n} log c_e + log k \qquad \text{Eq. 4}$$

k and n are the Freundlich equilibrium constant and exponent; 1/n giving an indication of adsorption intensity. Generally, for a regular adsorption, 1/n < 1 indicates o favourable adsorption, while 1/n > 1 means an associative adsorption.



A B

Figure 2: The linearization of Langmuir (A) and Freundlich (B) models for BNR2

The parameters of Langmuir and Freundlich models were calculated and reported in Table 2.

Table 2: The related parameters of Langmuir and Freundlich models

Compost	Langmuir model			Freundlich model		
	q_{max} mg/g	a L/mg	R^2	k mg/g	1/n	R^2
BNR2	42,73	0,0016	0,9207	0,0378	1,009	0,9757

The experimental data are well fitted with both models, with better correlation coefficient when Freundlich liniarization is applied, indicating combined adsorption mechanism. Freundlich model is specific for both chemical and physical adsorption onto energetically heterogeneous surfaces with random distribution of the active sites as the compost surface is. The values of the Freundlich parameters (1/n>1) corresponds to a cooperative adsorption of the copper ions onto tested compost. Chemical adsorption via electrostatic interactions of the copper ions and the functional groups of the compost surface of are also possible, following the Langmuir model; the likely active sites for this adsorption type are on the anionic sites, resulted either due to carboxyl dissociation of on the inorganic oxide, existing in traces in the sludge .

Kinetic studies: As described in literature, the mechanism of adsorption depends on the physical and chemical characteristics of the adsorbents. Various kinetic models such as the pseudo-first-order, pseudo second-order and interparticle diffusion have been applied to fit the experimental data to predict the adsorption kinetics. The kinetic parameters are presented in Table 3.

The pseudo-first-order and pseudo-second-order kinetic models are applicable mostly for less porous surfaces, while the interparticle diffusion model give better results for surfaces with small porous.

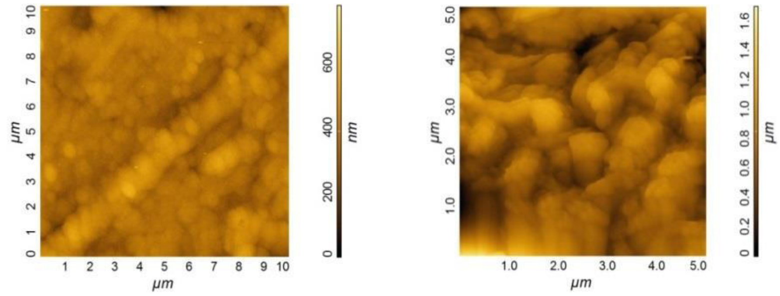
Table 3: The kinetic parameters for the adsorption of Cu^{2+} onto BNR2

Compost	Pseudo first order			Pseudo second order			Interparticle diffusion		
	k_1 min^{-1}	q_e mg/g	R^2	k_2 g/mg min	q_e mg/g	R^2	k_{id} mg/g $min^{-½}$	C mg/g	R^2
(3-120')	0,002	13,14	0,405	0,025	9,64	0,991	0,2656	13,623	0,222
(3-20')	0,002	10,8	0,924	0,033	16,2	0,991	1,3843	8,76	0,903

Pseudo second order kinetic model proved valid for the entire process duration, and corresponds to a moderate concentration/number of active sites, comparable with the copper ions concentrations. The other two tested models fit well the data only in a limited time interval indicating that parallel/complex mechanisms are likely within this range, on the highly heterogeneous substrate.

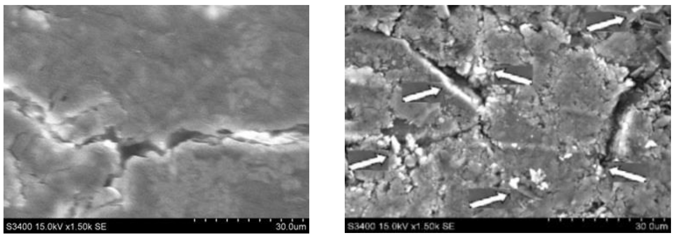
Changes in the surface morphology result as consequence of the copper adsorption onto the compost and are proved by the AFM and SEM images of the substrate before and after the adsorption.

The AFM analyses indicate large aggregates with different dimensions and an increase in the average roughness (see Figure.3.a and 3.b), certifying the adsorption process of Cu^{2+} only on specific sites on the compost surface.



a b

Figure 3: AFM images of the compost: (a) before (Roughness = 42nm;) and (b) after adsorption (Roughness = 51nm;)



a b

Figure 4: Morphology of compost adsorbent BNR2 before (a) and after (b) adsorption of copper ions

The changes in the compost morphology are outlined in the SEM images (Figure. 4.a and 4.b.) The bright spots after the adsorption can be attributed to the adsorption of Cu^{2+} on the compost surface, forming aggregates, in agreement with the AFM data.

4. Conclusion and outlook

Monitoring the composting process outlined interesting correlations between the chemical structure of different types of waste and their biodegradability. Through composting, organic matter undergoes partial mineralization and, to a varying degree, transformation into humus-like substances which can be used directly in agriculture as an organic bio-fertilizer. The results obtained evidenced that percentages of 10-20% of sewage sludge, wood ash and sawdust can be recycled together with vegetable waste for obtaining a good compost, which can be used as bio fertilizer or as low cost adsorbent of Cu^{2+} from polluted water. Considering the very good results obtained in the copper removal efficiencies of 98%, future research will be dedicated to the adsorption of other toxic metals on the compost substrates. It can be concluded that now, when the price of fertilizers is significant, also affecting the sanity of the land, the use of organic waste biodegraded by composting represents a low cost and interesting perspective, both for ecological soil fertilization and for removal of toxic metals from contaminated waters.

Acknowledgements

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References

[1] Dumitrescu, L.; Manciulea, I.; Zaha, C.; Sauciuc, A. (2014): Recycling Biomass Waste to Compost, Sustainable Energy in the Built Environment - Steps Towards *nZEB*, Ed. Springer, 229-243

[2] Zaha, C.; Manciulea, I.; Sauciuc, A. (2011): Reducing the volume of waste by composting vegetable waste, sewage sludge and sawdust, Environmental Engineering and Management Journal 10, 1415-1423.

[3] Vargas, C.; Brandão, P. F. B.; Ágreda, J.; Castillo E. (2012): Cr(VI) removal by compost, Bio Resources 7(3), 2711-2727.

TA-P_10 Evaluation of distillery organic waste compost efficiency on vineyard soil properties and grape quality – A case study in southeastern Spain

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1. Objectives

The objective of this work was to study the effects of a distillery organic waste compost with respect to different traditional organic fertilisers on some physico-chemical, chemical and biological characteristics of a vineyard soil and on the grape yield and quality during a year period.

2. Methodology

The field experiment was carried out during the 2013 season in a 15 year old 'Monastrell' vineyard (*Vitis vinifera* L.) of about 1.1 ha, situated in Monovar (Alicante-Spain; 38°27'4''N, 0°54'56''W; elevation 392 m a.s.l.). The climate in this area is semi-arid continental Mediterranean with an average cumulative annual precipitation of 382 mm and a mean annual temperature of 16.2°C. The soil at the site was a calcareous sandy-loam classified as a Xeric Haplocalcids [1], with an alkaline nature and low salinity and organic C content. The main characteristics of the soil are shown in Table 1.

Table 1: Characteristics of the soil used in the experiment

Parameter*	Value
pH	8.28
Electrical conductivity (dS m^{-1})	0.28
Sand (%)	79
Loam (%)	8
Clay (%)	13
Texture	Sandy-loam
Active $CaCO_3$ (%)	11.8
Oxidisable organic C (%)	0.33
Organic N (mg kg^{-1})	572
NH_4^+-N (mg kg^{-1})	0.57
NO_3^--N (mg kg^{-1})	2.38
Available P (mg kg^{-1})	11.7
Available K (mg kg^{-1})	400

*Values on a dry matter basis

Four treatments, in a completely randomised design with three replicates per treatment, were set up in experimental plots (87 m^2), each containing 20 vines planted with a row spacing of 3 m × 1.5 m (row x vine) equivalent to 2300 vines ha^{-1}. The treatments were: treatment without organic fertiliser (C); sheep/goat manure (SGM) (5.5 t ha^{-1}); distillery organic waste compost (DC) (5.6 t ha^{-1}) and sheep/goat manure compost (SGC) (17 t ha^{-1}). The main characteristics of these organic materials are shown in Table 2. The amendment application rate was adjusted to supply 170 kg ha^{-1} of nitrogen. No additional inorganic fertilisation was applied throughout the experiment and only one irrigation was carried out during the growing season (140 days).
pH, EC, C_{oox}, N_{org} and soil respiration were measured throughout the experiment, after the addition to the soil of the organic fertilisers. Also, grape yield and grape quality parameters were determined. All the analytical determinations in the manure, compost, soil and grape samples were determined according to the methods described by Rubio et al. [2]. All analyses were made in triplicate. Confidence intervals at P<0.05 were calculated for all figures to compare the effect of the different treatments. The mean values of the grape yield and quality parameters were tested

for statistically significant differences using one-way analysis of variance (ANOVA), considering the type of treatment. Comparison of the means of these parameters was performed by the Tukey-b test at $P < 0.05$. Data analysis was carried out using the SPSS v. 22.0 statistical software packages.

Table 2: Characteristics of the organic materials used in the experiment.

Parameter*	SGM	DC	SGC
pH	7.37	7.96	7.67
Electrical conductivity (dS m^{-1})	11.6	6.5	16.9
Total organic C (%)	22.2	42.5	24.6
Total N (%)	3.08	3.04	1.00
Total organic C/Total N	7.2	13.9	24.6
P (g kg^{-1})	2.08	1.84	2.04
K (g kg^{-1})	48.1	71.4	61.7

*Values on a dry matter basis

3. Results and discussion

3.1 Effect of the treatments on soil properties

At the beginning of the experiment, the soil addition of the amendments produced a decrease of pH values in comparison to control soil (Fig. 1a). However, after this initial acidification, the pH increased to values close to those of the control soil only in the case of DC treatment. The addition of the SGM and SGC treatments reduced the soil pH values throughout the whole experimental period. This fact could improve the plant micronutrient assimilation, especially in the case of Fe, reducing the common iron chlorosis of plant of this area. Hannachi et al. [3] also observed that the soil addition of composted wastes from vegetal and animal sources mixed with manure significantly decreased soil pH in cultivated dryland soils from southern Tunisia. These authors concluded that in alkaline soils, a pH lowering can be considered as an ameliorating process as it may help provide nutrients to plants.

The evolution of EC in the control and treated soils is shown in Fig. 1b. The EC values of the soils with the C and DC treatments did not statistically differ during the whole period of experiment. However, the addition of SGM and SGC increased the salinity of the soil. This parameter diminished in SGM soil at the end of the experiment, reaching EC final values similar to those of the control soil. In the case of SGC treatment, the addition of this organic fertiliser increased the soil salinity, possibly due to its high application dose (17 t ha^{-1}), as a consequence of its low nitrogen content (N_T = 1%). This fact could restrict the use of this amendment.

The addition of exogenous organic matter from the organic fertilisers significantly increased soil organic carbon content throughout the experimental period with respect to the control soil (Fig. 1c). The initial values of this parameter were 0.33, 0.86, 1.09 and 1.70 % and the final values 0.45, 0.88, 0.74 and 1.68 % for control, SGM, DC and SGC soils, respectively. This result was also obtained by Morlat and Chaussod [4] and Bustamante et al. [5] in long-term addition of different organic amendments to calcareous vineyard soils.

The incorporation of DC treatment into the vineyard soil did not induce notable effects in soil respiration with respect to the control treatment probably due to the high degree of OM stability of this compost ($C_{org}/N_T < 20$; Bernal et al. [6]) (Fig. 1d). This fact was also stated by Bustamante et al. [7] in a short-term experiment with several organic wastes added to soil after undergoing varying composting times, who reported highest soil respiration for the soils amended with the less stabilised organic material. However, the addition of SGM and SGC increased the initial values of soil respiration with respect to the control soil. Finally, this parameter gradually decreased towards similar values in all the treatments. By the end of the incubation, there were no great differences in the respiration rates among treatments.

The addition of the studied organic fertilisers produced an initial increase of the N_{org} concentration (Fig. 1e). In all the amended soils, an initial decrease in this parameter was observed, as a consequence of the N_{org} mineralisation, followed by an increase. The increases of the N_{org} concentration in the amended soils could be due to immobilisation processes [8] or due to the nitrogen biological fixation, which has been observed by others authors in a soil amended with organic wastes [9]. Finally, this parameter gradually decreased towards similar values in C, SGM and DC treatments. Only the SGC soils increased significantly the N_{org} content of soil at the end of the experiment.

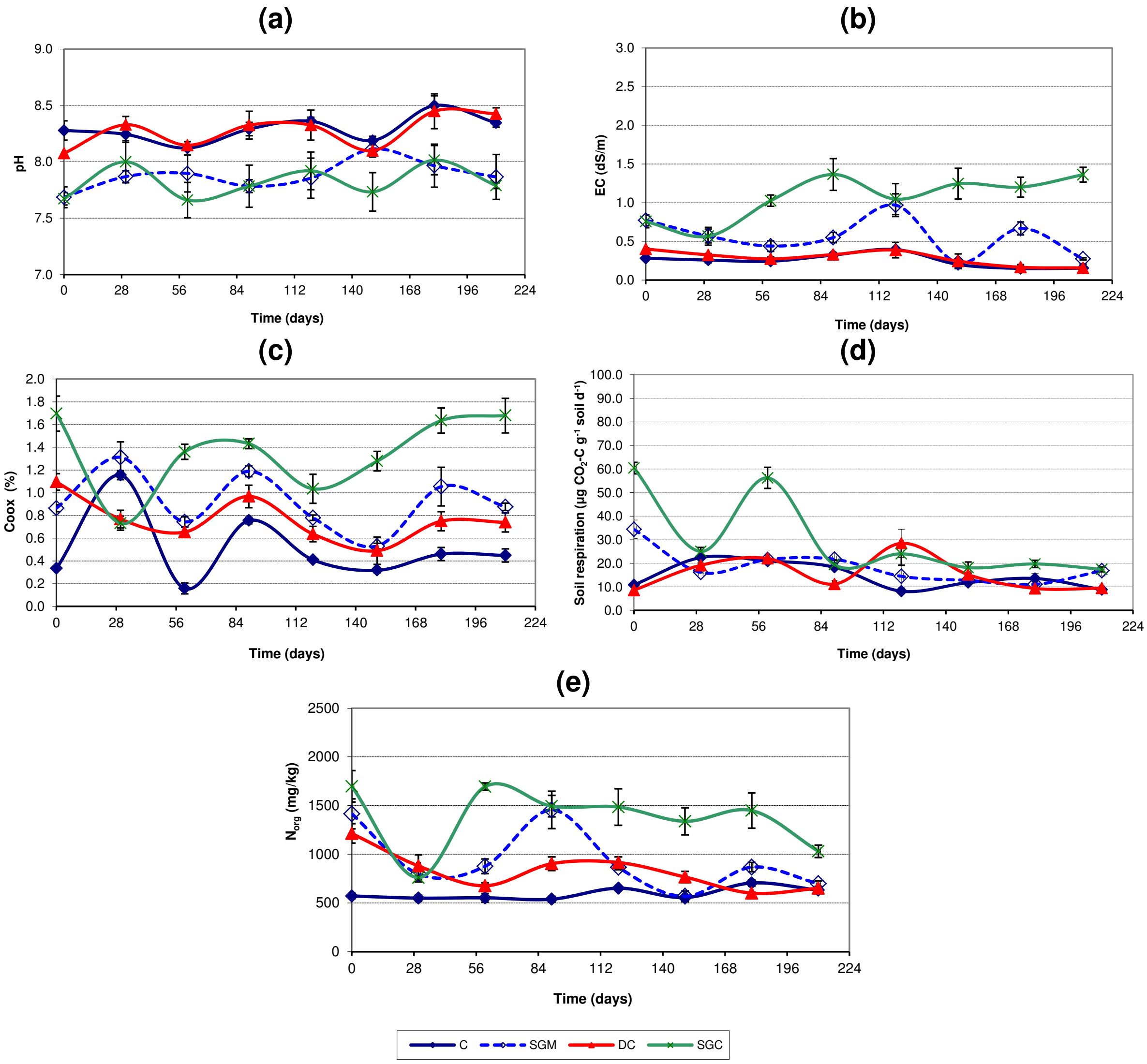

Figure 1: Evolution of pH (a); electrical conductivity (EC) (b); oxidisable organic carbon content (C_{oox}) (c); soil respiration (d) and organic nitrogen content (N_{org}) (e) in the control and amended soils during the experiment (C: control; SGM: sheep/goat manure; DC: distillery organic waste compost; SGC: sheep/goat manure compost).

3.2 Effect of treatments on grape yield grape and quality

The use of the studied organic amendments produced significant increases in the vine production in comparison to the control, especially in the case of SGC treatment (Table 3). However, other authors have reported an absence of any significant effect of organic fertilisers on grape yield [10, 11]. This discordance in the obtained results could be due to the difference age of the vineyards. The above-mentioned authors carried out their experiments in vineyards of 30-year-old vines, which usually have very deep-reaching roots. As the organic fertilisers were incorporated into the top layer of the vineyard soil, more pronounced effects may be seen in the upcoming years, when the finer particles of the amendments will likely move down into deeper soil horizons as observed by Schmidt et al. [11]. However, our experimental vineyard consisted of 15-year-old vines with shallow roots.

In this experiment, no significant differences were found in most of the quality parameters of grape between the different treatments (Table 3). These results were also observed by Mugnai et al. [12] and Schmidt et al. [11] in long-term application of green waste compost and biochar to vineyard soils, respectively. However, Morlat and Symoneaux [13] found a reduction of grape quality with long-term additions of organic amendments to calcareous sandy soil. This fact was presumably in connection with available nitrogen. In this experiment, the nitrogen supplied by the organic amendments did not affect negatively grape quality.

Table 3: Analysis of the effects generated by the addition of different organic materials on the grape yield and quality

Treatment	Production (kg/plant)	Berry weight (g)	pH	Total acidity (g/L)	º Be	Total anthocyanins (mg/L)	Extractable anthocyanins (mg/L)	TPI
C	0.67 a	1.36 a	3.50 a	5.01 a	12.20 a	407 a	333 a	16.53 a
SGM	3.02 b	1.60 a	3.37 a	5.34 a	11.66 a	448 a	288 a	18.73 a
DC	3.60 b	1.50 a	3.44 a	4.82 a	12.63 a	880 c	317 a	17.30 a
SGC	5.62 c	1.52 a	3.38 a	5.52 a	11.46 a	657 b	261 a	19.10 a
F-ANOVA	38.29***	0.99^{NS}	0.86^{NS}	1.53^{NS}	1.55^{NS}	19.30**	0.60^{NS}	0.58^{NS}

C: control; SGM: sheep/goat manure; DC: distillery organic waste compost; SGC: sheep/goat manure compost; TPI: total polyphenol index.

∗∗∗, ∗∗ and NS: significant at $P < 0.001$, 0.01 and not significant, respectively.

Values in columns followed by the same letter are not statistically different according to the Tukey's *b* test at $P < 0.05$.

4. Conclusion and outlook

From the data obtained it can be concluded that the use of the distillery organic waste compost improved soil quality and increased grape yield, obtaining similar results to those observed with the traditional organic fertilisers. However, no differences were found among the different treatments respect to the grape quality. This fact indicates that the use of distillery waste organic compost as organic fertiliser can be a method to maximise the use of residual nutrients from the organic wastes generated in the distilleries associated to the winery industry.

Acknowledgements

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References

[1] Soil Survey Staff (2014): Keys to Soil Taxonomy. 12th ed. USDA-Natural Resources Conservation Service Washington USA.

[2] Rubio, R.; Pérez-Murcia, M.D.; Agulló, E.; Bustamante, M.A.; Sánchez, C; Paredes, C.; Moral, R. (2013): Recycling of agro-food wastes into vineyards by composting: Agronomic validation in field conditions, Communications in Soil Science and Plant Analysis 44, 502-516.

[3] Hannachi, N.; Cocco, S.; Fornasier, F.; Agnelli, A.; Brecciaroli, G.; Massaccesi, L.; Weindorf, D.; Corti, G. (2015): Effects of cultivation on chemical and biochemical properties of dryland soils from southern Tunisia, Agriculture, Ecosystems and Environment 199, 249-260.

[4] Morlat, R.; Chaussod, R. (2008): Long-term additions of organic amendments in a Loire Valley vineyard. I. Effects on properties of a calcareous sandy soil, American Journal of Enology and Viticulture 59, 353-363.

[5] Bustamante, M.A.; Said-Pullicino, D.; Paredes, C.; Moral, R. (2011): Application of winery and distillery waste composts to a Jumilla (SE Spain) vineyard: Effects on the characteristics of a calcareous sandy-loam soil, Agriculture, Ecosystems and Environment 140, 80-87.

[6] Bernal, M.P.; Alburquerque, J.A.; Moral, R. (2009). Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresource Technology 100, 5444-5453.

[7] Bustamante, M.A.; Said-Pullicino, D.; Paredes, C.; Cecilia, J.A.; Moral, R. (2010): Influences of winery-distillery waste compost stability and soil type on soil carbon dynamics in amended soils, Waste Management 30, 1966-1975.

[8] Bustamante, M.A.; Pérez-Murcia, M.D.; Paredes, C.; Moral, R.; Pérez-Espinosa, A.; Moreno-Caselles, J. (2007): Short-term carbon and nitrogen mineralisation in soil amended with winery and distillery organic wastes. Bioresource Technology 98, 3269–3277.

[9] Mårtensson, A.M.; Torstensson, L. (1996): Monitoring sewage sludge using heterotrophic nitrogen fixing microorganisms. Soil Biology and Biochemistry 28, 1621-1630.

[10] Morlat, R. (2008): Long-term additions of organic amendments in a Loire valley vineyard on a calcareous sandy soil. II. Effects on root system, growth, grape yield, and foliar nutrient status of a cabernet franc vine, American Journal of Enology and Viticulture 59, 364-374.

[11] Schmidt, H.P.; Kammann, C.; Nigglia, C.; Evangelouc, M.W.H.; Mackied, K.A.; Abiven, S. (2015): Biochar and biochar-compost as soil amendments to a vineyard soil: Influences on plant growth, nutrient uptake, plant health and grape quality, Agriculture, Ecosystems and Environment 191, 117-123.

[12] Mugnai, S.; Masi, E.; Azzarello, E.; Mancuso, E. (2012): Influence of long-term application of green waste compost on soil characteristics and growth, yield and quality of grape (*Vitis vinífera* L.), Compost Science and Utilization 20, 29-33.

[13] Morlat, R.; Symoneaux, R. (2008): Long-term additions of organic amendments in a Loire valley vineyard on a calcareous sandy soil. III. Effects on fruit composition and chemical and sensory characteristics of a Cabernet franc wine. American Journal of Enology and Viticulture 59, 375-386.

TA-P_11 Procedure for defining new Swiss standard values for the nutrient excretions of dairy cows

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1. Background and objectives

For the compulsory farm nutrient balance calculation and fertilization planning, Swiss farmers need reliable standard values for livestock excretions of nitrogen (N), phosphorus (P) and other minerals. In principle, the excretions are derived from balance calculations of dietary nutrient intake minus nutrient retention in produced milk and body growth. The inputs and assumptions of these balance calculations should, as far as possible, reflect typical current standard practice on farms. These nutrient excretion standard values are also used to derive the composition of different manure types by combining them with standard values on the amount of liquid (before the addition of water from washing etc.) and solid manure produced per animal place per year in different housing systems.

For the new edition of the Swiss fertilization guidelines (SFG; to be published 2016; current version: Agroscope 2009) a major review and revision of the standard values on excretion is planned for all major livestock categories. The general approach used to derive such standard values is shown here using dairy cows as an example. Dairy cows are contributing about 50% of the total livestock N and P excretion and production has changed considerably in the recent past. Although tentative results will be available at the conference, the focus of this contribution is on the approach used.

2. Methodology

The following elements were used to determine nutrient excretions from dairy cows:

- Data on the components of the forage diet on Swiss dairy farms from a representative survey on livestock and manure management performed in 2010 (Kupper et al. 2015). From this, the share of major diet types and their relative importance were derived.
- Expert assumptions on the proportion of the forages within each diet type.
- New scientific results on the mineral content of forages used in the diet types.
- Mean nutrient content of three types of compound feeds and of three mineral feeds derived from the list of products found on the market in 2014.
- Statistical data on milk yield and average animal weight of dairy cows with consensus expert assumption of what levels to use as base values.
- Database values on the content of milk, animal (during first lactation) and placenta growth.
- The official current feeding recommendations for dairy cows (chapter 6 of Swiss "Feeding recommendations for ruminants" (SFRR; Agroscope 2015)
- A model based on the SFRR to estimate consumption, production and excretion of nutrients for dairy cows on a weekly basis.

2.1 Dairy cow diets

In the representative survey from 2010 on livestock and manure management on Swiss farms (Kupper et al. 2015) information from 1678 dairy farms keeping 35.939 cows could be analyzed. The twelve-page questionnaire was designed in such a way that farmers should be able to fill it in in maximum one hour without any special data query. Regarding the dairy cow diets, this meant that they just had to mark which of the forages (grass, hay, grass silage, maize silage and whole plant maize pellets) they used during summer and winter feeding. The additional questions concerning grazing and amount of compound fed were ignored in the present calculations.

According to the forage used, the farms in the survey were allocated to seven winter and nine summer diets (Table 1), to study the importance of each. As it would not be reasonable to produce different standard values according to geographic zones (valley, hill and mountain), the deviation between the average of all farms and those from the valley and hill zone, where 60% of the cows are kept, was checked and not considered as important. Taking into account the diets with an importance of at least 5%, it was decided what share each of the remaining three winter and six summer diets would have in the calculation of the weighted average of excretions. In the winter diet with hay, grass silage and maize, some sugar beet pulp silage was also considered, as this component is commonly used on many farms, but not during the whole winter feeding period. The value of 6% of total dry matter consumption is based on a calculation considering the amount of sugar beet pulp used in dairy production and the total (national) number of dairy cows receiving such a diet.

Table 1: a) Importance of different winter and summer diets (percent of cows in the 2010 survey) for the whole sample and the valley and hill zone only. Share allocated to each diet in the calculation of the weighted average of excretions for the standard values (SV).
b) Expert assumptions about the proportion of components in the selected diets.

a) Survey results				b) Expert assumptions for forage proportions in selected diets					
winter feeding	all	valley/hill	share for SV	G	H	GS	MS	MP	SBP
H only	28.6%	27.0%	30%		100%				
H + MP	12.5%	13.4%	13%		85%			15%	
H + MS	1.0%	0.8%							
H + GS	7.5%	3.9%							
H + GS + MS	48.0%	53.3%	57%		20%	40%	34%		6%
H + GS + MP	1.6%	0.7%							
H + GS + MS + MP	0.9%	0.9%							
			summer feeding						
G only	19.7%	14.3%	15%	100%					
G + H	23.7%	21.1%	25%	80%	20%				
G + MP	4.4%	4.7%							
G + MS	9.8%	10.9%	10%	80%			20%		
G + H + MP	12.3%	13.4%	15%	70%	15%			15%	
G + H + MS	29.2%	34.8%	30%	65%	15%		20%		
G + MS + MP	0.1%	0.1%							
G + H + MS + MP	0.7%	0.7%							
H + GS + MS	na		5%		20%	40%	40%		

* H - hay; G - gras; GS - gras silage; MS - mais silage; MP - maize pellets; SBP - sugar beet pulp silage

2.2 Nutrient and mineral contents in forage components

Swiss grassland is mainly composed of mixed herbage and classified according to botanical composition, growth development stage and growth number (described in SFRR). The classes of botanical composition depend on the proportion of the functional groups grass, legumes and herbs and on the presence of ryegrass: grass rich (G, >70% grass; G_R, >70% grass with more than half as ryegrass), equilibrated (E, 50-70% grass; E_R, 50-70% grass with more than half as ryegrass), forage legume rich (L, >50% legumes) and forage herb rich (H, >50% herbs with thin leaves). The classes of growth development stage are the following: 1 (tillering), 2 (stem elongation), 3 (begin heading), 4 (full heading), 5 (end heading), 6 (flowering) and 7 (fructification) with reference to *Dactylis glomerata* or *Lolium perenne*. The growth number is distinguished between 1st seasonal growth (1st) and re-growths (+2nd). The growth development stages 1 to 3 are typical for intensive management, 4 and 5 for medium intensive, 6 and 7 for extensive.

Based on an expert consensus the following botanical compositions and growth development stages were chosen for the nutrient balance calculations:

- Gras (barn feeding and grazed): mean of E_R and E, stage 2 during lactation, stage 3 for dry cows
- Gras silage: mean of E_R and E, stage 3 during lactation, stage 4 for dry cows
- Hay: mean of E_R and E, stage 4 during lactation, stage 5 for dry cows

The reference values of energy (NEL, net energy for lactation) and protein (crude protein – CP; absorbable protein in the duodenum, based on energy – APDE or based on N available in the rumen – APDN) of forages given in the SFRR (chapter 13) were used. New values based on the investigations of Schlegel (not yet published) were used as basis for the mineral content of herbage. This study differentiates between the growth number. Based on expert experience, the relative weight of the 1st growth was assumed as follows: growth development stage 2 23%, stage 3 29%, stage 4 42%, stage 5 66%.

Table 2 gives an overview of the nutrient contents of the forages used in the calculations.

Table 2: Energy (net energy for lactation – NEL), protein (crude protein – CP; absorbable protein in the duodenum, based on energy – APDE or N available in the rumen – APDN) and mineral content per kg dry matter of the forage components used in the model calculations

component	growth develop. stage	NEL	CP	APDE	APDN	P	K	Mg	Ca
		MJ	g	g	g	g	g	g	g
grass	2	6.4	180	107	120	4.0	32	2.3	7.9
grass	3	6.2	160	102	106.5	3.7	30	2.1	7.8
grass silage	3	6.0	168	81	106	3.7	30	2.4	7.8
grass silage	4	5.7	149	77	94	3.3	27	2.1	7.6
hay	4	5.4	128	87	81	3.3	27	1.8	5.3
hay	5	5.2	108	80	68	2.9	23	1.6	5.0
maize silage		6.3	77	65	47	2.2	10	1.1	1.8
maize pellets		6.4	76	69	49	2.6	13	2.1	0.9

2.3 Nutrient contents in compound feeds and mineral feeds

The nutrient contents of concentrates and mineral supplements are the mean values derived from the list of products found on the market in 2014 (Table 3).

Table 3: Energy (NEL, net energy for lactation), protein (crude protein – CP; absorbable protein in the duodenum, based on energy – APDE or N available in the rumen – APDN) and mineral content per kg fresh matter of the concentrates and mineral supplements used in the model calculations

compound feed type	NEL	RP	APDE	APDN	P	K	Mg	Ca
content per kg FM	MJ	g	g	g	g	g	g	g
energy rich	7.4	115	85	80	5.0	6.0	2.5	8.5
protein rich	6.9	400	230	300	7.0	14.0	3.5	11.0
balanced	7.3	200	130	140	6.0	9.0	3.0	11.0
Mineral feed type					P		Mg	Ca
content per kg FM					g		g	g
Ca rich					50		55	170
P rich					110		65	80
Mg rich					40		120	100

2.4 Animal characteristics and other assumptions

The average 305 day milk yield in the 2010 survey was exactly 7000 kg. The herdbook average of the main breeds ranged from 6736 kg (Swiss Fleckvieh) to 8523 kg (Holstein). It is clear that the standard values must be variable to account for farm specific milk yields. This was so far done with a correction factor (SFG, 2009): basal values for 6500 kg yield and 650 kg average adult live weight (lactation 2+); -10% per 1000 kg lower yield; +2% per 1000 kg higher yield. Whether it will again be done in a similar way or using a simple formula will be decided after the calculations. Milk yields from 5000 to 10,000 kg will be considered. A milk yield of 7500 kg and 650 kg average adult live weight will be assumed for the future basal values. This is higher than today's average, but takes into account the expected increase in milk yield up to 2020 (minimum period of validity of the new SFG). For cows in 1st lactation 93% of the adult weight will be assumed.

As calving in early winter is most common today, the main scenarios will use November 1st as calving date. Other calving dates will also be calculated to check if this would significantly influence the results and if this would have to be considered in the weighted average. The change between winter and summer feeding is set at April 20 and November 15.

Retained nutrient contents in milk and animal growth are the following:

- Per kg milk: 5.5 g N, 1.0 g P, 1.55 g K, 0.1 g Mg, 1.22 g Ca
- Per kg growth during the first lactation: 25 g N, 6.7 g P, 1.6 g K, 0.4 g Mg, 12 g Ca
- For fetus and maternal tissue, weeks 8 to 4 and 3 to 0 before calving, respectively: 17.1/17.1 g N, 4.5/5.2 g P, 1.0/1.0 g K, 015/0.3 g Mg, 6.5/9.0 g Ca

2.5 Model calculation – feeding and excretion

The annual dairy cow feeding model (Münger, unpublished) is based on the official feeding recommendations for ruminants (SFRR; Agroscope 2015), chapter dairy cows. The SFRR are comparable with French and German recommendations. Based on the energy, protein and mineral requirements which depending on production and body mass, the model calculates feed and nutrient intake and the required amounts of compound feed and minerals on a weekly basis. This is done over a whole year, i.e. a lactation-gestation cycle including the dry period. Selectable user entries are possible for milk yield (per lactation) and average life weight of the cows, lactation number (first or higher), calving date, the composition of the basal/forage diet during winter and summer (type and share of forage components) and the dates for dietary changes (forage composition, compound feeds and mineral feeds). The choice of milk yield level determines a lactation curve which was derived from analysis of data from a large number of experimental animals. The selection and supplementation of compound feed and mineral feed is determined weekly by calculating the difference between the animal's nutrient requirements and nutrients supplied by the basal diet (forage). Mobilization of body reserves is included in the model, decreasing over the first twelve weeks of the lactation for energy, six weeks for protein. Rebuilding of both is accounted for over the remaining lactation period, as an additional, linearly increasing requirement. For the optimization the model can choose between energy and protein balancer concentrate or one of the balancers plus a balanced feed. The mineral requirements (P, Ca, Mg) not covered by the basal diet and supplements is supplemented with mineral concentrate (choice of two mixtures, rich in P or Ca). The total annual consumption of each component of the diet as well as energy, protein and minerals can be summed up. The excretion is calculated as consumption minus retention in milk and animal growth.

Compared to the version of the model used for previous excretion calculations, the current version was updated to fully reflect the current state of knowledge and current SFRR. One major update are the new recommendations on mineral requirements of dairy cows that were released in April 2015. Especially for P, they will tend to reduce the excretion values. However, this is probably partly compensated by an increase of the P content in some herbages.

3. Results

The first preliminary results of the calculated excretions of dairy cows will be shown exclusively at the RAMIRAN conference.

In general (not only for dairy cows), the excretion values for N, P, K, Mg, Ca plus the annual forage consumption of the main categories of cattle, pigs, poultry, horses and small ruminants (sheep, goats) will be compiled in one big table. For major influencing factors like the milk yield of dairy cows or the protein and P content of pig diets, footnotes will give instructions how excretions can be adapted. The main assumptions used for the calculation of the excretion values (definition of category, productivity etc. will be documented in additional footnotes or in a special annex.

4. Conclusions and outlook

The level of detail of the approach used to calculate the new standard values for the nutrient excretions of dairy cows might look a bit too elaborated. However, as the values are quite important for policy implementation (compulsory N and P balance) they have to be reliable and robust. It must also be mentioned that the same level of detail will not be possible and necessary for all livestock categories. The use for policy implementation is also responsible for the fact that this information is still provided in the form of standard values rather than as a flexible calculation software.

References

[1] Agroscope, (2009). Grundlagen für die Düngung im Acker- und Futterbau (Fertilizer Guidelines; also available in French). Agrarforschung 16 (2), 1-100

[2] Agroscope, (2015). Fütterungsempfehlungen für Wiederkäuer (Feeding Recommendations for ruminants (SFRR); also available in French). Access: http://www.agroscope.admin.ch/futtermitteldatenbank/04834/index.html?lang=de. (5.7.2015)

[3] Kupper, T., Bonjour C., Menzi H., (2015). Evolution of farm and manure management and their influence on ammonia emissions from agriculture in Switzerland between 1990 and 2010. Atmospheric Environment 103, 215-221

TA-P_12 Changes to the nutrient contents of pig and poultry manures in England and Wales

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1. Objectives

In England, Wales and Northern Ireland, advice for farmers on the typical nutrient contents of livestock manures is published in the 8th edition of Defra's "Fertiliser Manual (RB209)" [1]. The underlying data for supporting the nutrient content figures were mainly derived from samples of stored slurry, solid manures and dirty water (from cattle, pig, poultry and sheep units) collected between April 2000 and December 2002, which were analysed for a range of chemical properties including dry matter (DM), total nitrogen (N), phosphorus (P), potassium (K), sulphur (S), magnesium (Mg), ammonium-N, nitrate-N and uric acid N (poultry manures only). However, these data on livestock manure composition may now be out of date due to changes to pig and poultry genotypes, diets, and housing and manure management systems which have occurred since the original data were collected.

The aim of this study was to ensure that advisory data on the 'typical' N, P, K, S and Mg concentrations of pig and poultry manures are up-to-date and representative of current farm practice, so that the published guidance continues to provide accurate and relevant nutrient management advice for farmers and growers.

2. Methodology

A targeted sampling scheme was undertaken to provide updated information on the nutrient concentrations of pig and poultry manures. Samples of pig slurry and farmyard manure (FYM), layer manure, broiler litter, turkey litter and duck FYM were collected throughout England and Wales during 2014 (112 samples in total). The sample distribution broadly reflected the overall contribution of each manure type to nutrient inputs to agricultural land in England and Wales, with the sampling programme targeted in areas where pig and poultry production is most common. Samples were collected using well established methodologies to ensure that representative samples of the manures were collected and additional information about each sample was recorded (e.g. type and age of livestock, diet, storage method, length of storage, etc.) to aid with the interpretation of the data. All samples were analysed for DM, total N, ammonium-N, nitrate-N, uric acid-N (poultry manures only), and total P, K, Mg and S, using standard methods.

These data were combined with supplementary data collected during the course of other recent research projects (181 samples) to provide a comprehensive and up to date manure analysis database for pig and poultry manures. Where possible, the data were analysed using two-sample *t*-tests to determine whether there were significant differences between the nutrient contents of the current data and the old (2000-2002) database for each manure type. This allowed us to determine whether the results from the sampling programme would justify an update of the 'typical' manure nutrient contents used in the guidance documentation.

3. Results and discussion

3.1 Pig manure

For pig slurry, the concentrations of all nutrients were significantly related to the slurry DM content (e.g. Figure 1), although there was a weaker relationship between DM and K concentrations because most K is dissolved in the liquid fraction of the slurry. Revised nutrient values for pig slurry were derived based on the 'best fit' relationships between DM and nutrient concentrations (Table 1 and 2).

Compared with current guidance, pig slurry total N and readily available N (RAN i.e. NO_3-N and NH_4-N) concentrations have increased by a small amount (from 3.6 to 4.1 kg/t total N and from 2.5 to 2.7 kg/t RAN for a 4% DM slurry), whilst phosphate concentrations have decreased (from 1.8 to 1.6 kg/t for a 4% DM slurry) reflecting the widespread use of dietary phytase (currently used in over 70% of pig diets [2]). Concentrations of the other nutrients in pig slurry were very similar to current guidance values (Tables 1 and 2).

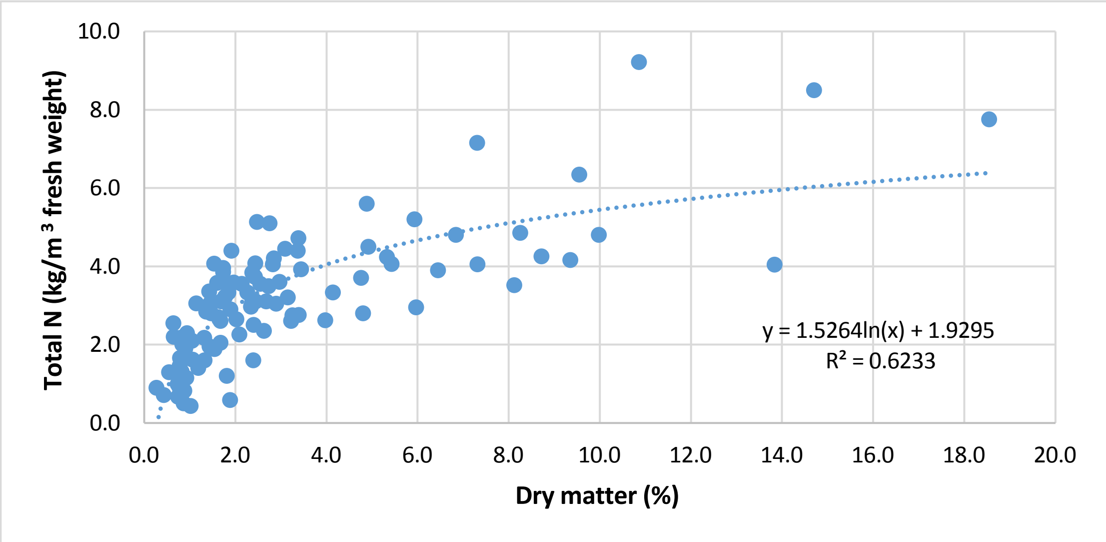

Figure 1: Relationship between dry matter and total N concentrations for pig slurry

Table 1: Current [1] and suggested revised total and readily available N (RAN) contents for pig slurry.

Dry matter (%)	Total N (kg/m^3 FW)		RAN (kg/m^3 FW)	
	Current	Revised	Current	Revised
2	3.0	3.0	2.2	2.2
4	3.6	4.1	2.5	2.7
6	4.4	4.7	2.8	2.9
8	-	5.1	-	3.1

FW: fresh weight

Table 2: Current [1] and suggested revised nutrient contents for pig slurry.

Dry matter (%)	P_2O_5 (kg/m^3 FW)		K_2O (kg/m^3 FW)		SO_3 (kg/m^3 FW)		MgO (kg/m^3 FW)	
	Current	Revised	Current	Revised	Current	Revised	Current	Revised
2	1.0	0.6	2.0	2.1	0.7	0.5	0.4	0.3
4	1.8	1.6	2.4	2.5	1.0	0.9	0.7	0.6
6	2.6	2.5	2.8	2.8	1.2	1.3	1.0	0.9
8	-	3.5	-	3.1	-	1.7	-	1.3

FW: fresh weight

There were no changes to the DM content or nutrient concentrations of pig FYM (P>0.05) despite recent reductions in dietary crude protein contents and the increased use of dietary phytase, which might have been expected to lead to reduced N and P excretion.

3.2 Poultry manures

The mean dry matter content of layer manure increased from 35 to 48%. British poultry farmers invested heavily to ensure they were compliant with the EU Laying Hens Directive (1999/74/EC) that came into force in 2012. This banned the use of barren battery cages and introduced the enriched or colony cage which is designed to hold up to 90 birds and give the hens more freedom. As a result, there has been a reduction in the numbers of older deep-pit housing systems (i.e. where manure is collected over a period of months in a pit below the house) to newer houses with belt removal systems (i.e. where manure is collected on moving belts below the cages and removed daily), many of which dry the manure and remove it from the house at regular (often daily) intervals. Manure from belt systems is drier than that from the old deep pit houses, thus ammonia emissions during storage are likely to be lower which in turn would be expected to increase the total N content of the manure. More laying hens are also reared on deep litter systems (with or without forced drying of the manure) or in an aviary system with an outside range or scratching area.

For both layer manures and broiler/turkey litter a greater proportion of the RAN was present in the form of uric acid-N which may decrease ammonia losses during housing and manure storage, and following land spreading of manure. The increased uric-acid N content and reduced ammonia losses are also likely to affect manure N use efficiency.

The P concentration of broiler/turkey litter has significantly decreased following the introduction of dietary phytase; however, K and Mg concentrations have significantly increased. Potash concentrations may have increased (from 18 to 21 kg/t) because of the use of higher protein feeds to meet the requirements for higher amino acid specifications which are needed for modern poultry genotypes. This leads to more soya being used in feed than previously, with many companies reducing or excluding fishmeal from their formulations in response to customer needs. Soya is high in K so both of these changes will have tended to increase K concentrations in the feed and hence in the excreta. It is not clear why Mg concentrations have also increased (from 4.4 to 5.8 kg/t) as Mg is not routinely added to poultry diets, although it may be related to changes in feed digestibility.

When the data on layer manure and broiler/turkey litter were combined it was clear that nutrient contents were related to manure DM content, as has been previously reported [3]. There were strong and highly significant (r^2= 0.58); p<0.001) linear relationships between poultry manure DM contents and the total N (Figure 2a), P, K, S and Mg contents, with no distinction between the relationships for layer manure and broiler litter. The relationship between poultry manure DM and RAN concentration was weaker (r^2= 0.14; p<0.01), although when RAN was expressed as a percentage of the total N content, there was a significant (r^2= 0.28; p<0.001) inverse relationship with DM content (Figure 2b).

a) b)

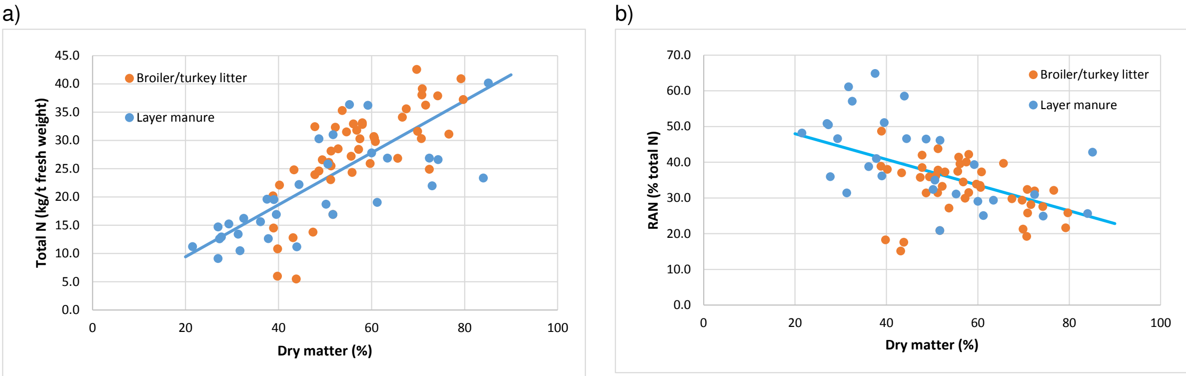

Figure 2: Relationships between dry matter and a) total N and b) RAN concentrations for poultry manures

Table 3: Current [1] and suggested revised total and readily available N (RAN) contents for poultry manures.

Dry matter (%)	Total N (kg/t FW)		RAN (kg/t FW)	
	Current	Revised	Current	Revised
20	-	9.4	-	4.5
35 (layer)[1]	19	-	9.5	-
40	-	19	-	7.8
60 (broiler)[2]	30	28	10.5	9.4
80	-	37	-	9.8

[1]Currently a typical dry matter content of 35% for layer manure is given [1]
[2]Currently a typical dry matter content of 60% for broiler litter is given [1]
FW: fresh weight

Table 4: Current [1] and suggested revised nutrient contents for poultry manures.

Dry matter (%)	P_2O_5 (kg/t FW)		K_2O (kg/t FW)		SO_3 (kg/t FW)		MgO (kg/t FW)	
	Current	Revised	Current	Revised	Current	Revised	Current	Revised
20	-	8.0	-	8.5	-	3.0	-	2.7
35 (layer)[1]	14	-	9.5	-	4.0	-	2.6	-
40	-	12	-	15	-	5.6	-	4.3
60 (broiler)[2]	25	17	18	21	8.0	8.2	4.4	5.9
80	-	21	-	27	-	11	-	7.5

[1]Currently a typical dry matter content of 35% for layer manure is given [1]
[2]Currently a typical dry matter content of 60% for broiler litter is given [1]
FW: fresh weight

These findings suggest that standard figures for poultry manure (layer manure, and broiler and turkey litter) could be combined and given for a typical range of dry matter contents as is currently the case for slurries. Suggested values are presented in Tables 3 and 4.

4. Conclusion and outlook

The likely financial, agronomic and environmental impacts of revising ‘typical’ nutrient values depends on the magnitude of the change to the mean nutrient contents of pig and poultry manures in relation to current typical values in the published guidance. Data from this study have indicated that there have been some changes to the mean ‘typical’ manure dry matter and nutrient concentrations. Most notably, for poultry manures the phosphate concentration has decreased whilst the K and Mg concentrations have increased; there have also been some increases in the total N concentration of pig slurry The nutrient concentrations in pig FYM were not significantly different from the previously published guidance values.

Typical manure nutrient contents are used to calculate manure nutrient supply and adjust manufactured fertiliser applications to meet crop demand. The manure analysis data are also used by farmers and regulators to show compliance with Nitrate Vulnerable Zone (NVZ) Action Plan requirements. Accurate information on manure nutrient content is essential to maximise nutrient use efficiency, farm profitability (through avoiding unnecessary fertiliser applications or reductions in crop yield and quality) and minimise the risks of nutrient losses to air (i.e. ammonia and nitrous oxide) and water (i.e. nitrate-N, ammonium-N and phosphorus). Under-estimating the nutrient content of livestock manures could lead to nutrient applications in excess of crop demand, increasing the risk to crop yields and quality including lodging in cereals, elevated amino-N concentrations in sugar beet, high grain N content of barley, excessive ammonia concentrations in grass silage and reduced tuber numbers in potato crops. Excessive nutrient applications will increase the risks of ammonia and nitrous oxide emissions to air and nitrate-N, ammonium-N and P losses to water. Over-estimating manure nutrient contents may result in nutrient applications that are insufficient to achieve optimum yields reducing farm incomes and food production.

Further work is now required to quantify the effect of elevated uric-acid N content of poultry manures on manure N use efficiency and on ammonia emissions during housing and manure storage and following application to land. The information will be important to meet the requirements of future revisions of the NVZ Action programme and to support national policy to meet the requirements of the National Emissions Ceilings Directive.

Acknowledgements

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References

[1] Anon. (2010). The Fertiliser Manual (RB209). 8th Edition. The Stationary Office, Norwich.
[2] Partridge, G. (2014). Formulating cost effective pig diets with an advanced phytase. International Pig Topics, Vol. 29, No.2, 24-25
[3] Nicholson, F. A., Chambers, B. J & Smith, K.A. (1996). Nutrient composition of poultry manures in England and Wales. Bioresource Technology, 58, 279-284.

TA-P_13 Phosphorus availability in the tropical soils of Reunion – Comparison of various methods

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1. Objectives

Effect of organic residues application on P availability in soils depends on soil properties [1]. Phosphorus availability in the soils of Reunion (a French tropical island in the Indian Ocean) has been poorly studied [2]. The objectives are:

- To determine P availability in a wide range of soils of Reunion supplied or not with various mineral and organic fertilizers
- To evaluate the relevance of 3 soil P tests to distinguish P availability as a function of soil types and fertilization management

2. Methodology

Organic, mineral or no fertilization were applied on 5 field trials, encompassing the main soils types (i.e. hydric andosol, chromic andosol, andic cambisol, nitisol and arenosol), and cropping systems (fodder, sugar cane, market garden crops) of Reunion. Then, each soil type had different fertilization managements and have received different amount of P.

Phosphorus availability was measured in fifty soil samples collected in these trials with three extraction methods (table 1):

- $CaCl_2$ (0.001M) extraction that mimics soil solution,
- Olsen extraction [3] that targets the moderately available P pool. Olsen method is a soil P test widely used to assess P availability.
- Olsen-Dabin extraction [4] that targets the weakly available P pool. Olsen-Dabin method is the soil P test used in Reunion Island to assess P availability.

The molybdate reactive P concentration in the extracts was determined by colorimetry, using the malachite green method [5] for $CaCl_2$ and Olsen extracts and the molybdate blue method [6] for the Olsen-Dabin extracts. Malachite green method was not used for Olsen-Dabin extracts because of the interference of reactive fluoride with the coloration. We considered the molybdate reactive P as the inorganic P (Pi), even if a small part of the organic P can react with the molybdate [7].

Table 1: Methods used

Extraction method	Chemical extractant	Extraction time	S:L $g.mL^{-1}$	Colorimetric method
$CaCl_2$	0.001M $CaCl_2$	24h	1:10	Malachite green
Olsen	0.5M $NaHCO_3$ at pH 8.5	30min	1:20	Malachite green
Olsen-Dabin	0.5M $NaHCO_3$ + 0.5N NH_4F at pH 8.5	1h	1:35	Molybdate blue

S:L ratio soil:liquid

3. Results and discussion

3.1 Correlation between soil P tests changes with soil types

The three soil P tests yielded to very different concentrations of Pi extracted: in average 3.25 ± 1.55 $mg.kg^{-1}$, 52.87 ± 20.84 $mg.kg^{-1}$ and 307.98 ± 92.50 $mg.kg^{-1}$ for $CaCl_2$, Olsen and Olsen-Dabin methods respectively (table 2). This was expected, since $CaCl_2$ is the least aggressive extractant, with a Pi concentration supposed to be close to the concentration of the soil solution. Olsen-Dabin method yielded to a larger concentration of Pi extracted than Olsen method. The fluoride ions contained in the Olsen-Dabin extractant can explain this higher concentration, because they release more P, especially from $Al\text{-}PO_4$, than $NaHCO_3$ alone. The longer extraction time for Olsen-Dabin method can moreover explain this difference. The very large concentration of

Pi extracted with Olsen-Dabin method (up to 1268.22 mg.kg^{-1} in a sample of andic cambisol) show that this method extracts a high part of P strongly sorbed onto the soil.

Concentrations of Pi extracted changes with soil type, but the order of soil types according to their concentration of Pi extracted is different between the three methods (table 2): with $CaCl_2$, the concentration of Pi was the highest in the arenosol (mean: 8.76 ± 2.23 mg.kg^{-1}) and the lowest in the chromic andosol (mean: 0.56 ± 0.21 mg.kg^{-1}). For seven samples, the P $CaCl_2$ was below the quantification limit (0.019 mg.kg^{-1}). With the Olsen and Olsen-Dabin methods, the highest Pi concentrations were obtained for the andic cambisol, but with Olsen the lowest Pi concentration were obtained for the chromic andosol; while it was obtained for the nitisol with the Olsen-Dabin method.

Table2: Average concentration of Pi extracted with the 3 methods in each soil type

Soil type	P $CaCl_2$	P Olsen	P Olsen-Dabin
		mg.kg^{-1}	
Andic cambisol	4.49 ± 1.27	133.05 ± 19.18	648.69 ± 78.76
Hydric andosol	1.73 ± 0.36	30.17 ± 2.49	322.21 ± 25.78
Chromic andosol	*0.56 ± 0.21	13.43 ± 1.23	263.38 ± 18.85
Arenosol	8.76 ± 2.23	47.08 ± 5.08	201.25 ± 21.39
Nitisol	0.72 ± 0.08	40.61 ± 1.34	104.36 ± 7.14
All soils	3.25 ± 1.55	52.87 ± 20.84	307.98 ± 92.50

P $CaCl_2$, P Olsen, P Olsen-Dabin: concentration of Pi extracted with $CaCl_2$, Olsen and Olsen-Dabin methods

*For the calculation of the mean, some concentrations were below the quantification limit

P $CaCl_2$ correlation with P Olsen or P Olsen-Dabin changes with soil type: P $CaCl_2$ is poorly correlated with P Olsen or P Olsen-Dabin for all soils together, with global coefficients of determination (R^2) equal to 0.27 and 0.16, respectively. However, the correlations between P $CaCl_2$ and P Olsen or P Olsen-Dabin according to the soil type are high for two soils: the arenosol and andic cambisol (R^2 from 0.75 to 0.87). In these two soils, the relationship between P $CaCl_2$ and P Olsen or P Olsen-Dabin is flat until a threshold and then P $CaCl_2$ increases linearly with P Olsen or P Olsen-Dabin. This sudden change of slope can be explained by the saturation of soil P sites. For the three others soils, the correlation between P $CaCl_2$ and P Olsen or P Olsen-Dabin are low (R^2 from 0.08 to 0.58) and P $CaCl_2$ remains stable with augmentation of P Olsen or P Olsen Dabin.

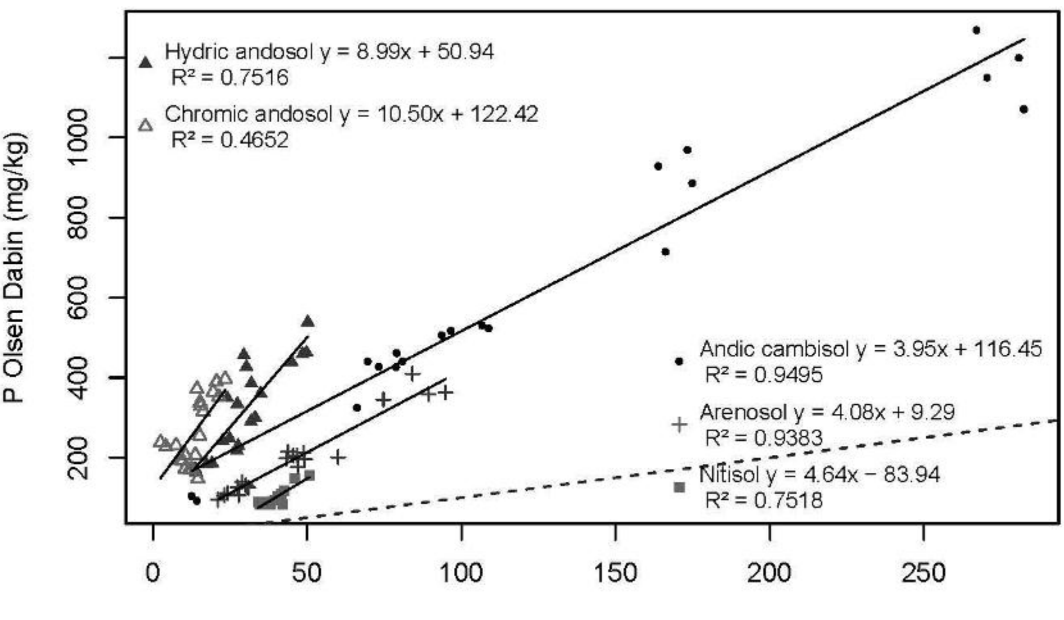

Figure 1: Linear regressions between P Olsen-Dabin and P Olsen for each soil type. Lines represent the linear regressions, their equations and coefficients of determination (R^2) are given in the legend. Dotted line: line 1:1.

P Olsen-Dabin and P Olsen relationship changes with soil types: P Olsen and P Olsen-Dabin are positively correlated for all soils together, with a coefficient of determination (R^2) equal to 0.79. This is logical since both extractants have a close composition. Linear regressions between P Olsen-Dabin and P Olsen according to the soil type have acceptable coefficients of determination (R^2 from 0.75 to 0.94) except for the chromic andosol (R^2: 0.47) (figure 1). The slopes of regression lines are close on the one hand for the andic cambisol, the arenosol and the nitisol (average slope: 4.22 ±.0.21). On the other hand, the slopes are close and higher for the two andosols (average slope: 9.75 ± 0.76). These results show that the use of Olsen-Dabin instead of Olsen method increased more the concentration of Pi extracted in the andosols than in the other soils, that's why the order of soil types are different with the two methods. As fluoride contained in Olsen-Dabin extractant released preferentially P from $Al\text{-}PO_4$, the andosols have probably a high part of P sorbed to Al. This pool of P, which is not extracted with the Olsen method, is poorly available.

3.2 Relevance of soil P tests to distinguish samples according to the soil type and the fertilization management

$CaCl_2$ extraction shows an interaction between soil type and fertilization management: For high amount of P applied, fertilization affected P $CaCl_2$ in the arenosol, the andic cambisol, and at a lower rate in the hydric andosol. In the arenosol, the P $CaCl_2$ in the sample that received an organic fertilization at a high rate was 12 times higher than in the control without fertilization (30.70 $mg.kg^{-}1$ and 2.42 $mg.kg^{-1}$ respectively). In contrast, fertilization did not affect the P $CaCl_2$ in the chromic andosol and in the nitisol (figure 2a). The nitisol samples have just been amended for one year, that's why no difference with fertilization was expected. All other soil samples have been amended for 10 years. Even with the organic fertilization at the highest rate, which represented a supply of 220 $kgP.ha^{-1}.year^{-1}$, andosols show a small P $CaCl_2$. This highlights the high P sorption capacity of the andosols [2]. P $CaCl_{2,}$ is indeed an intensity parameter supposed to increase when P sorption sites of the soil are saturated, that's why there is a high interaction between soil type and fertilization management.

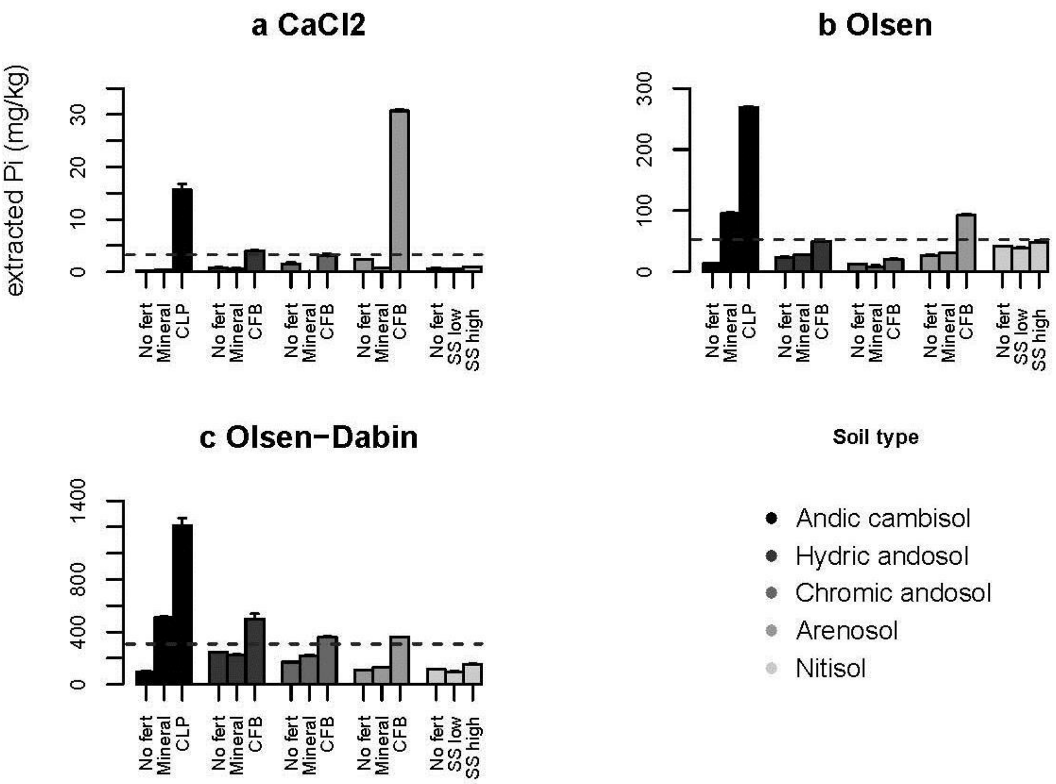

Figure 2: Concentration of Pi extracted with $CaCl_2$ (a), Olsen (b) and Olsen-Dabin (c) methods in the 5 soil types. For each soil type, 3 modalities of fertilization are shown, that represent increasing amount of P applied: for all soil types, control with no fertilization (No fert). For the andic cambisol: NPK fertilization (Mineral) and composted pig manure at high rate (CLP). For the 2 andosols and the arenosol: NPK fertilization (Mineral) and cow manure at high rate (CFB). For the Nitisol, sewage sludge at low rate (SS low) and sewage sludge at high rate (SS high). Dotted line: mean for all soils.

P $CaCl_2$ seems to differ between soil types (table 2), but these variations are probably due to this interaction between soil and fertilization management. $CaCl_2$ extraction is a soil P test used to approximate the concentration in the soil solution, but amount of P extracted with this method is often poorly correlated with P available to plant [8].

Olsen extraction shows effects of soil type and fertilization management: Fertilization management affected P Olsen particularly in the andic cambisol, and at a lower rate in the arenosol and the hydric andosol (figure 2b). Soil type and fertilization had effects on P Olsen. Olsen extraction is a soil P test frequently used to assess P phytoavailability, but its relevance varies with soil types. In tropical soils, a previous study showed that the correlation between P Olsen and P uptake by plant varied from 50% to 100% according to the soils [9].

Olsen-Dabin extraction shows an effect of fertilization management whatever the soil type: Fertilization management affected P Olsen-Dabin particularly in the andic cambisol, and at a lower rate in the arenosol and the two andosols (figure 2c). Andosol samples show in average higher P Olsen-Dabin than arenosol samples (table 2) for the same fertilization managements. Since andosols have a higher sorption capacity than arenosols, results show that Olsen-Dabin method extracted a high part of P from a weakly available P pool. That's why P Olsen-Dabin could probably not reflect the P phytoavailable in all soil types, especially in andosols. Morel et al. (1987) showed that the Olsen-Dabin method extracted a high amount of P that was not available for plant in some tropical soils [9].

4. Conclusion and outlook

Olsen-Dabin extraction, which is the traditionally soil P test used in Reunion to fit P fertilization to crop requirements, extracted large concentrations of Pi, with a high part weekly available. $CaCl_2$ method extracted small concentrations of Pi, and is affected by the interaction between soil type and fertilization management. Andosols showed small amount of Pi extracted with $CaCl_2$ even in samples that received an organic amendment at a high rate for 10 years. Then, $CaCl_2$ extraction is not usable for the most representative soil of Reunion, i.e. andosols. Olsen extraction varied with both soil type and fertilization management. The Olsen method could then be used to discriminate the respective contribution of soil types and organic fertilization on P availability in tropical soils of Reunion. The next steps of our study are to test one other method: the DGT (diffusive gradient in thin films) technique, that fits well with P phytoavailable in some tropical soils [8], and then to compare the P available assessed with each method and the P concentration in plants.

References

[1] Shafqat, M.N.; Pierzynsky, G.M. (2013): The effect of various sources and dose of phosphorus on residual soil test phosphorus in different soils, Catena 105, 21-28

[2] Truong, B.; Bertrand, R.; Burdin, S.; Pichot, J. (1974): Contribution to the study of phosphorus in the soils derived from volcanic rocks in Reunion Island (Mascareignes). Effect of calcium carbonate and siliacte, L'agronomie tropicale 29, 663-674

[3] Olsen, S.R.; Cole, C.V.; Wanatabe, F.S.; Dean, L.A. (1954): Estimation of available phosphorus in soils by extraction with sodium bicarbonate. US Dep of Agric Circ No. 939

[4] Dabin, B. (1967): Sur une méthode d'analyse du phosphore dans les sols tropicaux. Colloque sur la fertilité des sols tropicaux. Tananarive (Madagascar), Nov. 1967, p.99-115. I.R.A.T.

[5] Ohno, T.; Zibilske, L.M. (1991). Determination of low concentrations of phosphorus in soil extracts using malachite green, Soil Science Society of American Journal 55,892-895

[6] NF ISO 11263 (1995) - Soil quality. Determination of phosphorus. Spectrometric determination of phosphorus in sodium hydrogen carbonate solution

[7] Van Moorleghem, C.; Six, L.; Degryse, F.; Smolders, E.; Merckx, R. (2011): Effect of organic P forms and P present in inorganic colloids on the determination of dissolved P in environmental samples by the diffusive gradient in thin films technique, ion chromatography, and colorimetry, Analytical Chemistry 83, 5317–5323

[8] Six, L.; Pypers, P.; Degryse, F.; Smolders, E.; Merckx, R. (2012): The performance of DGT versus conventional soil phosphorus tests in tropical soils - An isotope dilution study, Plant and Soil 359, 267–279

[9] Morel, C.; Fardeau, J.C. (1987): Available phosphorus in intertropical soils: its relation with phosphorus determined by two chemical extraction methods, L'Agronomie Tropicale 42, 248-257

TA-P_14 Utilization of precipitated phosphorous in plant production

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1. Objectives

The aim was to find out whether plants can utilise phosphorus from the sludge collected from agricultural drainage water. The water treatment method developed by Saloy Ltd uses iron sulphate to bind organic matter and phosphorus from drainage water in an emission catcher. A vertical separation pipeline connected to the sediment pond reduces the amount of suspended solids in the water. The pipeline also operates as a sediment pond for the sludge precipitated by the emission catcher.

2. Methodology

Sludge from two sources (Pien-Saimaa and Sauvo) were tested at Häme University of Applied Sciences (HAMK) during summer 2014 in a growth tunnel. The experimental species was a barley variety ‘Barke’. Sand used as a substrate in the golf courses was used as reference. There was a total number of six treatments (table 1). The fertilizer used was Kekkilä Garden’s Puutarhan yleislannoite.

Table 1: Volumes of sludge and sand in the six treatments are presented in the table. Also addition of fertilizer is stated. Calcium was added to all treatments. Numbering of the treatments is used later in figures.

Treatment	Sludge		Sand		Fertilizer	Calcium
	Sauvo 20%	Pien-Saimaa 20%	80%	100%	0,5 ‰	3 kg/m^3
1	x		x		x	x
2		x	x		x	x
3	x		x			x
4		x	x			x
5				x	x	x
6				x		x

Barley was sown into 10 litre pots filled with mixed substrate. The pots were placed into a growth tunnel at the HAMK nursery. The experiment was covered in order to control the amount of water, but the sides of the tunnel were kept open. There were six replications, experimental design was a Latin square with protection rows that were not measured.

The pots were watered once or twice a week, depending on the weather. An excess of watering was done three times during the summer in order to cause enough drainage for nutrient analyses. The nutrient content of the substrate was analysed also from the substrate in the beginning and at the end of the experiment. The total plant biomass and the yield was measured at the end of the experiment.

3. Results and discussion

3.1 Phosphorous

The results show a huge difference between the two tested sludge, Pien-Saimaa and Sauvo. The phosphorous content from the drainage of the treatments with Pien-Saimaa sludge decreased most during the summer (treatments 2 and 4 in Figure 1). Phosphorus is known to bind to iron. The biggest iron contents were observed in the drainage analyses of Pien-Saimaa sludge. It seems that Pien-Saimaa sludge as such easily releases nutrients. There were no observations of an increase in pH during the summer. When slowly soluble fertilizer is added the phosphorous content actually increases during the summer in the reference treatments 6 with sand and fertilizer (Figure 1).

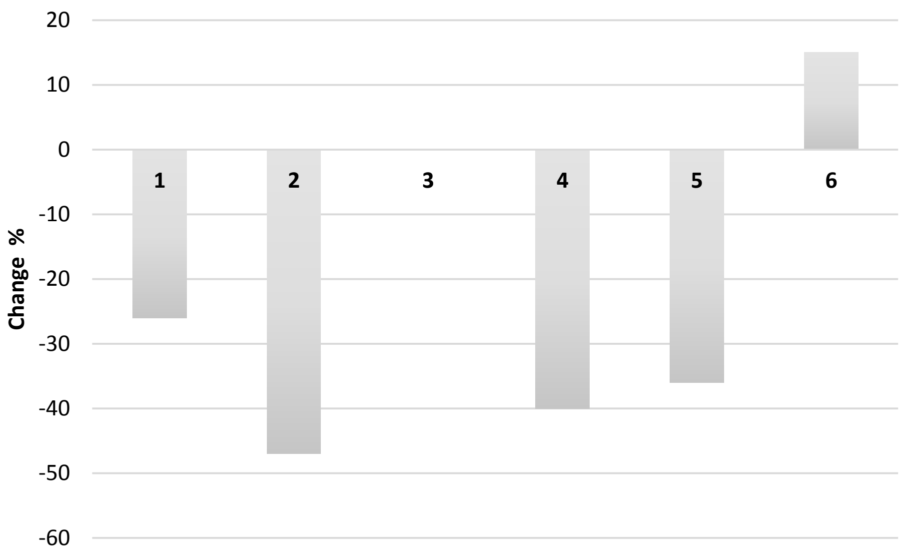

Figure 1: Change in the phosphorous contents of the treatments. For codes on x-axis, see table 1.

3.2 Plant growth

Barley was observed to grow without symptoms of over dosing or low nutrient content. The fresh weight of straws in different treatments is presented in figure 2. The highest weight of the straw was measured in the Pien-Saimaa treatment number 2 with added commercial fertilizer. There was no significant difference between Pien-Saimaa treatment number 2 and the control treatment number 5 with commercial fertilizer (df=5, F=1,2, p=ns). Both treatments that contained sludge from Sauvo resulted in lower straw fresh weight than the control (Treatment 1: df=5, F=38,1, p=0,02. Treatment 3: df=5, F=53,7, p=0,001).

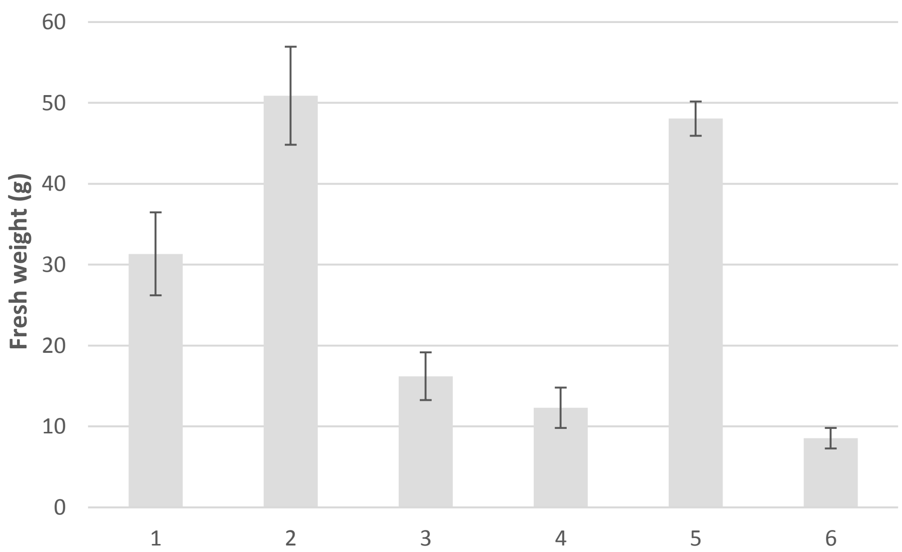

Figure 2: Fresh weight of the straw biomassa of the treatments. For codes on x-axis, see table 1.

The total yield was measured by counting the grains and weighing the total mass. There was great variation in the grain weight (figure 3), but again the overall trend showed the highest yield for Pien-Saimaa treatment number 2 and fertilized controls. However, the only significant difference was between unfertilized sludge (treatments number 1 and 2) and fertilized control treatment 5): (1 vs.5: df=5, F=48,7, p=0,01. 2 vs.5: df=5, F=27,0, p=0,003).

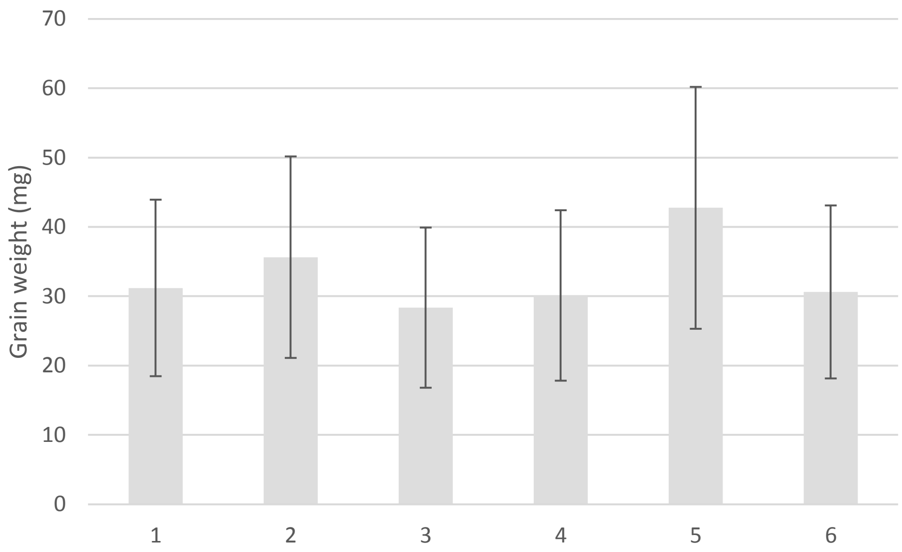

Figure 3: Average weight of barley grains in different sludge treatments. For codes on x-axis, see table 1.

4. Conclusion and outlook

The results show that the method used by Saloy Ltd for phosphorous catchment produces sludge that does not reduce plant growth. However, it needs to be investigated further whether the sludge could act as a source of phosphorus.

There is a great need to find new ways to recirculate phosphorus. Agricultural production is dependent on phosphorus, but today many of the world's remaining reserves will be depleted during the next decades (Pearce 2011). The excess of phosphorous use in the past is bound to various organic matters causing serious eutrophication of lakes and seas around the world.

The Baltic Sea Agreement from 2010 also binds Finland to improve the state of the Baltic Sea markedly until the year 2020. An important task is to diminish the amount of nutrients, especially nitrogen and phosphorous, in run-off waters and also to promote research and development work in order to find new innovations to recirculate nutrients.

A work-group set by the Finland's Ministry of Agriculture and Forestry has described four main themes for achieving the goal:

1. Sustainable use of nutrients;
2. Decrease of the amount of organic waste and nutrients in it;
3. Effective and safe recirculation of nutrients; and
4. Harvest of nutrients from waters and recirculating to agricultural use.

Sustainable primary production and the recirculation of nutrients are also the focus areas in the future for the Research Unit of Bioeconomy at Häme university of Applied Sciences.

References

[1] Pearcce, F. (2011): Phosphate: A Critical Resource Misused and Now Running Low, Environment 360. http://e360.yale.edu/feature/phosphate_a_critical_resource_misused_and_now_running_out/2423/

TA-P_15 Acidification of cattle slurry – Effect on ammonia emissions and nitrogen fertilizer replacement value of spreading non-digested and digested slurry on grassland

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1. Objectives

The objective of the project was to determine the effect on NH_3 emissions and dry matter yield of acidification of digested and non-digested slurry, applied on two different occasions to grassland. The overall aim was to help develop future guidelines on whether acidification should be recommended for Swedish conditions from an environmental and economic perspective.

2. Methodology

Field experiments were conducted on a grass ley at two Swedish sites in 2013 (Rådde and Bjertorp) and at one site in 2014 (Lanna), all approximately 100 km north-east of Gothenburg. The effect on grass yield and nitrogen (N) offtake of anaerobically digested and non-digested cattle slurry, with and without acidification, was compared with the effect of a mineral N fertiliser treatment. Slurry was band-spread with trailing hoses on two occasions in each experiment, after the first and second cut in 2013 and in spring and after the first cut in 2014. The experiments were organised in a randomised complete block design. Treatments without acidification were applied first, followed by acidified slurry treatments. Actual rates are shown in Table 1. Sulphuric acid was added with a dosage device just before spreading to achieve pH below 6, which required 2-3 L m^{-3} for non-digested slurry and 6 L m^{-3} for digested slurry (Table 2). Dry matter yield and forage N content were measured for two cuts. Nitrogen fertiliser replacement value (NFRV) for total N in digested and non-digested cattle slurry with and without acidification was calculated from the respective effect on N offtake in relation to that of mineral fertiliser.

Ammonia emissions were measured in the experiment at Lanna with an equilibrium concentration method [1] in spring and after first cut in three blocks, with repeated measurements (3-5) until the emissions subsided. In each small plot, two ventilated chambers were used to estimate the equilibrium concentration and one device took ambient samples.

3. Results and discussion

3.1 Slurry properties

The properties of the cattle slurry, non-digested and digested, are shown in Table 1. The proportion of ammonium-nitrogen (NH_4-N) in total N (Tot-N) was higher and the C/N ratio lower in digested slurry than in undigested slurry. At the start, the pH was around 7.4-7.9 for all slurry types and spreading times, and after acidification it was just below 6.

Table 1: Properties of non-digested and digested slurry with no acid added used in spring and summer treatments

Slurry type	DM content (%)	pH	Tot-N	NH_4-N	P	K	S	Tot-C	C/N ratio
			(kg tonne^{-1})						
Spring									
Cattle slurry	8.5	7.6	4.1	2.1	0.42	2.3	0.31	39.6	9.7
Cattle slurry, digested	4.2	7.4	3.7	2.5	0.31	2.2	0.32	18.0	4.9
Summer									
Cattle slurry	6.9	7.9	3.4	1.7	0.34	2.4	0.32	32.2	9.5
Cattle slurry, digested	3.8	7.4	3.6	2.5	0.30	2.2	0.29	15.9	4.4

DM-dry matter; Tot-N-total nitrogen; NH_4-N-ammonium nitrogen; P-phosphorus; K-potassium; S-sulphur; Tot-C-total carbon; C/N ratio-ratio of Tot-C and Tot-N

3.2 Ammonia emissions

Overall, acidification reduced NH_3 emissions (Table 2). In spring the reduction was significant ($p<0.05$), about 50% for undigested slurry, but was even more pronounced for the digested slurry, with more than 90% reduction. In summer, the emissions were slightly lower than in spring. Acidification also lowered ammonia emissions compared with no acid addition, by about 75% for cattle slurry and more than 90% for digested slurry. This was not statistically verifiable, as significant interactions between treatment and block excluded pairwise comparisons. The higher amount of acid added to the digested slurry and differing buffer capacity of digested and non-digested slurry may have contributed to the better effect of acidification on digested than non-digested slurry. The efficiency of acidification of animal slurry in minimising NH_3 emissions has been documented in previous studies [2].

Table 2: Application data for the different slurry types, acidified or not, and ammonia emissions per ha and as percentage of NH_4-N and Tot-N applied

Type of slurry	Acid, L tonne^{-1}	Application rate: Tonne ha^{-1}	Application rate: kg NH_4-N ha^{-1}	Application rate: kg Tot-N ha^{-1}	Ammonia emissions: kg N ha^{-1}	Ammonia emissions: % of NH_4-N applied	Ammonia emissions: % of kg Tot-N applied
Spring application							
Cattle slurry	0	37*	78	152	71.9	92.5^{a}	47.4^{a}
Cattle slurry	1.7	24	50	98	22.7	45.1^{b}	23.1^{b}
Cattle slurry, digested	0	24	60	89	30.5	50.8^{b}	34.3^{ab}
Cattle slurry, digested	6.2	24	60	89	2.1	3.5^{c}	2.4^{c}
Early summer application^{A}							
Cattle slurry	0	25	42	85	17.5	41.2	20.6
Cattle slurry	3.0	25	42	85	4.3	10.2	5.1
Cattle slurry, digested	0	25	62	90	16.6	26.5	18.4
Cattle slurry, digested	6.0	25	62	90	1.1	1.7	1.2

$^{a, b, c}$ Means with different letters within each column and application time are significantly different ($p<0.05$).

^{A} Significant interaction between treatment and block, no pairwise comparisons made.

* High application rate due to technical problems with valve setting on slurry tanker.

3.3 Nitrogen offtake and NFRV

Dry matter yield and N offtake were greater in acidified treatments than in the corresponding treatments without acidification (Table 3). On average, NFRV increased with acidification from 21% to 35% in undigested cattle slurry and from 39% to 67% in digested slurry. At Lanna, where ammonia emissions were measured, in most cases NFRV increased on average by about 20% units of Tot-N applied when slurry was acidified, which corresponds to the mean reduction in ammonia emissions from acidification as percent units of Tot-N. However, for digested cattle slurry application in spring, the increase in NFRV was negligible and did not match the reduction in ammonia emissions.

Table 3: Nitrogen (N) offtake with harvested grass and nitrogen fertiliser replacement value (NFRV) in different treatments at the Bjertorp, Rådde and Lanna sites

Type of slurry, acid addition	Nitrogen offtake (kg N ha^{-1})			NFRV (% of total N)		
	Bjertorp	**Rådde**	**Lanna***	**Bjertorp**	**Rådde**	**Lanna***
Spring application						
Cattle slurry			45			25
Cattle slurry, acid added			49			45
Cattle slurry, digested			55			62
Cattle slurry, digested, acid added			56			63
Early summer application						
Cattle slurry	26	33	25	23	31	11
Cattle slurry, acid added	35	46	31	40	45	20
Cattle slurry, digested	40	48	31	49	52	18
Cattle slurry, digested, acid added	59	67	57	83	72	51
Late summer application						
Cattle slurry	7	24		7	29	
Cattle slurry, acid	10	32		17	42	
Cattle slurry, digested	12	22		23	29	
Cattle slurry, digested, acid added	22	41		61	72	

* Site where NH_3 measurements took place in 2014.

4. Conclusion and outlook

Ammonia emissions can be reduced and grass yield improved by acidification of cattle slurry. The effect on ammonia emissions is larger for anaerobically digested slurry than for non-digested slurry. For non-digested slurry, this study showed a reduction of 50% and 75% for spring and early summer application, respectively. For digested slurry the reduction in NH_3 emissions from acidification exceeded 90%, although at a higher acid dosage of 6 L m^3.
With both digestion and acidification, the nitrogen fertiliser replacement value can be about threefold that of untreated slurry and N offtake is around twice that of untreated slurry, due to high plant availability of N in combination with reduced ammonia losses.

Acknowledgements

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References

[1] Svensson, L. (1994): Ammonia volatilization following application of livestock manure to arable land, Journal of Agricultural Engineering Research 58(4), 241-260

[2] Fangueiro, D.; Hjort, M.; Gioelli, F. (2015): Acidification of animal slurry – a review, Journal of environmental management 149, 46-56

TA-P_16 Solubility of Copper and Zinc and particle size fractionation in compost made from the solid fraction of pig slurry

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1. Objectives

Certain heavy metals, such as Cu and Zn, are essential micronutrients for plant and crop growth. However, the high concentrations of these metals found in pig slurry, derived from their use as ZnO and $CuSO_4$ in feed additives (especially during pregnancy and in the post-weaning phase), and the fact that they remain mostly in the solid fraction after solid-liquid separation can be limiting factors for the use of this waste material to obtain high quality composts for agricultural use. The use of pig slurry derived composts in crop production (as soil amendment or organic fertiliser) may suppose a concern of metal contamination of soil, water and the surrounding environment. Therefore, the objective of this work was to study the influence of solid phase of pig slurry composting on the concentration and solubility of heavy metals, and their distribution in the different particle size fractions, and evaluating their potential toxic effects on plant germination and seedling growth.

2. Methodology

The solid fraction (FS) was obtained from a piglets and sows farm located in south-eastern Spain, after slurry separation by a screw-press (without flocculants). Two composting piles were prepared in the farm, mixing FS1 with cereal straw (Pile 1) and FS2 with cotton gin waste (Pile 2) (3:2 and 2:1 v:v ratios, respectively) by the turning pile system. The bio-oxidative phase of P1 lasted for 75 days and of P2 for 120 days, and the total composting times (including maturation period) were 170 and 187 days, respectively. During composting, the piles were turned three and five times, respectively, when the temperature started to decrease - to improve both the homogeneity of the materials and the composting process. Compost samples were taken by mixing seven subsamples from seven representative sites of the pile, from the whole profile (from the top to bottom of the pile). The original materials and compost were chemically analysed using standard methods. The mature composts were mechanically sieved at different particle sizes (mm): < 0.05; 0.05-0.5; 0.5-1.0; 1-2; and >2. The samples of compost and the different particle size fractions of mature compost were analysed for total Cu and Zn concentrations by microwave assisted acid digestion (HNO_3/H_2O_2) and for soluble and exchangeable Cu and Zn concentrations by 0.1 M $CaCl_2$ extraction, and all measured in an atomic absorption spectrometer (Thermo iCE 3000 series). The potential phytotoxicity of compost was evaluated through plant growth tests (ISO 15 799.1999) using different mixtures of mature compost and artificial soil (prepared according to OECD 207.1984), and seeds of *Zea mays*. For each treatment the growth index was expressed as a percentage of fresh weight (g per plant of the aerial part) with respect to the control (without compost). The EC_{50} and LC_{50} values were also calculated. The germination index (GI) was assessed using *Lepidium sativum* seeds.

3. Results and discussion

3.1. Evolution of metals during composting

The solid fractions obtained by screw press separation had high concentrations of Zn and Cu: 3098-3397and 249-269 mg kg^{-1}, respectively. The concentrations of Cu in the solid fraction of pig slurry were similar to those reported in other pig slurry separation studies [1], but the concentrations of Zn were higher in the present study, probably because the pig slurry came from piglets, since Zn is provided as feed additives to piglets in order to avoid digestion problems and to improve nutrients assimilation. A study with different growth stages of pigs indicated that Zn and Cu are excreted at relatively higher rates than other feed constituents [2] and over 90 % of Cu and Zn was reported to be retained in the solid phase in solid-liquid separation treatment with different technologies[3].

The total concentrations of Cu and Zn increased during composting (Table 1) due to the degradation of organic matter (OM) and the consequent weight loss of the pile. The increase was more evident in the earlier stage of composting, when OM degradation was highest (data not shown) [4]. In the present experiment, the concentrations of the metals in soluble and exchangeable forms were rather low in comparison with the total values (low recovery percentages), and even decreased during the composting process, especially in compost 2 (Table 1). These results also suggest that increased OM stabilisation and humification during composting facilitated heavy metal interaction with ligands of greater complexing strength, rendering them less bioavailable.

Table 1: Average total and soluble metal concentrations of composting materials and mature compost.

	Total Zn	Zn $CaCl_2$		Total Cu	Cu $CaCl_2$	
	(mg kg^{-1})	**(mg kg^{-1})**	**(% Total)**	**(mg kg^{-1})**	**(mg kg^{-1})**	**(% Total)**
SF1	3098	82	2.65	249	1.9	0.76
Pile1 Initial	2931	9.9	0.34	203	0.4	0.21
Compost 1	5552	16.1	0.29	351	1.4	0.39
SF2	3397	61.6	1.81	269	1.8	0.66
Pile 2 Initial	2380	40.6	1.71	175	1.2	0.67
Compost 2	5650	16.8	0.30	391	0.6	0.16

The bulking agent used had a significant effect on the degradation of the OM during composting and on the OM humification process; the cotton gin waste gave higher humic acid-like C formation (compost 2) than the cereal straw (compost 1; Table 2). The particle size distribution of the mature compost was also affected: compost 2 showed a greater proportion of the 0.05-0.5 mm particle size fractions than compost 1 (Table 2). However, the evolution of the total and soluble forms of Cu and Zn during composting, as well as their distribution in the different particles size, was not significantly influenced by the different bulking agents used.

Table 2: Characteristics of the different particle size fractions in both composts

Particle size (mm Ø)	Size distribution (%)	OM (%)	TOC (g kg^{-1})	TN (g kg^{-1})	C_{HA} (g kg^{-1})	C_{EXT} (g kg^{-1})	Zn (mg kg^{-1})	Cu (mg kg^{-1})
	Compost 1							
<0.05	0.2	45.8	200	31.0	5.1	9.7	6168	270
0.05-0.2	4.1	45.3	206	21.9	5.3	10.6	3684	319
0.2-0.5	13.2	52.9	255	24.2	5.2	11.2	3184	362
0.5-1	18.3	57.2	271	26.1	5.1	11.6	3949	362
1-2	18.2	57.4	273	25.4	6.8	13.2	2684	380
>2	45.9	55.8	270	25.8	7.0	13.4	3110	397
Full compost	100	54.4	250	26.5	13.4	24.5	5552	350
Compost 2								
<0.05	0.5	44.4	211	32.1	10.2	17.5	6333	269
0.05-0.2	5.0	42.9	208	27.5	11.2	18.1	4674	286
0.2-0.5	14.8	51.0	219	23.7	10.8	17.1	4354	292
0.5-1	20.3	59.9	268	26.5	13.7	21.5	4021	290
1-2	23.6	62.2	276	30.2	13.5	20.3	4256	309
>2	35.6	60.2	295	27.6	16.9	24.6	4365	313
Full compost	100	59.7	259	28.1	18.1	29.3	5651	391

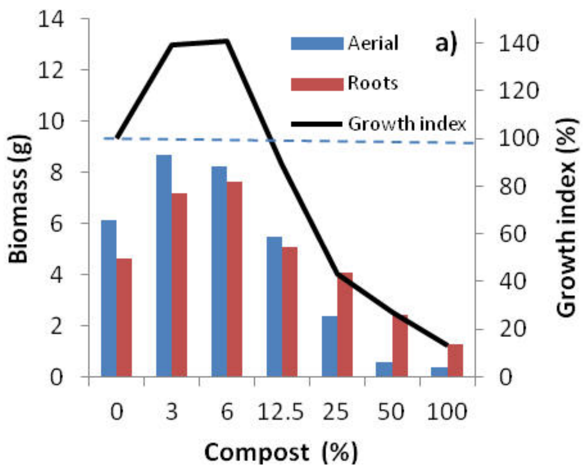
Regarding the separation in different particle size fractions, in both composts the highest Zn concentration was found in the smallest particle size fraction (0.05 mm), with the lowest TOC and humic acid-like C concentration, while the Cu concentration was highest in the largest particle size fraction (>2mm), that had the highest TOC and humic acid-like C concentrations. These results suggest that Zn in mature compost was mainly linked to hardly soluble inorganic compounds. The highly significant relationships between Cu concentration and TOC (r = 0.925 and 0.834 for compost 1 and 2; significant at $p<0.05$ and 0.01, respectively) and OM (r = 0.931 and 0.815 for compost 1 and 2; significant at $p<0.05$ and 0.01, respectively) reveal the ability of this metal to form stable complexes with organic compounds, favouring its retention in the fraction richest in OM.

The elimination of the smallest particle size fraction would remove only a small, insignificant amount of the total Zn content of the composts (12 and 31.6 mg kg^{-1}), since this fraction represented only a minor part of the compost.

3.2. Phytotoxicity test and germination index

The GI increased in both piles throughout the composting process, from very low values in the initial mixtures (data not shown) to 87.8 % in compost 1 and 80.2 % in compost 2, which indicates absence of phytotoxicity. Therefore, in spite of the relatively high total concentrations of Zn and Cu present in the composts, the low solubility of the metals did not cause any relevant toxic effects in *L. sativum* seed germination.

The results of the growth test performed with *Zea mays* (figure 1) gave values of EC_{50} of 42% for compost 1 and 66% for compost 2, indicating that it is necessary to apply at least such percentages of compost to the clean soil to find a reduction of 50 % in the growth of the plants. In addition, the values of LC_{50} (related with seedling emergence) indicate that when compost 1 was applied at 62 % it provoked a lethal effect in 50 % of seedlings, while for compost 2 no harmful effects were found. Both composts stimulated the growth of maize seedlings when applied at up to 3-6 % to the soil, with respect to the control soil without compost; compost 1 also stimulated root growth. But when high proportions of the composts were applied, compost 1 was not able to maintain correct root growth, giving a higher decrease in the growth index than compost 2.



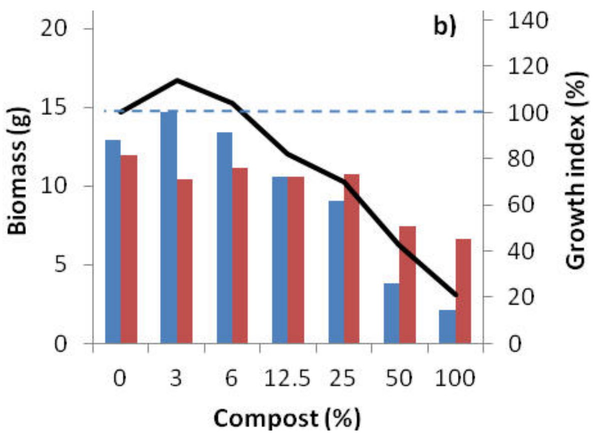


Figure 1. Growth of aerial parts and roots of *Zea mays* (g per plant) and growth index at different proportion of compost in the toxicity test. a) Compost 1; b) Compost 2.

All these results indicate that the presence of Zn and Cu at high concentrations in these composts did not negatively affect plant germination and growth when the composts were used in soils as fertilisers at agricultural doses. But, the application of high proportion of both composts can cause negative effects on plants, and the long-term accumulation of Cu and mainly Zn in the soil.

4. Conclusions and outlook

The composting of the solid phase of pig slurry produced a material with a stable OM content and with humic characteristics. But, the high concentrations of Cu and especially Zn in the solid phase of the pig slurry led to metal rich composts. However, despite the elevated total concentrations, the low solubility of both metals in the mature composts avoided any significant toxic effect on the plants when used as fertiliser, promoting plant growth.

The elimination of the smaller particle size fraction would not reduce significantly the total Zn concentration of the composts. The use of pig slurry from piglets should be avoided in composting in order to prevent high Zn concentrations in the compost produced.

Acknowledgements

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References

[1] Popovic, O., Hjorth, M. Jensen, L. (2012): Phosphorus, copper and zinc in solid and liquid fractions from full-scale and laboratory-separated pig slurry, Environ. Technol. 33, 2119-2131.
[2] Shepard, S.C., Sapinelli, B. (2012): Trace elements in feed manure and manured soils. J. Environ. Qual. 41, 1846-1856.
[3] Riaño, B. García-González M.C. (2014): On-farm treatment of swine manure base on solid-liquid separation and biological nitrification-denitrification of liquid fraction, Environ. Mange.132. 87-93.
[4] Hsu, J.H., Lo, S.L. (2000): Effect of composting on characterization and leaching of copper, manganese, and zinc from swine manure, Environ. Pollut. 114, 119-127.

TA-P_17 Risk arising from disposal of animal wastes to soil

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1. Objectives

This study was conducted to investigate microbiological and parasitic risks related to disposal of animal manure to soil. We focused on survival of Salmonella typhimurium and Ascaris suum eggs in raw pig slurry stored at 4°, 20° and 42°C for 115 days with monitoring of their devitalisation at days 0, 7, 12, 22, 32, 40, 55, 90 and 115 of storage. Besides influence of temperature the potential effect of physico-chemical parameters during the storage was also investigated.

2. Methodology

The experiment was carried out on raw pig slurry obtained from a pig farm. The slurry was stored for 115 days in closed plastic containers of volume 5 at one of three temperatures: 4, 20 or 42°C.
Before storage, lyophilised strain S. typhimurium SK 14/39 (SZÚ Prague, CR) was inoculated into the investigated slurry (initial count of S. typhimurium 3.6 x 109 CFU.ml^{-1}).
Also special polyurethane containers, each containing 1500 A. suum eggs, obtained by dissection of distal ends of the uterus of A. suum females were introduced into the slurry.
Plate counts of S. typhimurium were determined at days 0, 7, 12, 22, 32, 40, 55, 90 and 115 of storage. At the same intervals devitalisation of A.suum eggs was observed in comparison with A.suum eggs stored in distilled water at the same temperatures.
The following changes in physical and chemical properties of the slurry were monitored: pH (measured by HACH electrode), dry mater (DM) determined by drying at 105°C to a constant weight, chemical oxygen demand (COD_{Cr}) determined by HACH method, and ammonium ions (NH_4^+) analysed by titration.
The physical and chemical properties (pH, DM, COD and NH_4^+) of pig slurry, as well as the number of damaged eggs were expressed as mean values ± standard deviation (x ± SD).
Significance of differences between experimental and control groups of parasites were determined using Student t-test, ANOVA and Dunnet Multiple Comparison test at the levels of significance 0.05; 0.01 and 0.001 (Statistica 6.0).

3. Results and discussion

With regard to animal wastes we are concerned particularly with representatives of the family Enterobacteriacea, the majority of which have zoonotic character, such as Salmonella sp., Escherichia coli, Mycobacterium sp., Enterococcus sp., etc., which are a threat to farm animals. Excrements of farm animals are also a source of endoparasites (cysts, eggs, larvae of genera Ascaris sp., Oesophagostomum sp., Trichuris sp., Giardia sp., and others) that may cause massive parasitic infections in both specific hosts and non-specific ones, such as man. An important factor in spreading of endoparasitoses is high tenacity of some propagative stages of parasites (Papajová and Juriš, 2012). It is generally known that some eggs, infectious larvae (L3), oocysts or sporocysts can survive for considerable time, frequently for several years, even under unfavourable environmental conditions. The most dangerous are highly resistant eggs of some parasitic nematodes, e.g. Ascaris spp., Trichuris spp. and coccidial oocysts. After application of manures to land there is some movement of the pathogens through the soil matrix, both horizontally and vertically. In soil this movement is affected by moisture retaining properties of soil.

3.1 Devitalisation of Salmonella typhimurium

According to Reissbrodt et al. (2000), salmonella species are important food-borne pathogens that represent a significant and increasing public health problem in industrialized countries. The most common pathogens S. enteritidis and S. thyphimurium are responsible for nearly half of all illnesses. The persistence of these bacteria in the environment depends on the long-term survival of heavily stressed cells. Therefore the risk to public health arising from application of insufficiently treated animal manure to soil may be higher than detected by common methods. At 4°C the initial concentration of the tested S. typhimurium strain decreased three orders of magnitude by day 90 and on day 115 of storage the test strain was not detected. At 20°C a marked decrease by 7 orders of magnitude was observed on day 32 and from this day the test strain was investigated only qualitatively. The most marked decrease in plate counts of test bacteria was recorded in pig slurry stored at 42°C (Fig. 1). Our results showed that that viability of bacteria in stored pig slurry was significantly affected by the temperature during the storage. Arrus et al. (2006) observed influence of temperature on S. typhimurium and concluded that while Salmonella did not grow in hog manure, storage reservoir temperatures would facilitate Salmonella survival over winter enabling contamination of fields at spring application. Since untreated liquid hog manure may contain Salmonella, manure should be held for 60 days without commingling with fresh manure in stores before application to fields. Key to controlling salmonella is to follow the general rules that have been successfully applied to other infectious diseases (Burton and Turner, 2003).

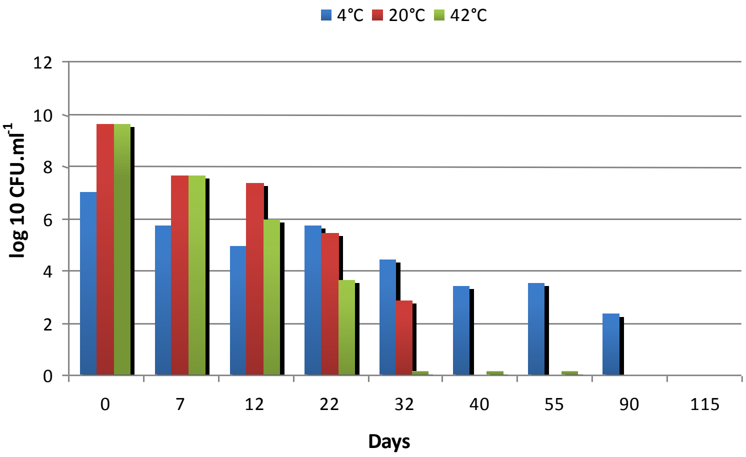

Figure 1 Survival of Salmonella typhimurium in the raw pig slurry stored at three different temperatures

3.2 Devitalisation of A.suum eggs

Parasite survival in animal manures may also be related to temperature, but the trends are not as pronounced as those reported for bacterial pathogens. This is likely due to their ability to form cysts and oocysts for protection from environmental pressures. A. suum eggs are highly resistant to inactivation in faeces, potentially remaining infectious for years. Ascaris infects over a quarter of the world´s human population (1.47 billion people worldwide) and clinically affects ~335 million people (Crompton, 1999).

The above-mentioned helminthozoonoses are classified among epidemiologically ”low-risk” parasitozoonoses, because the propagative stages develop in the outdoor environment into the infectious stage and potentially secondarily contaminate the food chain. Therefore, a direct contact with infected animal, but also contaminated environment, or contaminated food chain (water, vegetables) are considered as a potential risk factor (Papajová and Juriš, 2012).

Our investigations showed that at 4°C the number of devitalized eggs increased with the length of storage up to day 115, with the exception of day 20, when it was almost doubled compared to the control. The difference between storage in slurry and distilled water was significant ($P<0.05$) after 55 days of storage. More marked decrease in survival was observed at 20°C but the viability of non-embryonated eggs of A .suum varied considerably during storage at this temperature. Observations at 42°C showed still more rapid devitalization of A. suum eggs exposed to pig slurry. Significant differences in the number of devitalized eggs were observed on days 12, 40 and 90 ($P< 0.01$) and 7, 20, 55 and 115 ($P< 0.001$) of storage (Table 1).

Our study showed that the number of devitalised Ascaris eggs generally increased with the length of storage and the temperature. However, considerable number of Ascaris eggs remained viable even after 115 days of storage at 4° and 20°C. Only at 42°C more than 90 % of eggs were devitalised after 12 days of storage. However, such temperature can only rarely be reached in animal slurries, thus the risk of persistence of this zoonotic parasite is very high.

Beside temperature also changes in physico-chemical properties of slurry could affect the vitality of model non-embryonated A. suum eggs at long-term storage (Venglovsky et al., 2009).

Table 1: Survival of non-embryonated model A. suum eggs in raw pig slurry stored at three different temperatures

Exposure time (days)	Devitalized eggs of A. suum ($\overline{X}$ % ± SD)		
	4°C	20°C	42°C
0	15.55 ± 2.52	10.13 ± 3.17	14.90 ± 4.06
7	16.86 ± 2.39	16.72 ± 1.38	74.30 ± 0.82***
12	20.33 ± 9.31	17.17 ± 5.74	92.39 ± 4.95**
20	17.04 ± 11.25	16.86 ± 3.64	99.23 ± 1.09***
32	21.54 ± 16.67	26.90 ± 4.38	92.85 ± 10.10**
40	22.87 ± 4.06	21.85 ± 10.69	96.15 ± 5.44**
55	24.90 ± 1.28*	23.81 ± 13.46	98.24 ± 2.48***
90	25.87 ± 5.84	36.28 ± 10.91	95.83 ± 5.89**
115	26.97 ± 5.14	37.65 ± 8.34	99.65 ± 1.34***
Control	14.14 ± 0.82	15.14 ± 0.92	14.14 ± 0.22

**Significance at the level P<0.05, **Significance at the level P<0.01, ***Significance at the level P<0.001*

3.3 Changes in physico-chemical parameters

Besides temperature and time of storage the survival of pathogens in the slurry may well depend on factors other than temperature and duration of heat treatment, e.g. moisture content, free ammonia concentration, pH, the presence of other micro-organisms and other physico-chemical properties (Turner, 2002; Venglovský et al., 2006) This was the reason why the stored pig slurry was subjected also to physico-chemical examination.

With some fluctuations the pH level increased throughout the storage at all three temperatures from the initial 6.95 ± 0.11 to 8.35 ± 0.58 at 4°C, to 8.55 ± 0.17 at 20°C and to 9.35 ± 0.05 at 42°C. On the other hand, we observed a temperature-dependent decrease in DM content toward the end of storage from the initial 28.00 ± 6.26 g.kg^{-1} DM to 15.09 ± 1.23 at 4°C, 13.09 ± 4.12 at 20°C and 12.89 ± 1.05 g.kg^{-1} DM at 42°C. The NH_4^+ level fluctuated considerably and the final level of this parameter differed between the treatments. By day 115 it increased from the initial 50.00 ± 21.11 g.kg^{-1} DM to 84.13 ± 21.23 g.kg^{-1} DM at 4°C and decreased to 27.11 ± 1.06 g.kg^{-1} DM at 20°C and always down to 0.56 ± 0.01 g.kg^{-1} DM at 42°C. The COD generally decreased from 804.00 ± 2.73 g.kg^{-1} DM to 734.09 ± 2.89 at 4°C, 489.09 ± 42.12 at 20°C and to 222.89 ± 10.05 g.kg^{-1} DM at 42°C.

The observed pH increase in stored pig slurry at all three temperatures was not in correlation with the level of ammonium which varied considerably. This could be due to its release as ammonia and the related decrease in total nitrogen by the end of the experiment except for temperature 4°C. The covers were not airtight and were removed at each sampling. Dry matter content decreased according to expectations, so did other parameters, which may be related to production and release of some volatile compounds during the storage. The processes that take place in slurry are, however, very complex and the extent of our examinations did not allow us to draw any definite conclusions in this respect.

4. Conclusion and outlook

There are significant microbiological risks related to animal wastes spread onto land subsequently used for crop production or livestock grazing. Animal slurries are of particular concern, as the temperature in these substrates during their storage and some ways of common processing does not ensure complete devitalisation of potential bacterial and viral pathogens and eggs of parasites, as indicated by our observation of devitalisation of S. typhimurium and A. suum eggs at 4°, 20° and 42°C during 115 days of storage. In advanced countries relevant legislative regulations require acceptable procedures for the disposal, processing and application of animal manures. However, there are still aspects that may raise some risk for safety of human food chain and require further investigations. The best way is to put stress on preventive actions and measures that may eliminate any known or suspected danger resulting from pathogens present in animal manures applied to the soil that is used for animal grazing or growing of crops for human consumption.

Acknowledgements

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References

[1] Arrus, KM, Holley, RA, Ominski, KH, Tenuta, M, Blank, G. (2006): Influence of temperature on Salmonella survival in hog manure slurry and seasonal temperature profiles in farm manure storage reservoirs. Livestock Science, 102, 226-236

[2] Burton, C. H., Turner, C. (2003): Manure management - treatment strategies for sustainable agriculture. 2nd Edition, Silsoe Research Institute, Bedford, UK, 490

[3] Crompton, D. W. (1999). How much human helminthiasis is there in the world? Journal of Parasitology, 85, 397-403

[4] Papajova, I., Juriš, P. (2012): The sanitation of animal waste using anaerobic stabilisation. In: Kumar, S., Bharti, A. (Eds.) Management of organic waste. Rijeka: InTech, 49-68

[5] Reissbrodt, R., Heier, H., Tschäpe, H., Kingsley, R. A., Williams, P. H. (2000): Resuscitation by ferrioxamine E of stressed Salmonella enterica serovar typhimurium from soil and water microcosms. Appl. Environ. Microbiol., 66, 9, 4128-4130

[6] Turner, C. (2002): The thermal inactivation of E. coli in straw and pig manure. Bioresource Technology, 84,1, 57-61

[7] Venglovsky, J., Martinez, J., Placha, I. (2006): Hygienic and ecological risks connected with utilization of animal manures and biosolids in agriculture. Livestock Science, 102, 3, 197–203

[8] Venglovsky, J., Sasakova, N., Placha, I. (2009): Pathogens and antibiotic residues in animal manures and hygienic and ecological risks related to subsequent land application. Bioresource Technology, 100, 22, 5386-5391

TA-P_18 Phosphorus recovery prior to land application of biosolids using the "Quick wash" process developed by USDA

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1. Objectives

A new process, called "Quick Wash", for extraction and recovery of phosphorus from animal manure solids was developed by USDA-ARS scientists [1, 2]. Additional research has shown the technology is equally effective to recover phosphorus from biosolids prior to application to soil. Our objective is to describe the case study of adaptation of this process for the recycling of municipal sludge and biosolids and show an example of its byproducts (low phosphorus biosolids and recovered phosphorus materials). Further observations are provided to show its potential environmental benefits for urban and rural communities.

2. Case study description

2.1 The "Quick Wash" process

The Quick Wash process is an alternative treatment process to improve the quick wash (N) and phosphorus (P) balance in animal wastes. It was developed for rapid wet extraction of P from raw solid manure and recovery of manure P in solid concentrated form [1, 2]. This process consists of selectively extracting P from solid animal waste using mineral or organic acid solutions, and recovery of P from the extract by adding lime and an organic polymer forming a calcium-containing P precipitate. The Quick Wash process has two solid byproducts: 1) a washed solid residue with an N:P ratio optimal for use in crop production; and 2) a concentrated solid P material that can be used as an effective P fertilizer. The final liquid effluent can be applied to nearby cropland or recycled into the treatment system [2]. Figure 1 shows the schematic of the process invented by a team of scientists at the ARS- Coastal Plains, Soil, Water & Plant Research Center, and (U.S. Patent 8,673,046 [1]. An exclusive license has been granted by USDA to a small business -- Renewable Nutrients LLC, Pinehurst, NC (www.renewablenutrients.com) -- for commercial use of the Quick Wash process. The Team worked closely with Renewable Nutrients to develop approaches for commercializing this new technology for municipal biosolids and the poultry markets. To take advantage of the nutrient credits in the Chesapeake Bay Watershed, a mobile pilot plant was constructed and first tested at the Borough of Ephrata Wastewater Treatment Plant (Ephrata, PA) (Figure 2).

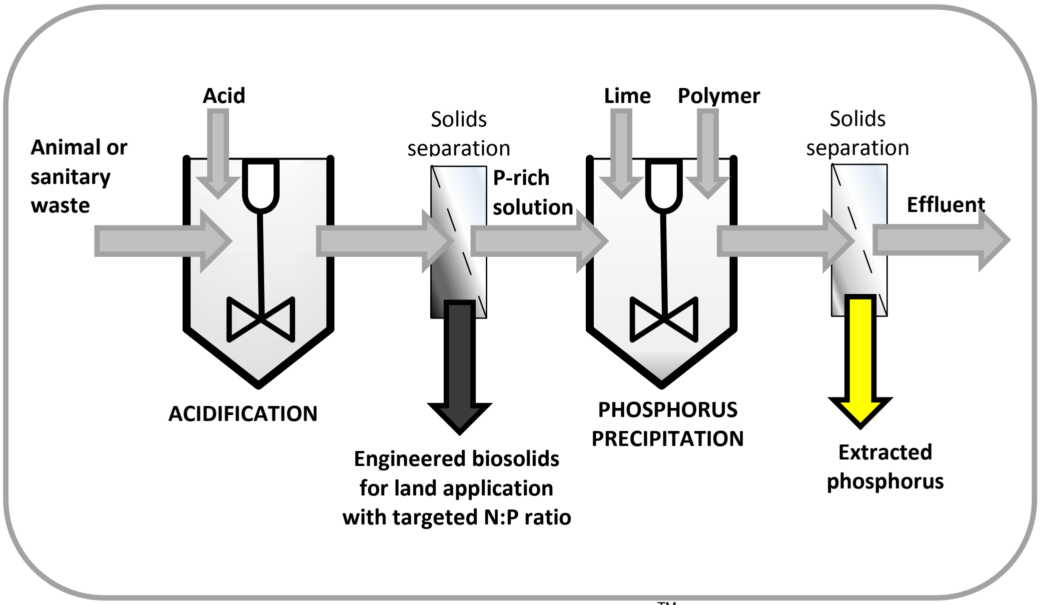

Figure 1: Schematic of the Quick Wash^{TM} process.

Figure 2: Exterior and interior of mobile pilot plant deployed at the Borough of Ephrata Wastewater Treatment Plant.

2.2 Phosphorus extraction and recovery

The Wastewater Treatment Plant #1 at Ephrata, PA, produces annually 370 dry tons of biosolids (Class B with an average total P content of 3.9%), most of which are landfilled. The sludge tested for P recovery had a composition of 1/3 activated sludge (secondary digest) and 2/3 waste activated sludge treated with ferrous chloride prior to dewatering.

This first step of the Quick Wash, consists of mixing mineral or organic acid with the municipal sludge to selectively extract P at a pH lower than 5.0. In the laboratory, the optimum citric and sulphuric acid doses were determined to be 7.87 g L^{-1} (or 40 m*M*), and 8.17 g L^{-1} (or 80 m*M*), respectively, by titrating the sludge with the acids. After mixing the acid with the sludge, a liquid extract is formed. With the citric acid treatment the pH of the extract was 4.1 and 83% of the total P was extracted while the sulphuric extract had a pH of 1.73 while extracting 87% of the total P. After sludge dewatering (centrifuged at 3000 rpm for 10 minutes), the acid treated biosolids had an N:P elemental ratio of more than 4.0 (Table 1).

Table 1: Mass of biosolids per volume of sludge, total N and P content, and N/P ratio (dry weight) after acid extraction.

Acid treatment	Solids	TN	TP	N/P
	kg m^{-3}	%	%	
Untreated	19.3	4.68	4.08	1.2:1
Citric 40 m*M*	15.4	5.43	1.21	4.5:1
Sulphuric 80 m*M*	14.7	5.54	1.13	4.9:1

Once biosolids are separated, P is recovered from the extracting solution by adding hydrated lime [$Ca(OH)_2$] to reach a pH of 10. Thereafter, a small amount of anionic polymer (polyacrylamide) was added to enhance P precipitation (Table 2). With respect to total P in untreated sludge, a mass balance indicated that P recovered in the precipitate after lime application was 75% for citric acid and 65% for sulphuric acid. The recovered P materials are a good source of P with contents of more than 10% P_2O_5 (Table 2). These recovered P materials can contain more than 90% of their P in plant available form and are a recycled P source for use as crop fertilizer [3].

Table 2: Phosphorus recovered after precipitation with lime and anionic polymer addition.

Acid	Lime use	Polymer use	Precipitate [a]	TN	TP	P_2O_5 [b]
	kg m^{-3}	g m^{-3}	kg m^{-3}	%	%	%
Citric 40 m*M*	5.0	15	6.0	0.66	8.10	18.5
Sulphuric 80 m*M*	6.7	15	15.9	0.84	4.41	10.1

[a] Mass of precipitate per volume of extracting solution; [b] $\%P_2O_5 = \%P \times 2.29$.

3. Observations

Excess soil P beyond the assimilative capacity of soils is currently a major factor limiting application of biosolids to land near municipal wastewater treatment plants. For this reason, municipalities incur hefty fees for transportation and landfilling biosolids that otherwise could be used as soil amendments to maintain soil quality and enhance crop production. Nevertheless, recovering P from biosolids using the Quick Wash process can provide environmental benefits for both rural and urban communities. The Quick Wash process selectively recovers P from sludge while leaving most of the N in the washed solid residue (Fig. 3). Consequently, the washed solid residue has a more balanced nutrient composition for crop production and is safe for land application. Because the Quick Wash process is conducted at ambient temperature, it minimizes loss of organic carbon (C) and N from washed solid residues. Thus, the land application of washed solid residues contribute C and N to maintain soil quality while reducing the environmental risks of excess soil P. The P recovery aspect of this process is an attractive approach because of increasing global demand and cost of mined phosphates and declining world P reserves. The final P recovered by-product is a material rich in calcium phosphate that could be transferred in concentrated form to P-deficient croplands and used as fertilizer.

Figure 3: Washed biosolids with low P content and recovered phosphate produced with the Quick Wash^{TM} process.

Additional tests are needed to further adapt the Quick Wash to municipal sludge treatment. In particular, the quality and quantity of the recovered phosphates can be greatly improved by reducing iron and aluminum impurities. Iron and aluminum salts are usually used in municipal wastewater treatment plants to scavenge P from sludge side streams. Therefore, the Quick Wash process can help to reduce the cost of addition of iron and aluminum compounds for municipal wastewater treatment while recovering a high quality calcium phosphate.

Conclusion and outlook

Nutrient pollution, caused by too much P in the environment, is one of America's most widespread, costly and challenging environmental problems, impacting many sectors of the U.S. economy that depend on clean water. These environmental problems can be mitigated with the Quick Wash technology because P is selectively extracted from animal wastes and municipal biosolids prior to land application. The inclusion of this process in a waste management system offers farmers and municipalities a new and welcomed opportunity to minimize P losses into the environment and sustain soil quality while recovering and recycling P as a valuable product. A strategy to reduce costs of this new technology is to install it on a sufficient number of municipalities to facilitate engineering improvement and development of the market for its byproducts. Eventually, economic incentives such as share cost programs, tipping fees and environmental credits may be needed to stimulate the adoption of this P recovery technology.

Acknowledgements

Mention of trade names or commercial products in this article is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the U.S. Department of Agriculture. This study is part of USDA-ARS National Program 214: Manure and By-Product Utilization; ARS Project 6082-13630-005-00D "Innovative Animal Manure Treatment Technologies for Enhanced Environmental Quality." We would like to thank Renewable Nutrient, LLC for providing sludge samples and pictures of the pilot treatment plant.

References

[1] Szogi, A.A.; Vanotti, M.B.; Hunt, P.G. (2014): Process for removing and recovering phosphorus from animal waste. U.S. Patent No. 8,673,046.
[2] Szogi, A.A.; Vanotti, M.B.; Hunt, P.G. (2015): Phosphorus recovery from pig manure prior to land application. Journal of Environmental Management 157, 1-7.
[3] Szogi, A.A.; Bauer, P.J.; Vanotti, M.B. (2010): Fertilizer effectiveness of phosphorus recovered from broiler litter. Agronomy Journal 102, 723-727.

TA-P_19 Improvement of soil nitrogen's mineralization modeling with a better consideration of farm yard manure and crop residues inputs history

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1. Objectives

In France, soil N mineralization is currently [1] calculated by implementing a mineralization rate to an active N organic stock of arable layer. In most French tools for managing nitrogen fertilization based on the nitrogen balance sheet method, the size of active stock is considered equal to 35 % of the total arable layer stock. This part is then adjusted by a coefficient (Fsys) according to crop residues and exogenous organic products restitutions practices. But this method of calculation is not accurate enough in all situations. A study was carried out in 2014 to improve it.

2. Methodology

The model currently used in France (AMG [2]) to simulate organic carbon stock dynamic in the soil arable layer contains two compartments: a stable one (compared to human life duration) and an active one which supports all the carbon inputs and outputs.

The model put forward by COMIFER* to estimate N mineralization in balance sheet method for crops is based on the same principle. Its objective is to calculate annual N supplying from soil using organic N stock in the arable layer (measured by soil analysis).
For both models, the assessment of the active compartment of total stock (35% for organic N which is implemented by a mineralization rate) is discussed.

When yearly nitrogen and carbon inputs from crop residues and exogenous organic products are equal to yearly outputs with arable layer stocked organic nitrogen mineralization, the humic balance is equilibrated and active stock reaches a level which is proportional to inputs. So a method of estimation of soil N mineralization was implemented, taking into account a calculation of active N organic stock of arable layer using organic restitutions history which was parameterized on long term field experiments. The method was evaluated on 7 trials comparing mineral and organic fertilization.

3. Results and discussion

3.1 Soil N supplying measurements

The study was based on 7 long term trials (youngest one was 10 years old) on 5 locations (cf. map). Experiments studied the effects of repetitive manure and/or slurry supplies, and were carried out by *chambre régionale d'agriculture de Bretagne* (western France – 2 trials) and ARVALIS-Institut du végétal (5 trials).

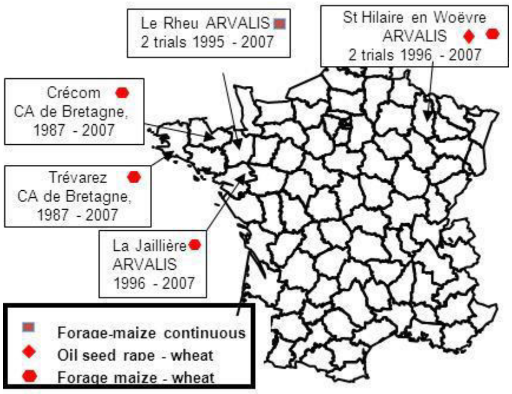

Figure 1 : trial locations

Each experiment protocol had one treatment with inorganic fertilizer only and others with farmyard manures or slurries only. To avoid their short term fertilizer effects, farmyard manures and slurries applications were stopped at least 18 month before the beginning of N mineralization measurements.

In each trial during 2006-2007 period, soil N supplying from organic mineralization was calculated with LIXIM [3]. This model was parameterized using monthly N mineral measurements in bare soil (at the depth of 90 cm and during 6 to 12 months).

Carbon inputs from crop shoot and root residues were estimated with the ratio (like harvest index) used in AMG model, implemented to forage maize, wheat and oil seed rape yields.
Farmyard manure and slurry quantities and chemical content have been measured for each application. Mean annual carbon input (manures and residues) has been calculated on a duration corresponding to 10 years before organic N stock measurement in the arable layer.

Active organic N stock has been calculated (Eq.1) with 1) mineralized N quantity (M) calculated with LIXIM model and 2) with its mineralization rate (m) from the COMIFER method which take into account two parameters: clay and $CaCO_3$ soil content:

$$Sm = M/(m * JNp) \qquad \text{Eq. 1}$$

S_m = Active organic N stock ($t.ha^{-1}$)
M = Mineralized N quantity
m = Mineralization rate of the active organic N stock ($kg.ha^{-1}.JN^{1}.t^{-1}$ of total soil organic N)
JNp = Number of "normalized time" days corresponding to the mineralization calculation period

3.2 Active N stock estimation taking into account crop residues and farmyard manure and slurry applications

Taking into account the available data, we decided to make the calculation during 10 years of carbon inputs. This duration corresponds to the youngest trial available and is consistent with the capacity of farmers to know their chronicle of yield and farmyard manures applications. Because this duration is not long enough to estimate the whole N active stock, older chronicle (> 10 years) is taken into account with total N stock of the arable layer measured in 2006-2007. In this older history, we could not distinguish carbon input sources from residues or manures. In more recent history (< 10 years), this differentiation could be made.

3.3 Using of older (>10 years) and younger (≤ 10 years) chronicles of carbon inputs from residues

Treatments with only inorganic fertilization during the last 10 years were used to estimate the parameterization factors of carbon inputs from residues for both trial durations. The obtained equation links the active N stock (S_m) of these methods with the mean annual carbon inputs from residues during 10 years and the total organic N stock of the arable layer (depth = 25 cm) measured in 2006-2007. This relation explains 72% of the active organic N stock observed variability.

$$\boldsymbol{Sm(r) = aX + bYr + c} \qquad \text{Eq. 2}$$

$Sm_{(r)}$ =Active organic N stock taking into account recent and long term chronicles of C inputs from crop residues ($t.ha^{-1}$)
X = Total organic N stock in 0-25 cm soil layer ($t.ha^{-1}$)
Y_r = Mean annual carbon inputs from residues during 10 years ($t.ha^{-1}.year^{-1}$)
a, b, c = Fitted parameters

3.4 Using of short term chronicle of C inputs from manures/slurries

In each trial, we calculated the difference of active organic N stock (S_m) between methods with only inorganic fertilizer applications and with manures/slurries applications.

$$Sm(m,s) = a'Y(m,s) + b' \quad \text{Eq. 3}$$

$S_{m(m,s)}$= Active N stock increase due to manures/slurries application (t.ha^{-1})
$Y_{(m,s)}$ = Mean annual C inputs from manures/slurries applications during the last 10 years (t.ha^{-1}.year^{-1})
a', b' = Fitted parameters

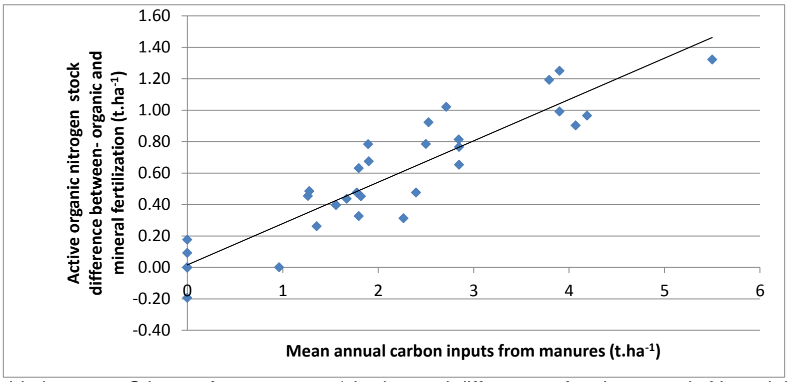

Figure 2: Relationship between C inputs from manures/slurries and difference of active organic N stock between methods with inorganic fertilizer applications and with manures/slurries applications

i. Active organic N stock calculation

Total active N stock is calculated adding $Sm_{(r)}$ and $Sm_{(m,s)}$:

$$Sm = Sm(r) + Sm(m,s) \quad \text{Eq. 4}$$

3.5 Soil N supplying

The mineralization rate of active N stock is implemented to Sm in order to calculate N supplying from its mineralization.

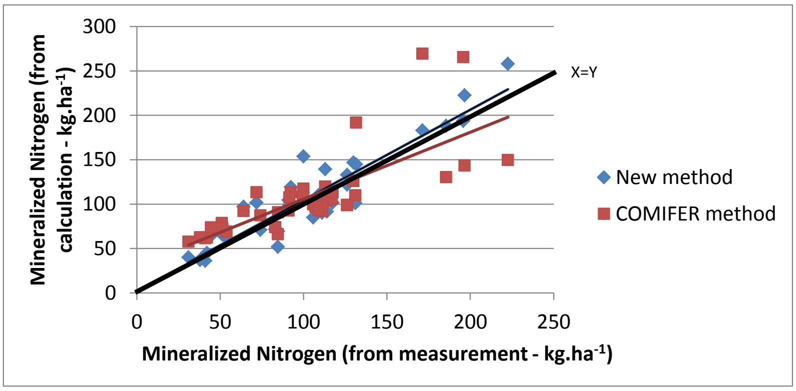

Figure 3: N supplying from mineralization calculated with both methods (COMIFER one and new one) compared to the measured N mineralization in trials.

We tested both methods (COMIFER one and new one) on the dataset of the 7 available trials.
COMIFER method uses a 0.35 factor to obtain active stock from total stock, and adjust this active stock (+/20 % range) according to C inputs history from crop residues and farmyard manures/slurries applications. The COMIFER method overestimates the active N stock (figure 3) in soils with high stable organic matter content (fields succeeding old acid moor or old wetland). In these cases, the active N stock is closer to the active N stock of soils with smaller organic matter content. In other cases where high organic matter content is really due to higher active N stock, the COMIFER method underestimates N supplying. In cases where active N stock represents less than 35% of total organic N stock, N supplying is slightly overestimated by this method (figure 3).
The new method calculates an active N stock due to old C inputs history with less influence of the total N stock (20 %), and adds the effect of recent C inputs chronicle from crop residues and manures/slurries.
Mineralized N supplying estimation is improved (figure 3). It is confirmed by the classical statistical criteria (table 1).

Table 1: Statistical criteria for the evaluation of the methods (mineralized N supplying measured vs calculated with both methods)

	New method	COMIFER method
Fitted parameters		
a	0.92	0.75
b	8.95	30.67
Statistical criteria		
bias	-4.7	-5.1
RMSE	19.7	32.9
r^2	0.90	0.56

4. Conclusion

The new method seems to significantly improve the estimation of soil N supplying along a large range of mineralized N quantities.
Nevertheless, we did not have enough data to precisely determinate the factor values for each parameters influencing active organic N stock (old and recent chronicles of C inputs from crop residues and farmyard manures/slurries applications).
More data from other long term trials or field surveys are needed to improve and validate this new calculation method.

Acknowledgements

We thank “Chambre Régionale d’Agriculture de Bretagne” for providing data of their 2 long term trials, and CASDAR financial support which permits to create a large database of results from more than 30 trials studying long term N effect manures and slurries applications in France.

References

[1] Anonymous (2013) Calcul de la fertilisation azotée, guide méthodologique pour l’établissement des prescriptions, COMIFER www.comifer.asso.fr
[2] K. Saffih-Hdadi, B Mary. (2008): Modeling consequences of straw residues export on soil organic carbon, Soil Biology & Biochemistry 40 594-607
[3] Mary B., Beaudoin N., Justes E., Machet J.M., 1999. Calculation of nitrogen mineralization and leaching in fallow soil using a simple dynamic model. European Journal of Soil Science, 50, 549-566.

TA-P_20 Impact of pollutants from animal farms on quality of potable water

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1. Objectives

The aim of the study was to examine the quality of drinking water from sources used on dairy farms situated in an area potentially contaminated by animal farms. We focused on chemical indicators of contamination, presence of selected and on bacteriological examination that included counts of total coliform bacteria, E. coli, and bacteria cultivated at 22°C and 37°C.These parameters indicate the safety of drinking water for humans and animals and its suitability for operations on dairy farms.

All activities in primary milk production essentially affect the safety of milk and milk products (cleaning and disinfection, rinsing). Drinking water becomes a part of products and thus its quality and safety must be monitored. Water is important also for personal hygiene and hygiene of the working environment.

Individual components of the environment, air, water, soil and organisms affect each other and contaminants shift from one place to another and accumulate in certain locations. Considerable attention is paid to vulnerable regions that were in the past or are still endangered by human activities.

The quality of ground water is frequently reduced by intensive agriculture and the use of nitrogen fertilizers and pesticides. In order to ensure a sufficient quantity of safe drinking water we must prevent penetration of nitrates originating from agriculture into water sources and eliminate all interventions and operations that could affect negatively the quality of water supplies (Howd & Fan, 2008).

Intake of contaminated water by cattle can cause diarrhoea, loss of appetite and hepatotoxic disorders with fatal consequences. Insufficient intake of water decreases productivity and has adverse health consequences. High-yield cows drink daily approx. 75 l of water. Water in drinkers should adequate quality and not be contained contaminants related to feed residues or excrements. The drinking system should be frequently cleaned (Code, 2004).

2. Methodology

Investigations were carried out on 5 farms, three were supplied with water from own wells (farms 2, 3 and 4) and two (farms 1 and 5) were connected to mass public supply. Water samples were collected over a period of one year.

The methods used for analysis of water corresponded to the Governmental Order of SR No. 496/2010 Coll. on requirements for drinking water and control of quality of drinking water, compatible with EU Drinking water directive (WHO, 2008).

Physical and chemical examination focused on parameters of contamination of water with faeces of humans or people (pH, ammonia, nitrates, nitrites, and chlorides), residual chlorine and chemical oxygen demand (COD) which reflects the risk of formation of byproducts related to disinfection of water with active chlorine. First qualitative examination was performed, followed by quantitative colorimetric examination in case of positive qualitative result.

Levels of selected heavy metals (Hg, Cd, Pb, As, Cr, Ni, Cu, Al, Fe, Mn, B) were determined by Atomic Absorption Spectroscopy (AAS).

Bacteriological examination included determination of colony forming units (CFU) of coliform bacteria (CB), an indicator of pollution, E. coli (EC) reflecting the risk of contamination with faeces, and bacteria cultivated at 22°C and 37°C (BC22 and BC37) indicating general pollution and potential pollution with the content of digestive tract of humans and warm blooded animals. We used cultivation methods based on inoculation of samples onto relevant nutrient agars and cultivation for prescribed time at optimum temperatures.

3. Results and discussion

The lack of high quality drinking water is one of the most important problems to which humans have to face in this century. Due to anthropogenic activities we commonly find territories where the drinking water sources are contaminated to such degree that the water from them is neither suitable for drinking and watering of animals nor for food processing and other purposes related to our everyday life (Sasáková, 2009).
Polluted water can affect adversely the animals that have to consume it (abortion of calves, ketosis or acetonemia of cattle, chronic diarrhoea, liver damage, spreading of infections) (Bitton, 1999).
Ground water is the type of water located in the rock environment. It is a well available and quantitatively, qualitatively and economically most suitable source of drinking water (Younger, 2007).
It has constant temperature and usually does not require for special treatment, only removal of some minerals or ensuring the health safety, for example by chlorination. However, the number of substances that may contaminate ground water increases which poses a risk of their transfer into the food chain (Mackler & Merkle, 2000).

3.1 Chemical examination of water

Chemical examination of drinking water on dairy farms showed that pH level was within the recommended range in water from all sources. All ammonium ions (NH_4^+) were detected only in samples collected from source 2 and reached the highest concentration in spring which could be related to melting of snow or extensive rainfall. Nitrites (NO_2^-) were present in this source in samples collected in autumn. Nitrates (NO_3^-) were found in all sources at every sampling in spring and only in traces in the remaining seasons. Chlorides (Cl^-) were detected particularly in winter in samples from sources 2 and 3 and in other sources only in trace levels.
The limit values for nitrates and chlorides (50 $mg.l^{-1}$ and 100 $mg.l^{-1}$, resp.) were exceeded in samples from sources 3 and 4 (wells).
The limit for COD_{Mn} (3 $mg.l^{-1}$) was exceeded in source 4 in winter. This parameter reflecting the presence of chemically oxidisable organic matter is of health importance because it is related to potential production of disinfection byproducts and the risk of cancer. Drinking water for mass supply is monitored for the presence of 4 such byproducts and the sum of them, but such examination is not carried out for individual sources.
Water intended for mass supply is disinfected, mostly with active chlorine, to a level that produces no risk to consumers. Our examinations of free chlorine levels showed presence of low residual levels of free chlorine in water on farms 1 and 5 and on farm 2 in summer and autumn.

3.2 Bacteriological examination of water

Coliform bacteria (CB) as an indicator of potential faecal contamination were present only in individual sources (wells) (Table 1.). They should be absent in any 100 ml water sample. In all wells coliform bacteria were present in 100 ml of water at all seasons. No coliforms were present in any 100 ml water sample from farms 1 and 5. E coli were present only in samples of water from farms 3 and 4 and the highest counts were determined in autumn.
Limit for BC22 were exceeded (>300 CFU) only in source 3 in spring and in source 4 in summer
Bacteriological contamination of wells could be related to unsuitable location of wells and their insufficient protection. This is indicated by the potential influence of melting of snow in spring, heavy precipitations and agricultural activities, particularly application of manure. Animal grazing and run-off from farms could also contribute to contamination of poorly protected wells (Council Directive 91/676/EEC).

3.3 Determination of heavy metals

Water from mass supply sources presents no problem in this respect as the water intended for mass supply is appropriately treated and monitored. Monitoring for increased levels of heavy metals in individual wells enables to identify potential sources and risks to animals and humans and try to take some measures for improvement. Our examination of water from wells on farms 2, 3 and 4 for 11 heavy metals (Hg, Cd, Pb, As, Cr, Ni, Cu, Al, Fe, Mn, B) showed that their levels did not exceed the limits set by relevant legislation.

Table 1: Results of microbiological examination of drinking water on investigated farms

Farm	Season	EC	CB	BC22	BC37
		Limit			
		0 CFU/100 ml	0 CFU/100 ml	200 CFU/ 1 ml	200 CFU/ 1 ml
1	winter	0	0	0	0
	spring	0	0	0	0
	summer	0	0	140	0
	autumn	0	0	0	0
2	winter	0	8	0	0
	spring	0	220	170	160
	summer	0	60	148	80
	autumn	0	48	80	50
3	winter	1	26	104	30
	spring	4	240	> 300	270
	summer	6	138	124	140
	autumn	13	35	90	103
4	winter	8	80	120	90
	spring	6	260	180	165
	summer	66	288	> 300	290
	autumn	130	240	0	150
5	winter	0	0	1	0
	spring	0	0	0	0
	summer	0	0	0	0
	autumn	0	0	0	0

4. Conclusion and outlook

Monitoring of drinking water focuses on parameters that indicate some risk to its consumers and allow one to take adequate measures. Examination of water from wells on farms 2, 3 and 4 showed increased levels of some chemical parameters related to faecal contamination that was further confirmed by the presence of total coliforms and even E. coli and suggested the necessity of such measures, particularly increased protection of the immediate area of wells and elimination of activities that may contribute to risks. Disinfection of water can help to protect consumers from microbiological risks but does not provide long-term solution. Complex evaluation of water and checking on geological situation (location and potential contamination of the relevant aquifer) in the entire location can provide further insight into the problems.

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References

[1] Bitton, G. (1999): Waster water microbiology, Willey-Liss, New York.
[2] Code of Good Hygienic Practice for Milk Products, CAC/RCP 57–2004.
[3] Council Directive 91/676/EEC (1991): On protection of water against contamination with nitrates from agricultural sources.
[4] Howd, RA & Fan, AM. (2008): Risk assessment for chemicals in drinking water. 1 st ed., John Wiley and Sons, Inc. Hoboken.
[5] Mackler, BA & Merkle, JC. (2000): Current knowledge on groundwater microbial pathogens and their control, Hydrogeol J. 8, 1, 29-40.
[6] Sasáková, N, Papajová, I, Ondrašovičová, O, Laktičová, K, Gregová, G, Bis-Wencel, H, Venglovský, J, Mareková, J, Hromada, R, Halán, M. (2009): Evaluation of sources of drinking water in environmental polluted region from chemical and microbiological point of view. Folia Veterinaria 53, 3, 132-133.
[7] Younger, PL. (2007): Groundwater in the Environment, 1st edn. Malden, Victoria Blackwell Publishing, Oxford.
[8] WHO Guidelines for Drinking-water Quality (2008): WHO/SDE/WSH/05.08/55.

TA-P_21 Vegetable crop residues as feedstock for composting and silage – Production cost and product quality

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Objectives

The crop residues of vegetables are important for nutrient and organic matter cycling. However, these residues often may lead to nitrogen leaching risks. This study investigates the feasibility of removing crop residues, followed by composting or ensiling as a means to reduce N leaching losses.

Methodology

Crop residues of cabbage were mechanically harvested. Leek residues were obtained from preparing harvested leek plants for market. For composting, residues of white cabbage or leek were mixed with wood chips and bark, grain and maize straw. Residues of leek were also mixed with straw and alternative brown materials, i.e., heath chopper or used tomato or strawberry substrate. For silage, crop residues of white cabbage, celery, cauliflower or leek were mixed with chopped maize straw.

Results

Collection of both cabbage residues and maize straw was not very effective as the collected material had a high soil particle load. This resulted in a suboptimal composting process and a low organic matter content in the composted end products. A high quality compost was obtained when using leek crop residues combined with the alternative brown materials with a high degree of purity. However, compared to the used growth substrates, heath chopper seemed to induce a better structural condition to the compost pile guaranteeing a sufficient oxygen supply. The establishment of a compost pile based on wood chips and crop residues implied a net variable cost of 55 to 77 euro per ton crop residues, which primarily reflected the cost of wood chips used as a structural material.

Silage quality was optimal for the mixtures with leek and celery, and less optimal for the other mixtures. This was related to higher NH_4^+-N concentrations and lower compressibility of the mixtures with cauliflower and white cabbage. Ensilaged leek residues showed potential to be used as fodder. The net variable costs for ensilage of leek residues amounted to 9.9 euro per ton crop residues.

Conclusion

Ensilaging conserves the nutrients and organic matter for reuse on the field after the winter or for other applications. As ensilaged residues remain highly biodegradable, they will possibly cause a temporary N-immobilization when applied as a soil improver. Composting results in stabilization of the organic matter before application on the field and lowers the risk of nutrient losses after application. Composting allows to add residues during the process in function of their availability.

References

[1] Agneessens, L., Vandecasteele, B., Van De Sande, T., Goovaerts, E., Crappé, S., Elsen, A., Willekens, K., De Neve,.S. (2014). 'Onderzoek naar het beheer van oogstresten bij vollegrondsgroenten en mogelijkheden van vanggewassen en teeltrotaties met het oog op de waterkwaliteitsdoelstellingen van het Actieprogramma 2011-2014 (MAP4): Hoofdrapport', studie uitgevoerd in opdracht van de Vlaamse Landmaatschappij (VLM), 149p.

TA-P_22 Bioavailability of phosphorous in thermally treated sewage sludges and pig manure

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1. Objectives

Mineral phosphorus (P) reserves are finite, increasing demand for novel methods to utilize P rich by-products. Sewage sludges are among the least utilized P sources as well as pig manure produced in centralized large units. Hydrothermal carbonization (HTC) and pyrolysis may be used to produce safe fertilizer products from these materials. We evaluated P solubility and bioavailability of sewage sludges (SS) and pig manure (PM) after thermal treatments.

2. Methodology

Anaerobically digested sewage sludges (SS) represented waste water treatment plants that use either high (SSH) or low (SSL) doses of iron for P precipitation. Sludges were obtained from Helsinki, Finland (SSH) and from Helsingborg, Sweden (SSL). Pig manure (PM) was obtained after separation of solid and liquid fraction and thereafter was further treated with chemical flocculant to remove remaining solid fraction from the liquid phase and dried at 70 °C. Sewage sludges were hydrothermally carbonized at 200 °C for 20 hours, and PM was pyrolysed at a temperature of 410 °C for one hour. Target temperature was reached with temperature increase of 2 °C per minute. Elemental composition of the materials was analyzed with ICP-OES after aqua regia digestion.

Solubility of P was tested by using a modified Hedley fractionation scheme [1]. Hedley fractionation consists of sequential extractions with water (twice), 0.5 M $NaHCO_3$, 0.1 M NaOH and 1 M HCl with the extraction ratio of 1:60. Extraction times were 16 h (4 h with the first water extraction). Following extraction, the samples were centrifuged (3000 g, 15 min) and inorganic P (Pi) (supernatants filtered through a 0.2 µm nucleopore membrane (Whatman, Maidstone, UK)) and total P (unfiltered supernatant digested at 120°C with sulfuric acid and peroxodisulfate) concentration in the supernatant was analyzed according to Murphy and Riley [2]. Organic P (Po) was taken as the difference between total and Pi concentration.

Phosphorus bioavailability was tested in a growth experiment by using barley (*Hordeum vulgare* var. Kunnari) as a test plant. Barley was grown under a glass roof in a P deficient sandy soil (6.5 kg). Soil was limed with $Ca(OH)_2$ (20.8 g pot^{-1}) two weeks before seeding to reach a target pH of 6.5. Before fertilization and seeding, 0.5 l of soil was removed for covering the seeds, after which P sources were mixed with the soil according to total P concentration. SSH and SSL were added to soil as air-dried and sieved (6 mm). Addition of P as sewage sludges with or without HTC treatment was 150 mg P kg^{-1} soil and as PM with or without pyrolysis 40 mg P kg^{-1} soil. Superphosphate (SP) was used as a reference P source at rates of 10, 50 and 150 mg kg^{-1} soil. The control treatment did not receive any P. Bioavailability of P in SS and PM was compared to that of SP. All other nutrients (N, K, Mg, S, Fe, Mn, Zn, Cu, B and Mo) were added as inorganic salts in order to prevent their deficiency. Treatments were replicated four times. Total of 25 seeds per pot was added and thinned to 20 shoots after emerging. Additional dosage of N was given at the stem elonganation stage. Barley was harvested after ripening 2 cm above the soil surface, dried at 65 °C and grain yield was weighed.

3. Results and discussion

3.1 Phosphorus and iron content

Concentrations of P in air-dried SSH and SSL were 33.3 g kg^{-1} and 27.4 g kg^{-1}, respectively. Hydrothermal carbonization increased P concentration to 43.4 g kg^{-1} and 37.8 g kg^{-1}, respectively, due to the concentrating effect. Pyrolysis of PM concentrated also P from initial 23.5 g kg^{-1} in raw material to 50.7 g kg^{-1} in the char fraction.

In Helsinki P is removed from waste water by using Fe for P precipitation. Because of this, Fe concentration was 120.9 g kg^{-1} in sludges, whereas it was only 29.1 g kg^{-1} in sludge originating from Helsingborg, where P is removed biologically. Molar ratios of Fe and P in sludges were 2.0 and 0.6 respectively, while this ratio for PM was 0.05. Thermal treatment did not change these ratios.

3.2 Phosphorus solubility

Inorganic P (Pi): Phosphorus was mainly in inorganic form in SSH, SSL and PM with 95.8%, 92.3% and 95.5%, respectively. HTC-treatment increased Pi share further in SS up to 98-99%, whereas pyrolysing of PM decreased it down to 79%.

Share of water-soluble Pi decreased from 15.3% to 0.1% as the molar ratio of Fe/P increased from 0.6 (SSL) to 2.0 (SSH). Most of the Pi was extracted by 0.1 M NaOH, constituting 65% of the total P in SSH and 47% in SSL (Fig. 1). This fraction is considered to represent Fe bound P. Labile Pi content (water + 0.5 M $NaHCO_3$ –extractable) in SSH and SSL was 2 and 24%, respectively (Fig. 1). Acid soluble Pi content was 29 and 22% in SSH and SSL, respectively.

Water-soluble Pi share in PM was 36%, less than normally found in our studies (data not shown). Also the high share of acid soluble Pi (46%) was more than commonly found in PM, probably due to the use of flocculants during manure separation.

HTC-treatment of SS depressed the share of water, 0.5 M $NaHCO_3$ and 0.1 M NaOH –extractable Pi, but increased the acid soluble P fraction (Fig. 1).

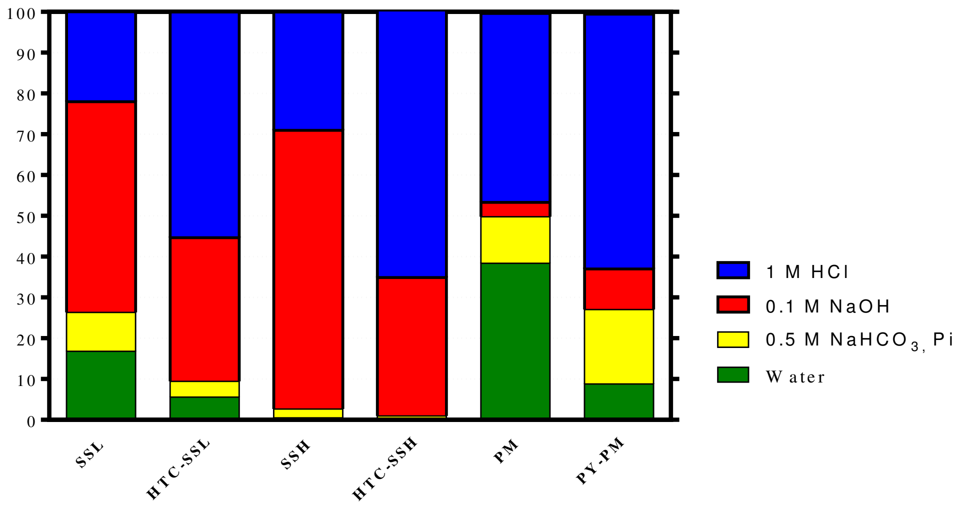

Figure 1: Solubility of P according to Hedley fractionation scheme in sewage sludges with low (SSL) and high (SSH) Fe to P molar ratios and in pig manure (PM) before and after HTC or pyrolysing (PY).

Different types of thermal treatments seemed to have a different effect on P solubility. Concentration of 0.5 $NaHCO_3$-soluble Pi decreased in SS after HTC, but increased in PM after pyrolysing. This does not seem be dependent on raw material, as pyrolysing of SS increased 0.5 M $NaHCO_3$ –extractable Pi as well (unpublished results) Also concentration of 0.1 M NaOH -soluble Pi either remained at the same level in SSL or decreased in SSH, but increased in PM (Fig. 1). However, all treatments increased the HCl-soluble fraction, which indicates low, or at least slow, plant availability of phosphorus.

Organic P (Po): Pyrolysing of PM increased 0.5 M NaHCO3 –extractable Po concentration from 0.2 to 5.1 g kg-1 and 0.1 M NaOH -extractable Po from 0.4 to 3.4 g kg-1. Although the HTC-treatment depressed the share of Po in SS, the total share of Po is minor compared to that of Pi. However, these results indicate that treating organic material either with HTC or pyrolysis, will have a different effect on the composition of end-products.

3.3 Bioavailable phosphorus content

Without P fertilization (control) phosphorus deficiency depressed barley growth drastically due to the low content of plant available P in the experimental soil; barley yield was only 3 g while with the highest SP application rate it was 71 g (Fig. 2). Barley yields after SS and PM applications were less than at the highest application rate of SP (Table 1), ensuring that bioavailability of P in these P sources can be estimated from the yield response curve obtained with SP.
Bioavailable P content in SSL was 68% but only 10% in SSH (Table 1), demonstrating a drastic decrease in P bioavailability after using P precipitation chemicals. For both SS bioavailability of P was at a higher level than labile P contents according to Hedley fractionation scheme, showing that barley was able to utilize less soluble P fractions as well. Bioavailable P content in PM was only 30% (Table 1). This is in contradiction to our previous experiments where PM based P was more available than SP (unpublished results). In this study, flocculants used for separating the solid and liquid phase of PM probably reduced P availability as well.

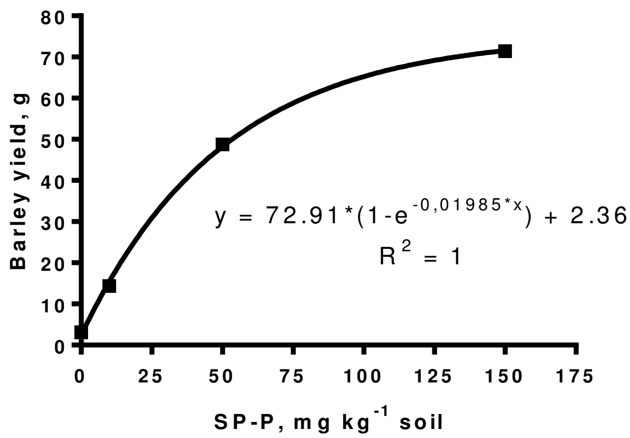

Figure 2: Barley yield in a pot experiment after addition of increasing amounts of superphosphate (SP) phosphorus.

Thermal treatment decreased the bioavailable P content in both SS and PM, but more drastically in SS, where bioavailability was depressed down to 6% and 1% in SSL and SSH, respectively (Table 1). This was due to the formation of less soluble compounds, as shown by the increased share of acid soluble P. Whether these P sources turn into a more bioavailable form in a longer time period is unknown, and growth experiments with a longer time frame are underway to elaborate this question. Share of acid soluble P content in meat and bone meal was up to 90% of the total P, but it turned gradually into a plant available form during a three-year period, instantly plant available P content being 19% [3]. Pyrolysing of PM depressed the bioavailability of P to 17%, which is less than the labile P content (27%) according to Hedley fractionation scheme. This unexpected result needs to be confirmed.

Table 1: Barley yield and bioavailable P share as compared to superphosphate in sewage sludges (SSH and SSL) and in pig manure (PM) before and after HTC-treatment (SSH, SSL) or pyrolysis (PM).

Sample	Barley yield ($g\ pot^{-1}$)		Phosphorus bioavailability, %	
	Before	After	Before	After
SSH	21.1	4.9	10	1
SSL	65.5	14.5	68	6
PM	18.0	11.6	30	17

4. Conclusion and outlook

Both the solubility of P and its bioavailability were depressed by the P precipitation chemicals. Because the bioavailability of P was higher than predicted from the solubility alone. Long-term studies are needed to evaluate P bioavailability of the sewage sludges before and after thermal treatments.
In order to improve recycling of SS based P, use of P precipitation chemicals at the waste water treatment plants needs to be evaluated. Depressing the content of iron in sewage sludge dramati-

cally increases P bioavailability. On the other hand, use of precipitation chemicals or thermal treatment of SS or PM may reduce P leaching risk after high application rates of these materials.

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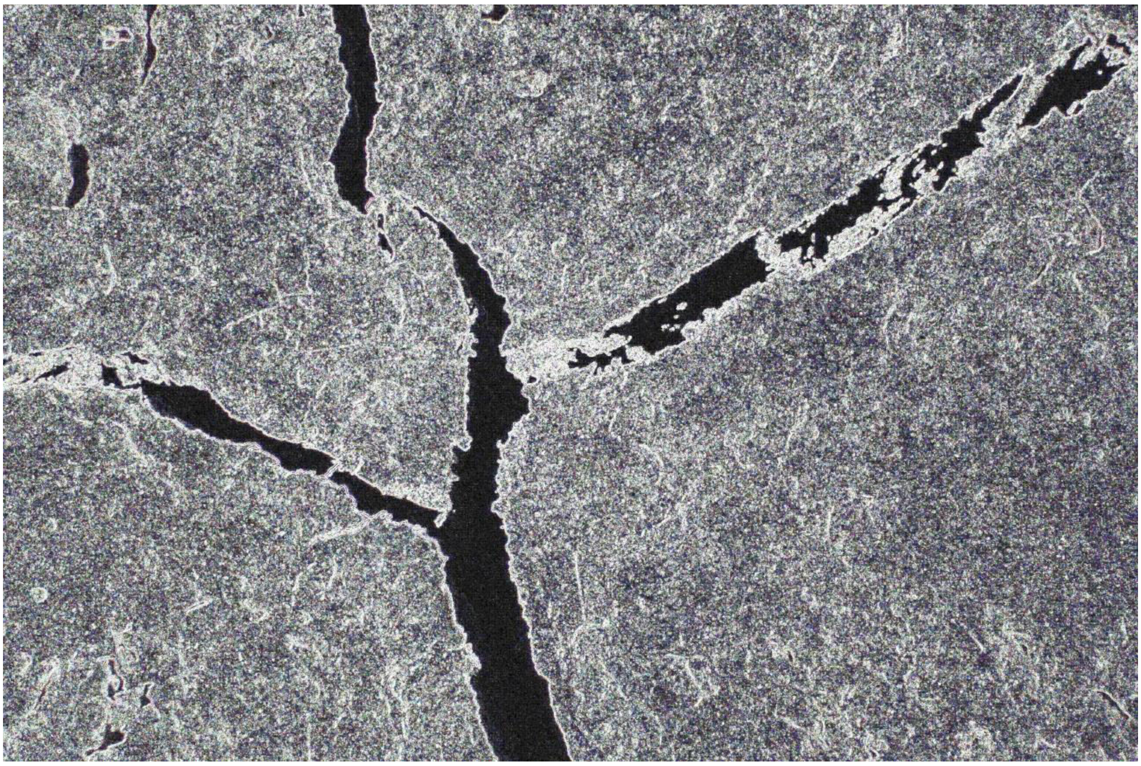
References

[1] Sharpley, A.N.; Moyer, B. (2000): Phosphorus forms in manure and compost and their release during simulated rainfall, Journal of Environmental Quality 29, 1462-1469
[2] Murphy, J; Riley, J.P. (1962): A modified single solution method for the determination of phosphate in natural waters, Analytica Chimica Acta 27, 31-36
[3] Ylivainio, K.; Uusitalo, R.; Turtola, E. (2008): Meat bone meal and fox manure as P sources for ryegrass (*Lolium multiflorum*) grown on a limed soil, Nutrient Cycling in Agroecosystems 81, 267-278

Thematic area TB – Sustainable soils
(Oral presentations)



Ina Körner: Dry soil, processed photograph

TB-K From soil application of sewage sludge to nutrient recycling – A soil science outlook

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1. Objectives

An efficient and as complete recycling of plant nutrients as possible back to arable land is a corner stone of a sustainable society and will characterize future agriculture. Despite many efforts improving recirculation of wastes during recent decades, some major tasks still need to be solved. In this paper, we want to point out some misconceptions about recycling of organic wastes that need to be corrected, stress neglected boundary conditions that must guide our thinking about recycling, and point towards possible solutions. Our position is that nutrient extraction from wastes rather than redistribution of organic residues on arable land enables efficient nutrient cycling in society. This means a conceptual shift from traditional waste recycling to only nutrient recycling. Wastes are used as raw materials for fertilizer production which would enable an equitable redistribution of nutrients back to soil. The aim of this paper is to motivate our position and to provide an example.

2. Conditions determining recycling of organic urban wastes

One can identify three major conditions having a large impact on recycling of organic residues from towns/cities to arable land. These are:

- Gathering of organic residues and accumulation of nutrients in densely populated areas
- Generally high water and low nutrient contents in wastes
- Risk for contamination with non-wanted compounds

It is projected that urban populations will increase from 3.4 billion in 2009 to 6.3 billion individuals in 2050[1]. Consequently, food consumption will mainly occur in densely populated areas. This means that plant nutrients present in food are accumulated in urban settlements. As most food originates from remote agricultural areas and arable fields adjacent to cities only provide a minor portion, equitable redistribution of wastes to cropland would mean long-distance transport. Since water contents in wastes typically range from 55% in mature composts, 70-80% in dewatered sewage sludge and up to 90% in biogas residues, recycling of organic residues means long-term transportation of large waste volumes per amount plant nutrient (Table 1).
As a result, organic residues are mainly recycled within the vicinity of cities and are not only applied to agricultural fields but also to green areas and as feedstock for bioenergy production, mushroom production etc., thus, substituting fertilizers and other resources. Although re-use of organic residues is a big progress compared to disposal, a main challenge of recycling still needs to be addressed, i.e., mining of soils for nutrients in remote areas and enriching nutrients in and around centers of food consumption. Finally, recycling of organic residues from towns/cities can be a risk due to high concentrations of unwanted metals, organic chemicals, pharmaceuticals and pathogens, than in agricultural wastes. In fact, sewage sludge is a sink for all type of compounds used in society ending up in waste water.

Table 1: Overview of phosphorus contents in organic waste products and mineral P fertilizer

Product	P content / % in wet weight	Amount product per amount nutrient / kg per kg P^{-1}
Waste water	0.001	100 000
Human urine	0.05	2 000
Biogas residues	0.075	1 333
Composts	0.15	667
Sewage sludge	1	100
Ash from sewage sludge	9	11
Mono-ammonium phosphate	22	4.5

3. Results and discussion

3.1 Misconceptions about recycling of organic town/city wastes

A central idea for recirculation of organic town/city wastes back to soil still is that organic residues are a valuable and useful soil amendment/fertilizer that should be used in crop or other biomass production. Thus, efforts to produce as 'clean' wastes as possible avoiding contamination with hazardous product has been prioritized. Our thinking that organic wastes are valuable products is based on the two particular perceptions, which are partly incorrect:

- Organic wastes are an important source of organic matter enhancing soil carbon levels and soil structure;
- The nutrient composition in organic wastes is adapted to the demand of crops and wastes act as fertilizer.

How large is the potential input of organic matter through sewage sludge to arable land?
Assuming an application of 700 kg sewage sludge dry matter to soil equivalent to 22 kg P ha^{-1} yr^{-1} (maximum amount according to Swedish legislation) with a carbon content of 30%, means an addition of 210 kg C or 420 kg of organic matter ha^{-1}. However, assuming an equitable redistribution of sewage sludge, i.e., relating the total amount of sewage sludge produced in a country in relation to the total arable land, the amounts of organic matter added are insignificant. For example, Sweden produces 220 000 Mg sewage sludge dry matter and the total area of arable land is 2 500 000 ha, which results in less than 30 kg C or 60 kg organic matter ha^{-1} when sewage sludge is uniformly redistributed. For comparison, crop residues such as roots and stubble amount to 1500-2000 kg C ha^{-1} yr^{-1}. Even considering that humification of sewage sludge is 2-3 times higher than of crop residues[2], the perception that sewage sludge is a significant organic matter source is not supported by data.

Do organic wastes have an optimal composition as fertilizer?
Examining the nutrient composition in sewage sludge, composts or biogas residues shows that ratios of common nutrients, nitrogen (N), phosphorus (P), potassium (K) and sulfur (S), seldom are in agreement with the composition of crops. For example, the N-P-K-S composition in sewage sludge is 1-1-0.1-0.3, whereas the relative nutrient composition in crops is 1-0.1-0.5-0.1 showing that too much P and S and too little K is provided by sludge. Similarly, composts of household wastes (relative N-P-K-S contents of 1-0.2-2.2-0.5) have too low nitrogen contents in relation to other nutrients[3].
In addition, nutrients in organic wastes may not be released in synchrony with crop demand. For instance, several long-term field experiments in Sweden have shown that crop utilization of N applied with sewage sludge is low indicating that nitrogen in sewage sludge is not sufficiently released during crop growth[4]. The same is true for compost[6] having no positive yield effect even when applied at rates of 6000 kg dry matter ha^{-1} yr^{-1} over several years.

3.2 Long-term field experiments with sewage sludge

Metal contents in Swedish sewage sludge have steadily decreased since the 1990th (Fig. 1). Concentrations decreased by 85%, showing that environmental policies to control metal emissions in Swedish society have been effective. As a result, the soil microbial biomass in long-term field studies, which previously was stressed and reduced in size in relation to the carbon content in soil, recovered during recent years (Fig. 2).
Despite the improvement of the quality of sewage sludge, ongoing Swedish long-term field experiments show that crop utilization of N and P applied in sewage sludge is low. This can be explained by the chemical composition of sewage sludge. Nitrogen is mainly bound to stable organic compounds and is therefore slowly released over time. Phosphorus is mainly present in Fe- or Al-precipitated forms, which are only slowly dissolved. In other words, only a small portion of nutrients in sewage sludge is water soluble and therefore plant available. From a farmer's view, sewage sludge is an inefficient fertilizer.
Metal accumulation in soil due to long-term application of sewage sludge to soils did not significantly affect trace metal contents of cereal crops, neither of essential nor of polluting metals[4]. However, sewage sludge acidified soils as a result of oxidation of organic S to sulfate and organic N to nitrate.

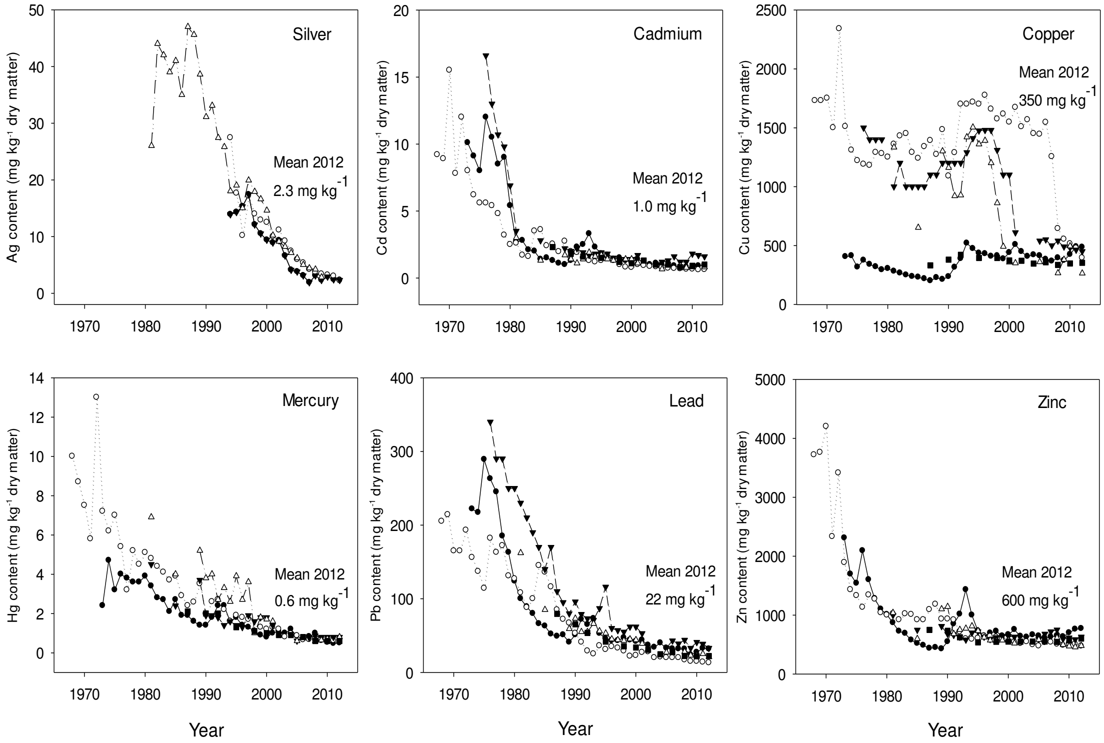

Figure 1: Declining concentrations of six metals in sewage sludge over time[4]. Mean Swedish data for silver were not available. (○ = empty circles refer sludge from Uppsala; • = black circles to sludge from Gothenburg; Δ = empty triangles refer sludge to Stockholm; ▼ = black down pointing triangles refer sludge from Malmö/Lund; ■ = black squares refer to mean data of Swedish sewage sludge).

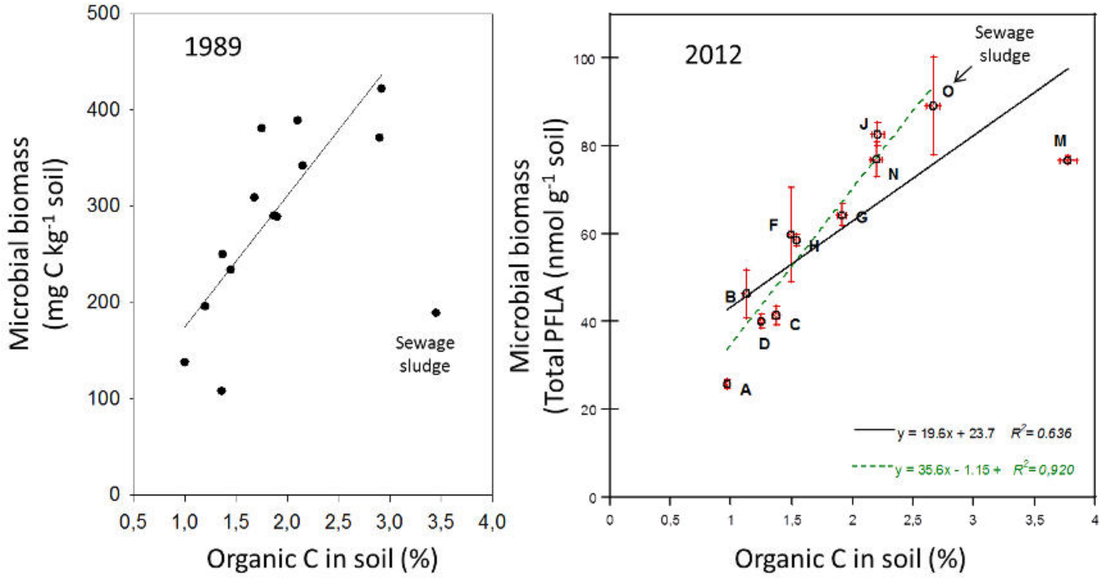

Figure 2: Change of the soil microbial biomass in the Ultuna long-term soil organic matter experiment[4,5].

3.3 Is nutrient extraction from wastes the way forward?

In this part, we discuss a possible solution for a more efficient P-cycling in society. The demographic trend makes towns and cities to be hot spots for accumulation of nutrients. The accumulation of wastes in mega-cities does not enable an efficient recycling of organic wastes anymore and the trend is to combust more and more of the sewage sludge produced, not only in mega-cities but also in towns. We consider ash to be the main waste product from increasing urbanization and processing of ash for nutrient extraction to be an important step to close nutrient cycling in society (Fig. 3). Ashes from combusted sewage sludge have relatively high phosphorus contents (Table 1) and a new process to extract phosphorus from these ashes was developed[7].

When developing the process to extract phosphorus from sewage sludge ash, the aim was to produce fully water-soluble (i.e. plant available), non-metal contaminated (i.e. environmentally adapted) and concentrated inorganic phosphorus fertilizer (i.e. enabling equitable redistribution to arable land). Furthermore, the idea has been to minimize new waste streams when processing ash. In-going chemicals required for ash processing should become part of final products. Internal bleed water should be re-used and out-going water can be released without problems.

Introduction of this process on a wide scale implies among others that a large amount of phosphate rock can be conserved by using phosphorus from ash instead. Such development may significantly contribute to a sustainable use of the world's phosphorus resources and safe and secure food production.

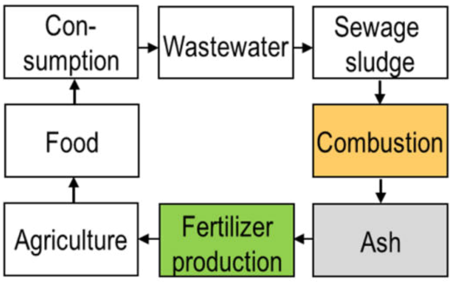

Figure 3: Closing P cycling in society.

4. Conclusion and outlook

Our position is that efficient cycling of plant nutrients in society requires

- equitable redistribution of nutrients on arable land;
- application of nutrients in plant available form;
- use of 'safe and clean' products.
-

These conditions can be achieved through nutrient extraction rather than redistribution of organic wastes on arable land [7]. We suggest that sustainable nutrient management is possible in future through inorganic fertilizers derived from organic wastes.

References

[1] United Nations, 2010. Department of Economic and Social Affairs, Population Division: World urbanization Prospects: The 2009 Revision, New York.

[2] Kätterer, T. et al. 2011. Roots contribute more to refractory soil organic matter per unit carbon than above-ground crop residues as revealed by a long-term field experiment. *Agriculture, Ecosystems and Environment* 141, 184-192.

[3] Eklind, Y. et al. 1997.Chemical characterization of source-separated organic household wastes. *Swedish Journal of agricultural Research* 27, 167-178.

[4] Börjesson, G., Kirchmann, H., Kätterer, T. (2014) Four Swedish long-term field experiments with sewage sludge reveal a limited effect on soil microbes and on metal uptake by crops. *Journal of Soils and Sediments* 14, 164-177.

[5] Witter et al. 1993. Size of the soil microbial biomass in a long-term field experiment as affected by different N-fertilizers and organic manures. *Soil Biology & Biochemistry* 25, 659-669.

[6] Kätterer,T., Börjesson, G., Kirchmann, H. (2014) Changes in organic carbon in topsoil and subsoil and microbial community composition caused by repeated additions of organic amendments and N fertilisation in a long-term field experiment in Sweden. *Agriculture, Ecosystems and Environment* 189, 110-118.

[7] Cohen, Y., Kirchmann, H., Enfält, P. (2011) Management of phosphorus resources – historical, perspective, principal problems and sustainable solutions. In S. Kumar (ed) Integrated Waste Management Volume II, pp 247-268. Open Access. www.intechweb.org.

TB-O_01 Sustaining soil quality by farm compost application and non-inversion tillage, and resulting nitrogen dynamics

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1. Objectives

In vegetable cropping systems, soil quality is seriously threatened by intensive tillage and fertilization practices and by limited crop rotations. Compost application and reduced tillage may help to sustain soil quality. Moreover, these systems are particularly prone to N losses due to inherent low N efficiency of vegetable crops related to superficial rooting, large amounts of crop residues and excessive N fertilization to boost crop productivity and quality (Armbruster et al., 2013). Recommendation of N fertilization should account for effects of soil management practices on N availability. We investigated to which extent compost application and a non-inversion tillage practice sustains soil quality and alters nitrogen (N) dynamics in a multi-year field trial with vegetables as main crops in the rotation. Our research hypothesis was that compost application and reduced tillage enhance soil quality in the short term affecting N availability in the soil and N use efficiency of the mineral N dressing.

2. Methodology

A two-factorial experiment was investigated from September 2008 until February 2012, a period that covered three full growing seasons. The field was located N 50°57´1.91´´, E 3°15´7.79´´, 25 m asl, on a sandy loam soil (63% sand, 30% silt and 7% clay). Farm compost, prepared at our institute (ILVO) based on vegetal residues (e.g. bark, straw, wood chips, grass clippings, leek residues, …)(Table 1) was applied each autumn, starting in 2008, at three different rates, namely 0, 15 and 45 Mg ha^{-1} (henceforth named C0, C15 and C45). The soil was tilled either conventionally (CT) by mouldboard ploughing or by reduced non-inversion tillage (RT) with a chisel plough (Actisol). Soil tillage depth was approximately 0.3 m for both soil tillage practices. Combining two tillage methods and three compost doses resulted in six different soil management regimes which were replicated four times and arranged according to a split-plot design with tillage as the main plot factor and farm compost application as the subplot factor. Individual subplots were 6 m by 18 m. The rotation started with winter rye (*Secale cereale*) as cover crop after the first compost application in 2008. Broccoli (*Brassica oleracea*, var. Italica Group) was grown as main crop in 2009 and followed by white mustard (*Sinapis alba*) as autumn cover crop. In 2010, carrots (*Daucus carota*) were grown. Before the main crop leek (*Allium porrum*) white mustard was used as spring cover crop in 2011. Top fertilization of the leek crop was added as a third factor, i.e. a sub-subplot factor. Calcium ammonium nitrate (27% N) was applied in three different doses, 0, 30 and 60 kg N ha^{-1}, randomized over sub-subplots of 6 by 6 m.

Soil quality was assessed at the start and at the end of the experiment based on chemical and biological characteristics. Undisturbed soil cores (100 cm^3) were taken with an auger (Eijkelkamp Agrisearch Equipment) for determination of bulk density (BD) at approximately 0.05 m, 0.2 m and 0.45 m below the soil surface (ISO 11272) for the 0-0.1, 0.1-0.3 and 0.3-0.6 m layer, respectively. Total organic carbon (TOC) concentration was measured in oven-dried (70 °C) soil samples by dry combustion at 1050 °C with a Skalar Primacs SLC TOC-analyzer according to ISO 10694. pH was measured potentiometrically in a 1 M KCl solution (1:5 v/v) according to ISO 10390. Changes in chemical soil properties over the three-year period were calculated (e.g., ∆TOC). Phospholipid fatty acids (PLFAs) were extracted and determined using techniques fully described in Moeskops et al. (2010). The soil layers 0-0.3, 0.3-0.6 and 0.6-0.9 m were sampled separately for analysis of the mineral N content, yearly at three sampling occasions, i.e. before the start (s1), during (s2) and at the end of the cultivation period (s3)(Table 2). Sampling was done per subplot, except for broccoli at s2 where samples were taken per main plot and for leek at s3 where samples were taken per sub-subplot. Soil mineral N content was extracted (1:5 w/v) in a 1 M KCl solution according to ISO 14256-2 and measured with a Foss Fiastar 5000 continuous flow analyzer. Soil moisture content was determined as weight loss at 105 °C. For the calculation of the mineral N stock (Nmin) in the different layers, average BD values of 0-0.3 and 0.3-0.6 m soil layer were used, determined per subplot at s2 in 2011. BD was neither affected by soil tillage nor by compost application. BD of the 0.6-0.9 m layer was assumed to be identical to the BD of the 0.3-0.6 m soil layer. Crop total biomass and N uptake were determined at s2 and s3 for broccoli and leek, whereas for carrots, only gross tuber yield was determined at s3. To determine broccoli and leek dry matter content, crop subsamples of whole plants were dried in a ventilated oven at 70°C during at least 48h. The N content was determined on ground dried plant material according to the Kjeldahl method (ISO 5983-2) for broccoli and the Dumas method (ISO 16634-1) for leek. The N use efficiency (NUE) of the top mineral N dressing for leek was determined by subtracting the N uptake$_{s2\text{-}s3}$ on the non-fertilized sub-subplots from the N uptake$_{s2\text{-}s3}$ on the fertilized sub-subplots and dividing this by the dose.

Table 1: Composition of the farm compost applied in the years 2008-2010 DM: dry matter; OM: organic matter; EC: electrical conductivity

	DM	OM	EP	Ph-H_2O	N-NH_4	N-NO_3	N	P	K
	%	% on DM	µS cm^{-1}	-	mg L^{-1}	% on DM	g kg^{-1}	g kg^{-1} DM	g kg^{-1} DM
2008	23.1	69.7	901	9.4	7.5	93.5	2.0	3.3	16.6
2009	38.3	43.4	615	8.2	< 5.0	24.0	1.2	1.5	7.4
2010	65.6	51.4	1308	8.4	< 5.0	46.5	1.7	3.2	10.7

Table 2: Calendar data sampling occasions s1, s2 and s3

s1	19.03.2009	14.04.2010	14.06.2011
s2	15.06.2009	28.06.2010	24.08.2011
s3	29.07.2009	27.09.2010	08.11/2011

3. Results and discussion

3.1 Soil quality parameters

A significant interaction between the tillage and layer factor was found for TOC ($p < 0.001$), indicating a stratification of soil organic matter due to a reduced tillage practice. For the 0-0.1 m soil layer, TOC tended to be higher under RT compared to CT. RT did not reduce TOC$_{0.1–0.3\ m}$. Under RT, TOC$_{0–0.1\ m}$ was significantly higher than TOC$_{0.1–0.3\ m}$ ($p < 0.001$) whereas in the case of CT, the TOC of both upper layers did not differ significantly (Table 3). No significant effect from the soil tillage factor on pH-KCl was found.

Table 3: Total organic carbon content (TOC); significant differences between soil layers are indicated with different lowe case letters and p-values (2-way ANOVA: compost x layer); soil layers 0-0.1 m, 0.1-0.3 m and 0.3-0.6 m; CT: conventional tillage (ploughing), RT: reduced (non-inversion) tillage.

		0-0.1 m	0.1-0.3 m	0.3-0.6 m	ANOVA	Scheffe
TOC	CT	0.88 ^{b}	0.90 ^{b}	0.61 ^{a}	0.001	0.001
%	RT	1.05 ^{b}	0.93 ^{b}	0.61 ^{a}	0.001	0.001

TOC$_{0\text{-}0.1\ m}$ was significantly higher for C45 than for C0 ($p < 0.01$)(Figure 1). A tendency for higher TOC$_{0.1\text{-}0.3\ m}$ with increasing compost doses was observed. ΔTOC$_{0\text{-}0.1\ m}$ was significantly different between C0 and C45 ($p < 0.01$). No change in TOC$_{0.1\text{-}0.3\ m}$ was observed for C45, but there was a

significant decline in TOC in the 0-0.1 and 0.1-0.3 m soil layers on C0 plots ($p < 0.05$ and $p < 0.001$). A significant increase in TOC in the 0.3-0.6 m soil layer was observed in compost amended plots ($p < 0.01$). From the results, it appeared that the highest compost dose more than compensated for organic matter losses by mineralization in the arable layer. pH-$KCl_{0\text{–}0.1\ m}$ was significantly higher on compost amended plots than on non-amended ones ($p < 0.01$). The decrease in pH was considerably limited by compost application, irrespective of the dose applied (Figure 2).
Application of compost and non-inversion tillage clearly resulted in more soil microbiota in the 0-0.1 m top layer (Table 4). For most of the functional groups in the soil food web, both RT and compost application (C45) were beneficial. However, fungi were only favored by RT and gram-negative (G-) bacteria only by compost application (C45).

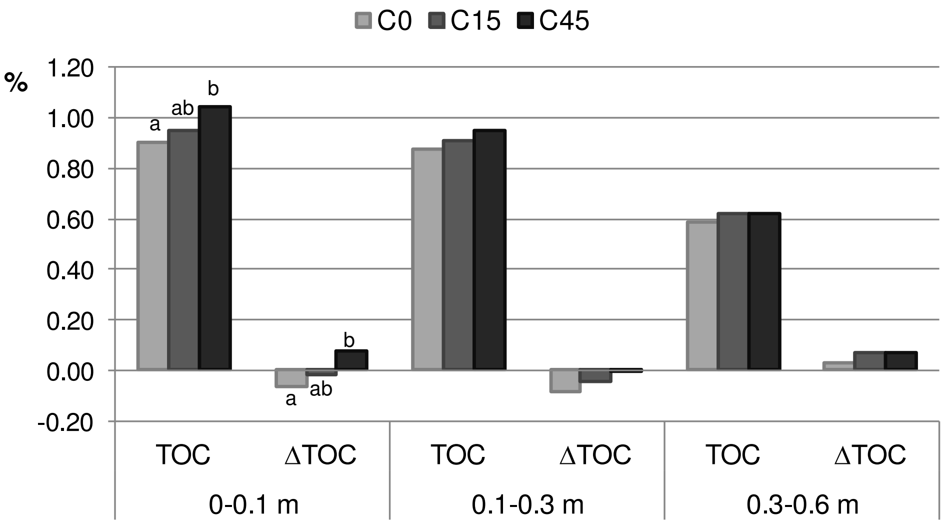

Figure 1: Total organic C content (TOC) at the end of the three-year experimental period and change in TOC (ΔTOC) during that period in the 0-0.1, 0.1-0.3 and 0.3-0.6 m soil layers for the different compost doses 0, 15 and 45 Mg ha^{-1} $year^{-1}$ (C0, C15 and C45, respectively). Significant differences are indicated by different lowercase letters.

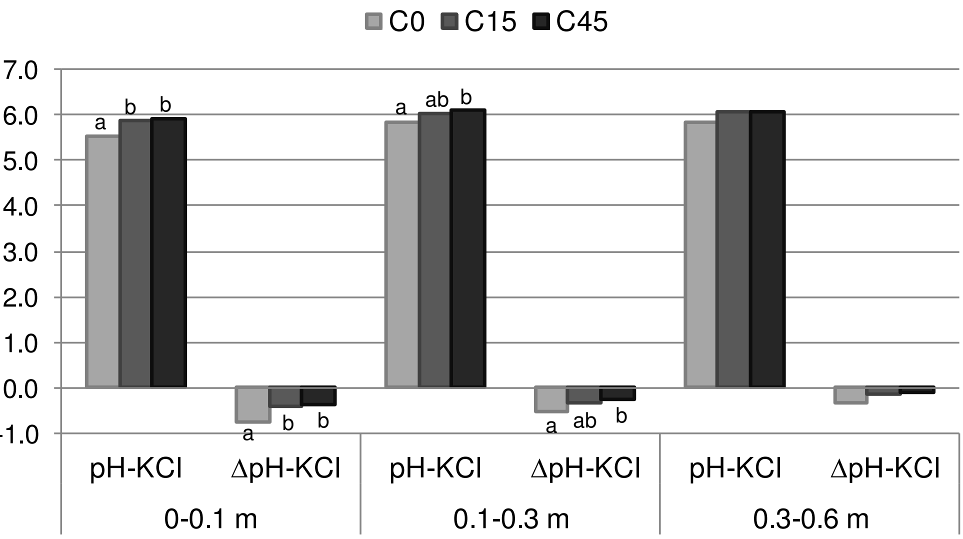

Figure 2: pH-KCl at the end of the three-year experimental period and change in pH-KCl (ΔpH-KCl) during that period in the 0-0.1, 0.1-0.3 and 0.3-0.6 m soil layers for the different compost doses 0, 15 and 45 Mg ha^{-1} $year^{-1}$ (C0, C15 and C45, respectively). Significant differences are indicated by different lowercase letters.

Table 4.: Final soil status with regard to soil biota in the 0-0.1 m soil layer (total microbial biomass (Total), gram-positive (G+) and gram-negative (G-) bacteria, actinomycetes, fungal PLFA marker 18:2ω6 and arbuscular mycorrhizal fungi (AMF); significant differences between tillage methods or compost doses are indicated by different lowercase letters and p-values (2-way split-plot ANOVA tillage x compost); CT: conventional tillage (ploughing), RT: reduced (non-inversion) tillage; C0, C15, C45: 0, 15 and 45 Mg farm compost ha^{-1}.

nmol g^{-1}	CT	RT	ANOVA	C0	C15	C45	ANOVA	Scheffe
Total	14.1 ^{a}	20.3 ^{b}	0.05	15.5 ^{a}	16.5 ^{a}	19.6 ^{b}	0.001	0.05
G+ bacteria	1.60 ^{a}	3.51 ^{b}	0.05	2.69 ^{a}	2.92 ^{a}	3.56 ^{b}	0.01	0.05
G- bacteria	1.59	2.01		1.65 ^{a}	1.70 ^{a}	2.05 ^{b}	0.01	0.01
Actinomycetes	1.12 ^{a}	1.54 ^{b}	0.05	1.21 ^{a}	1.25 ^{a}	1.54 ^{b}	0.001	0.05
Fungi 18:2ω 6	0.34 ^{a}	0.77 ^{b}	0.01	0.53	0.53	0.61		
AMF	0.66 ^{a}	1.11 ^{b}	0.05	0.72 ^{a}	0.84 ^{a}	1.10 ^{b}	0.001	0.05

3.2 N dynamics

Only in the second year (2010) the tillage and compost factor significantly affected $Nmin_{0\text{-}0.3\ m}$ (Table 5). At s1, $Nmin_{0\text{-}0.3\ m}$ was significantly higher for C45 compared to C0 (two compost applications) but was not affected by the difference in tillage practice in year 1. $Nmin_{0\text{-}0.3\ m}$ at s2 was significantly higher in case of CT compared to RT and significantly higher for C45 compared to C0. In 2010, at s1, $Nmin_{0\text{-}0.9\ m}$ was significantly higher for C45 (103 kg ha^{-1}) than for C0 (84 kg ha^{-1})($p < 0.05$). At none of the others sampling occasions in the three-year experimental period, significant differences in $Nmin_{0\text{-}0.9\ m}$ appeared.

Table 5: Mineral N stock in the 0-0.3 m soil layer ($Nmin_{0\text{-}0.3\ m}$) in the carrots growing season 2010; significant differences between tillage methods or compost doses are indicated by different lowercase letters and p-values (2-way split plot ANOVA tillage x compost); CT: conventional tillage (ploughing), RT: reduced (non-inversion) tillage; C0, C15, C45: 0, 15 and 45 Mg farm compost ha^{-1}; s1, s2 and s3: first, second and third sampling occasion

kg ha^{-1}	CT	RT	ANOVA	C0	C15	C45	ANOVA	Scheffe
s1	34	31		29 ^{a}	31 ^{ab}	38 ^{b}	0.01	0.01
s2	67 ^{b}	58 ^{a}	0.001	57 ^{a}	62 ^{ab}	68 ^{b}	0.05	0.05
s3	13	19		12	21	15		

As differences in plant N availability hardly occurred, differences in crop N uptake were not observed in the three-year experimental period except in 2009 where total N uptake by the broccoli crop was significantly higher under RT (212 kg ha^{-1}) compared to CT (196 kg ha^{-1})($p < 0.01$). At s2 in 2011, an interaction between the tillage and layer factor indicated a difference in distribution of mineral N in the soil profile between tillage systems. Under RT, $Nmin_{0\text{-}0.3\ m}$ was significantly higher than $Nmin_{0.3\text{-}0.6\ m}$, whereas under CT, $Nmin_{0\text{-}0.3\ m}$ did not differ from $Nmin_{0.3\text{-}0.6\ m}$ (Table 6). $Nmin_{0\text{-}0.3\ m}$:$Nmin_{0\text{-}0.9\ m}$ was significantly higher under RT (52.3%) compared to CT (45.1%)($p < 0.05$).

Table 6: Mineral N stock (Nmin) in the soil profile in the leek growing season 2011; significant differences between soil layers are indicated by different lowercase letters and p-values (3-way split-plot ANOVA tillage x compost x layer); soil layers 0-0.3, 0.3-0.6 and 0.6-0.9 m and soil profile 0-0.9 m; s1, s2 and s3: first, second and third sampling occasion

kg ha^{-1}		0-0.3m	0.3-0.6m	0.6-0.9m	ANOVA	Scheffe	0-0.9m
s1		22	16	21			58
s2	CT	97 [b]	93 [b]	27 [a]	0.001	0.001	218
	RT	115 [c]	77 [b]	26 [a]	0.001	0.001	217
s3		21 [a]	33 [b]	27 [b]	0.001	0,05	80

NUE from top mineral N dressing was neither affected by compost application nor by tillage practice and did not differ between both doses of top mineral N dressing. On average, NUE was only 34% and a large variation was observed (standard deviation +/- 86%). NUE assessment was complicated by a rather high and highly variable Nmin at s2. Residual mineral N at the 60 kg N ha^{-1} dose (114 kg Nmin ha^{-1}) was significantly higher compared to residual mineral N at the zero and 30 kg N ha^{-1} dose (80 and 76 kg Nmin ha^{-1}, resp.) at an equal total N uptake for all doses of top mineral N dressing, which indicates that N utilization of the top mineral N dressing was low or non-existent.

4. Conclusion and outlook

Farm compost application and non-inversion tillage counteracted soil degradation, which was otherwise inevitable in this intensive vegetable cropping system in the three-year study period. Adopting these soil improving practices hardly affected plant N availability in the short term and therefore fertilization schemes must not be adapted. The risk of N losses by leaching was hardly affected by farm compost application. A higher share of the mineral N stock in the upper soil layers under a reduced tillage practice might reduce the risk of N losses by leaching compared to a conventional tillage practice.

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References

[1] Armbruster, M.; Laun, N.; Heger, A.; Wiesler, F. (2013): Nitrogen efficiency in intensive field vegetable production. In: Fontana, E., Grignani, C., Nicola, S. (Eds.) International Workshop on Nitrogen, Environment and Vegetables, B. Abstr., Dipartimento di Scienze Agrarie, Forestali e Alimentari. Università degli Studi di Torino, 41-42

[2] Moeskops, B.; Sukristiyonubowo; Buchan, D.; Sleutel, S.; Herawaty, L.; Husen, E.; Saraswati, R.; Setyorini, D.; De Neve, S. (2010): Soil microbial communities and activities under intensive organic and conventional vegetable farming in West Java, Indonesia. Appl. Soil Ecol. 45, 112-120

[3] Willekens, K.; Vandecasteele, B.; Buchan, D.; De Neve, S. (2014): Soil quality is positively affected by reduced tillage and compost in an intensive vegetable cropping system, Applied Soil Ecology 82, 61-71

[4] Willekens, K.; Vandecasteele, B.; De Neve, S. (2014): Limited short-term effect of compost and reduced tillage on N dynamics in a vegetable cropping system, Scientia Horticulturae 178, 79-86

TB-O_02 Preplant compost application improves landscape plant establishment and sequesters carbon in compacted soil

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1. Objectives

Compost use is advocated as a component of sustainable landscape management in urban areas. Compost is often added to other soil components to provide a target level of organic matter in "topsoil" in urban projects. This study was conducted to (i) assess the potential benefit of compost application for a variety of landscape plant species, (ii) to evaluate the need for tillage following compost application, and (iii) to assess the longevity of compost effects on soil pH, nutrients, compaction, and soil carbon (C). To simulate soil compaction associated with typical urban construction activities, soil was mechanically compacted at the start of this trial. We hypothesized that preplant compost application would assist in plant establishment in compacted soil.

Composts were derived from urban sources: 1) municipal biosolids (from wastewater treatment) mixed with conifer sawdust, or 2) municipal yard debris (tree, shrub and grass trimmings from urban landscape maintenance). Although municipal composting has been common locally since the 1990s, quantitative understanding of long-term compost effects on urban landscapes is limited [1-4].

2. Methodology

Composts met environmental protection requirements for use in landscaping or gardening, and had typical chemical characteristics, reflecting differences in compost feedstocks (Table 1). The biosolids compost was made from a 3:1 v/v mix of conifer sawdust: dewatered municipal biosolids from the City of Newberg, OR. The yard debris compost was the product of a commercial composting facility (Rexius Forest Products, Eugene, OR). It was screened to remove sticks and large chunks of organic matter, and sold as a "Garden Compost" product. The biosolids compost had higher NH_4-N, P, and trace element concentration, and lower K than the yard debris compost. (Table 1). Soluble salts, as indicated by electrical conductivity were <2 dS/m, indicating suitability for a wide range of plants. Compost stability (resistance to decomposition) was high, as indicated by the respiration (CO_2 evolution) test. Both composts had typical C:N (near 20) for composts derived from woody feedstocks.

Table 1: Analysis of composts derived from municipal biosolids or municipal yard debris. Composts were applied to field plots at a depth of 7.5 cm (760 m^3/ha).

Compost source	Bulk density	C: N	Total C	Total N	NH_4-N	Total P	Total K	pH (1:5)	EC (1:5)	Stability (CO_2 evolved)
	kg DM/m^3		------	------	kg/m^3	------	------		dS/m	mg C/g OM/d
Biosolids compost	280	23	118	5.0	0.6	2.7	0.4	7.7	1.6	1.0
Yard debris compost	390	19	101	5.5	0.02	1.0	2.8	7.0	0.9	2.9

Compost analysis by Soil Control Lab, Watsonville, CA, USA using standard methods (U.S. Composting Council):
Bulk density = kg compost dry matter (DM) per cubic meter
Compost pH and electrical conductivity (EC) determined via 1:5 (compost:water; g DM/mL) method.
Stability = CO_2 evolution via 3 day incubation at 25 ^{o}C, expressed as mg C evolved per g organic matter (OM) per day.

The field trial was conducted at the OSU North Willamette Experiment Station (Aurora, OR, USA). Climate is Mediterranean, with a wet winter (Nov-Mar), and summer drought in July and August. Eight perennial plant species were installed prior to the winter rainy season (Sept 2008). No summer irrigation was applied during any year. To simulate soil compaction often present at urban sites, an agricultural soil (Willamette silt loam) was prepared by compacting with a vibrator-roller (typically used for parking lot construction) to a bulk density of 1.5 g/cm^3 (5 to 10 cm depth), as measured with a bulk density probe. Soil compaction was performed 5 days after 10 cm of overhead sprinkler irrigation. The vibrator-roller traversed the field 5X. After soil compaction, it was difficult or impossible to dig planting holes manually.

Experimental design was a 3 x 2 factorial with three compost treatments (biosolids compost, yard debris compost, no-compost control) and two compost placements (no tillage or rototilling after compost application), replicated 4X, for a total of 48 field plots. Field plots contained 3 transplants of each of 4 plant species (12 plants per plot). We installed 4 cultivars considered standard landscape plants for our area (*Nandina domestica* 'Compacta', *Vinca major* 'Bowles', *Viburnum davidii*, *Berberis thunbergii* 'Crimson Pygmy'), and 4 cultivars considered more drought-tolerant: *Rosmarinus officinalis* 'Blue Spires', *Cistus* 'Bicolor Pink', *Ceanothus gloriosus*, *Caryopteris x clandonensis* ‘First Choice’. Each field plot (3 x3 m) was surrounded by a 1.5-m border area that was seeded to turf-type tall fescue.

Prior to transplanting, compost (7.5-cm depth) was applied, followed by rototilling 5 to 10 cm depth (for tillage treatment only). Planting holes were then drilled with a 15-cm diameter screw-type power auger. Nursery transplants from 4-L pots were placed into the drilled holes, and then secured by placing a thin (<5 cm) layer of soil around transplant root cylinders. The soil used to refill planting holes contained incidental amounts of compost from the soil surface. Transplanting was performed by contract laborers who were instructed to work at a typical pace for commercial landscape operations. After transplanting, all plots were mulched with a 5-cm of Douglas-fir conifer bark, a standard practice for weed control in landscape planting beds.

Figure 1: Field trial installation, Sept 2008. Soil compaction with vibrating roller (a), tillage treatment (b), plant installation with power auger in no-till treatment (c), all plots planted and mulched with Douglas-fir bark (d).

Four of the eight plant species were harvested for dry weight determination at 13 months after installation. Plants were cut at the soil surface, and underlying soil was not disturbed. New transplants were installed to replace the harvested plants in March of 2010. Planting holes for the new (2010) transplants were offset at least 30 cm away from previous plant locations. The compost and mulch present on the soil surface was kept out of the new planting holes.

Each October, prior to heavy fall rains, mineral soil (0-20 cm) samples were collected using a 2-cm diameter soil probe. After scraping away the surface organic mulch, soil samples were collected from the area within each plot that was not disturbed by replanting in 2010. Soil testing was performed using standard procedures at Brookside Laboratories Inc. (New Bremen, OH, USA). Carbon was determined by a combustion analyzer, and organic matter was determined by loss-on-ignition (weight loss in a muffle furnace). Soil compaction determined using a recording penetrometer during the rainy season, after the soil profile reached field capacity.

3. Results and discussion

3.1 Plant response to compost

Regardless of treatment, over 90% of plants survived the first growing season. Compost increased plant growth for all plant species, including those species considered drought tolerant. Tillage did not affect plant survival or growth. Biosolids compost increased aboveground plant dry weight by 60 to 130% (average 84%) across the four plant species harvested at 13 months after installation (Oct 2009). Yard debris compost increased plant dry weight by 26 to 93% (average = 54%). Plants that grew the most in response to compost application had high leaf N, suggesting that they were responding to the N supplied by compost. A few plant species suffered greater summer injury with compost, because they had a second flush of vegetative growth in June, and continued growth into July, when days were hot and surface soil moisture was depleted.

When compost was not incorporated by tillage, it was visible on the soil surface for several years after application. A few plant species suffered partial dessication in the dry summer because of extensive rooting at the soil surface. But, even those plants recovered, and overall, plant growth was similar with or without preplant tillage.

Plant growth in this trial was not strongly affected by preplant soil compaction. A "border plot" (adjacent to experimental area, not part of experiment) that had been treated similarly (same plants installed at same time, but soil left uncompacted) had very similar plant growth response to compost.

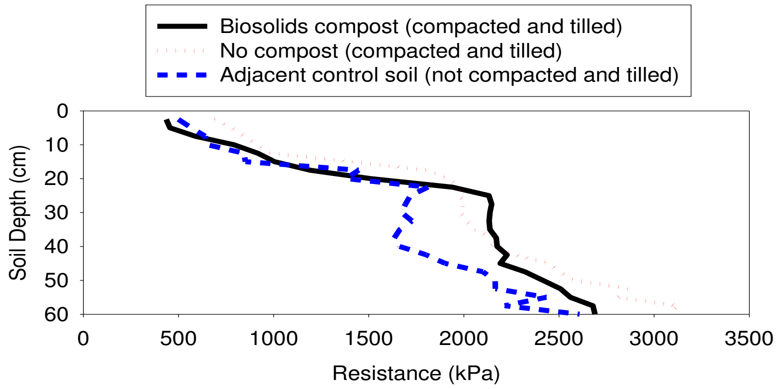

Figure 2: Resistance measured by recording penetrometer in wet soil at 16 months after preplant soil compaction (Dec 2009). Most of the residual compaction created by the vibrator-roller machine (preplant) was observed below 20 cm. Resistance was not affected by compost application.

3.2 Soil response to compost

Compost application did not alter surface soil compaction at 16 months after trial installation as measured by penetrometer (Figure 2), or by bulk density cores. Soil bulk density (0 to 15 cm) was the same with or without compost application (1.38 g/cm^3), but it was reduced by tillage from 1.38 to 1.30 g/cm^3. Penetrometer resistance readings reached 2000 kPa at 25 cm depth for all treatments. Preplant soil compaction increased soil resistance below 25 cm depth to values greater than recommended for optimal plant root growth. However, roots were confined to the top 20 to 30 cm soil for all plant species, based on visual observations made when the experiment was terminated in 2014 (plants removed with an excavator). We conclude that nutrients supplied by compost were the primary drivers of plant growth response (not compaction). When soil sampling in 2012, we noticed that soil was softer (easier to probe) and had evidence of worm activity in the compost-treated plots. The worms likely crawled from the adjacent grassy plot borders to our field plots.

Soil pH, nutrients and organic matter (C) were equivalent for both compost placements (surface or tilled-in), suggesting downward migration of surface-placed compost constituents. Biosolids compost supplied more plant-available nitrogen (N) than yard debris compost, as evidenced by higher soil nitrate-N (Figure 3). First-year N supplied by biosolids compost was excessive (2009), but it was much closer to plant needs in the second growing season (2010). Biosolids compost acidified soil, while yard debris compost increased soil pH slightly (Figure 3). Yard debris compost increased exchangeable cations: K +100%, Ca +20%, and Mg +33%, in the final soil sampling in 2012. With biosolids compost, exchangeable cations remained within 10% of values for the no-compost control.

Organic matter supplied by compost persisted throughout the four year measurement period. Fluctuation in measurements across years was probably the result of inconsistency in sample collection or laboratory analysis techniques. Organic matter (Loss-On-Ignition; LOI) or soil

C (combustion) analyses showed the same trend: yard debris compost > biosolids compost > control. Organic matter averaged 10 to 16% greater than control with biosolids compost, and 29 to 41% greater with yard debris compost. The main constituent of biosolids compost (by weight) was conifer sawdust, which was apparently decomposed more rapidly than the mixture of organic materials present in yard debris compost. .The average increase in soil C (0-20 cm depth) was 5% of compost-C applied for biosolids compost and 18% for yard debris compost (Table 2), assuming a soil bulk density of 1.3 g/cm^3.

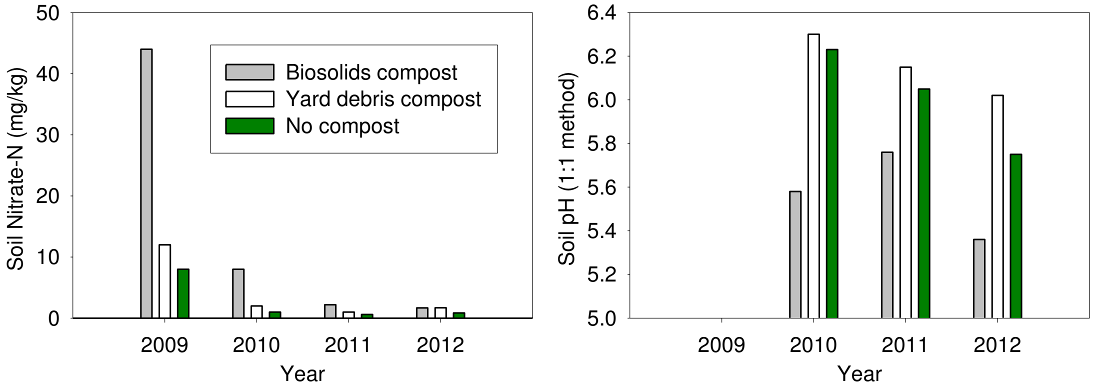

Figure 3: Soil nitrate-N and soil pH (0-20 cm depth). Samples collected in October of each year (end of dry season).

Table 2: Effect of preplant compost application on soil carbon (determined by combustion analysis) and soil organic matter (determined by Loss-On-Ignition; LOI). Compost applied Sept 2008. Soil (0-20 cm) sampled in Oct. each year.

	2009	2010	2011	2012	Average	Percent of control
			Carbon %			
Biosolids compost	1.8	1.5	1.7	1.8	1.7	116
Yard debris compost	1.6	1.6	2.3	2.7	2.1	141
No compost	1.5	1.4	1.4	1.6	1.5	
			LOI Organic Matter %			
Biosolids compost		4.9	3.5	3.4	3.9	110
Yard debris compost		5.1	4.1	4.7	4.7	129
No compost		4.7	3.1	3.1	3.6	

4. Conclusion and outlook

Preplant compost application increased plant growth and visual appearance of a wide variety of perennial landscape plants. The positive plant growth response to compost was similar for compost left on the soil surface or compost incorporated via rototilling. Compost application did not alleviate soil compaction. Plant growth response to compost in this trial was primarily related to N supply (and perhaps other nutrients). Biosolids compost acidified soil, this characteristic improved growth of an acid-loving plant species. Carbon sequestered in soil (5 to 18% of compost-C) was at the low end of C sequestration values reported in the literature.

Acknowledgements

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References

[1] Brewer, L.J. and D.M. Sullivan. 2003. Maturity and stability evaluation of composted yard trimmings. Compost Sci. Util. 11(2): 96-112.

[2] Cogger, C. 2014.Using biosolids in gardens and landscapes. FS156E. Washington State University Cooperative Extension. http://cru.cahe.wsu.edu/CEPublications/FS156E/FS156E.pdf

[3] Cogger, C.G. 2005. Potential compost benefits for restoration of soils disturbed by urban development. Compost Science & Utilization 13.243-251.

[4] Cogger, C.G. et al. 2008. Soil and redosier dogwood response to incorporated and surface-applied compost. HortScience 43: 2143-2150.

TB-O_03 Organic matter stability and accessibility characterization – Towards a tool for organic residue diagnostic before land spreading

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1. Objectives

Considering a zero waste production objective, the recycling of organic residues (with or without previous biological treatment) on cropped soils is becoming crucial for the valorization of carbon and nutrients. An indicator of residual organic carbon (IROC) [1] already exists based on fiber fractionation, and has been used to predict the evolution of soil organic matter after repeated applications of various organic residues [2]. However, it is not applicable to all organic residues [3]. Besides, it has been shown that the chemical nature of extracted organic fractions change during treatment like composting which explained the modification of biodegradability of the concerned organic fractions [4]. Therefore, this study presents the ability of a new methodology [6] to predict the IROC indicator. The main objective is to show that this method is able to (i) describe a wide diversity of organic wastes in terms of stability and accessibility, (ii) include both successive chemical extraction and characterization of the nature of extracted organic fractions in the prediction of behavior after application on soil and (ii) assess the efficiency and the effectiveness of organic waste treatment before its land spreading.

2. Methodology

Several organic residues were characterized with a new methodology to assess both their accessibility (fractionation) and complexity (fluorescence spectroscopy). They were also incubated with agricultural soil in order to assess their residual carbon and their mineralization rates. Thanks to statistical analysis, correlations between characterization and incubation parameters were investigated.

- Samples Samples set:42 samples were characterized in this study: 18 digestates (obtained after mesophilic anaerobic digestion of sludge, manure, agriculture, and municipal organic waste), 7 composts (digestates, sludge), 3 urban wastewater treatment sludge, 4 municipal organic wastes, 2 agricultural biowastes, 2 manures, 2 soils and 1 growing mix
- Chemical fractionation: A recently published method aiming to characterize the organic matter evolution during anaerobic digestion of municipal sludge [5] was extended to a wider range of biologically treated organic residues. This methodology is based on chemical extractions that indicate the chemical accessibility of organic residues. Initially, these extractions were made for sludge, mainly composed of proteins. But, the application of this protocol to lignocellulose-like compounds was not possible because of the high amount of carbohydrates not extracted. To overcome this issue, the Van Soest fractionation and this extraction methodology were merged to characterize all the organic residues type. The new extraction protocol [6] was based on [5] with one main modification which is the addition of the acid extraction from Van Soest protocol (Van Soest, 1963) in order to extract the carbohydrates (i.e. cellulose and cellulose). The sequential extractions (with 30 mL of each extractant) were performed on 0.5 g of dried and ground (1mm) sample. The obtained fractions are: (1) Soluble extractables from Particular Organic Matter (SPOM) (milli-Q water solution containing 10 mM of $CaCl_2$, 15 min × 4, 30°C, 300 rpm) (2) Readily Extractable Organic Matter (REOM) (NaOH 10 mM, 15 min × 4, 30°C, 300 rpm) (3) Slowly Extractable Organic Matter (SEOM) (NaOH 0.1 M, 4 h × 4, 30°C, 300 rpm), (4) Poorly Extractable Organic Matter (PEOM) (25 mL H_2SO_4, 72%, 3 h × 2, 30°C, 300 rpm) and (5) Non-Extractable Organic Matter (NEOM). Results are expressed in COD ($gO_2.gTS^{-1}$).
  These chemical extractions allowed to assess the chemical accessibility of the organic wastes, and to solubilize the organic matter in order to analyze its complexity by 3D fluorescence spectroscopy, as shown in Figure 1. This modified characterization methodology was relevant enough to classify a large panel of organic wastes according to their nature, their

complexity and their proportions of the successive less and less accessible fractions [6]. This result showed the relevance of such a tool to characterize and classify the organic residues according to their accessibility and their complexity.

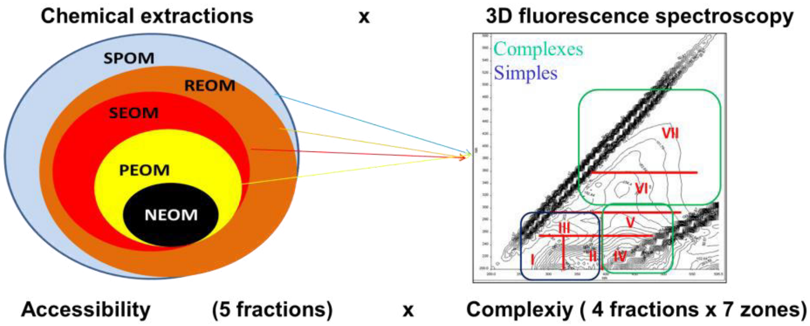

Figure 1: Methodology for accessibility and complexity characterization of organic matter from organic wastes

- 3D fluorescence spectroscopy: Similar to the procedure developed in [5], 3D fluorescence spectroscopy was applied on the extracted fractions, from which organic matter complexity could be quantified.The fluorescence spectrometer used was a Perkin Elmer LS55. Excitation wavelengths varied from 200 to 600 nm with increments of 10 nm. Based on [5], spectra were decomposed in seven zones corresponding to biochemical families-like fluorescence. The proportion of fluorescence of a zone "i" $P_f(i)$ was calculated from the fluorescence zone volumes $V_f(i)$ according to the Equations 1 and 2:

$$V_f(i)(U.A./mg.COD.L^{-1}) = \frac{V_f(i)}{COD_{sample}} \times 1 \Big/ \frac{S(i)}{\sum_{i=1}^{7} S(i)} \qquad \text{Eq. 1}$$

$$P_f(i)(\%) = \frac{V_f(i)}{\sum_{i=1}^{7} V_f(i)} \times 100 \qquad \text{Eq. 2}$$

with:

$V_f(i)$ (U.A. mg.COD.L^{-1}) :	the raw volume of the zone i,
COD_{sample} (mg.L^{-1}) :	the COD concentration of the sample,
S(i) (nm^2) :	the area of a zone i,
$P_f(i)$ (%) :	the fluorescence proportion of a zone i.

- Soil incubation test: :10 g of dried and ground organic waste was incubated with 100 g of cropped soil during 90 days under moisture and temperature controlled conditions. Carbon mineralization was monitored using a NaOH (1N) solution for CO_2 trap. Cumulated mineralized carbon was obtained as a percentage of initial TOC (cf. Figure 2). Thanks to the Gompertz equation (Equation 3), identification of cumulated curves parameters was possible, such as µ the mineralization rate (in % of mineralized C/day), τ the lag phase time (days) and C_{bio} the maximum mineralized C (gC.gC^{-1}).

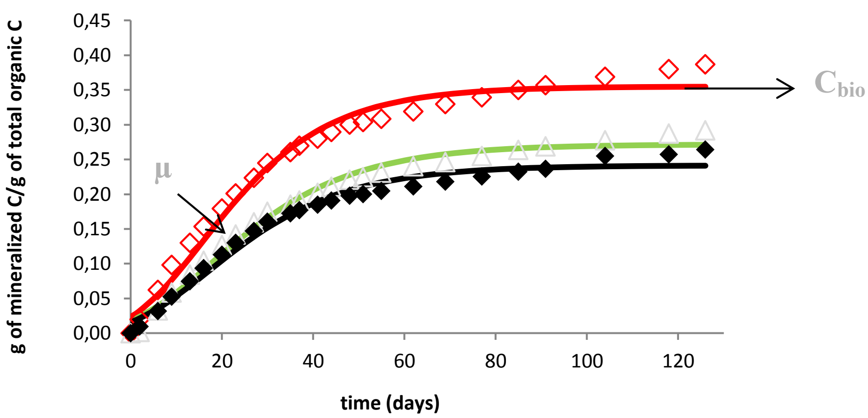

Figure 2: Example of mineralization curves obtained for several samples (red: sludge, green triangles: digested sludge, black: composted sludge)

$$Mineralized\ C = C_{bio} \times e^{-e\left(\frac{\mu \times e}{C_{bio}} \times (\tau - t) + 1\right)} \qquad \text{Eq. 3}$$

3. Results and discussion

3.1 Aerobic biodegradability and kinetics prediction

In order to investigate the correlations between the characterization and cropped soil incubation data, a Partial Least Square (PLS) regression was applied on using 33 explicative variables composed of the 5 fractions SPOM, REOM, SEOM, PEOM, NEOM (percentage of extracted organic matter) and fluorescence spectroscopy data (using fluorescence percentage of each zone of each fraction). A first calibration model was set-up for 20 observations and 8 samples were used as validation[1].

As shown by the Figure 3, the PLS model correlated well the potential carbon mineralization with the characterization of organic matter and its carbon accessibility in the different fractions. Consequently, two models were set up for both Y-variables (*i.e.,* C_{bio} and µ). Three and four components were needed for respectively C_{bio} and µ. The regression coefficients were 0.88 and 0.92 for respectively C_{bio} and µ. Both models fitted well the experimental data. The Root Mean Square Errors (RMSE) were respectively 4.97 $gC.gC^{-1}$ and 0.14 $gC.gC^{-1}.d^{-1}$ for C_{bio} and µ. The Root Mean Square Errors of Prediction (RMSEP) were 11.5 $gC.gC^{-1}$ and 0.42 $gC.gC^{-1}.d^{-1}$ respectively for C_{bio} and µ.

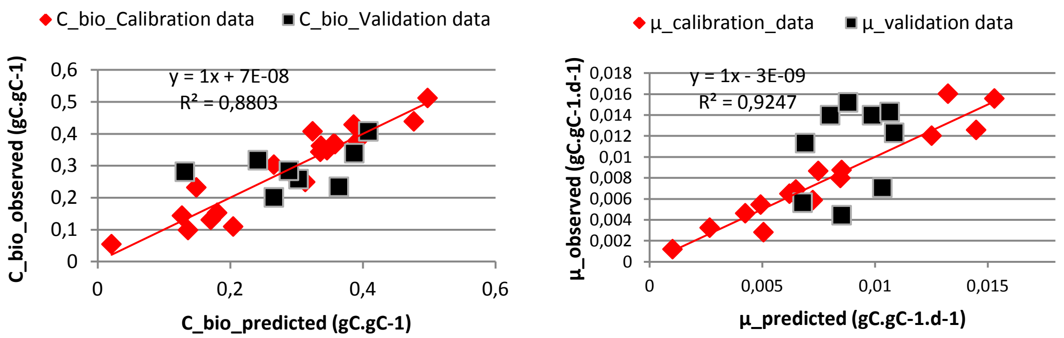

Figure 3: Observed versus predicted µ and C_{bio} obtained with the PLS models

Based on the correlation circles of scores and loading obtained for both models, Table 1 highlights the main impact of the most important X-variables on the Y-variables prediction. The scores and loadings of the PLS models highlighted that the first component represented the complexity axis and the second component represented the accessibility axis. For both models, the positive impact on the carbon mineralization is provided by the most accessible fraction SPOM and the simplest fluorescence zones represented by the protein-like in SPOM and REOM.

Table 1: Positive and negative impacts of the X-variables on Y-variables prediction

	X-variables Positive weight	X-variables Negative weight
C_{bio}	SPOM, Fluorescence zone from protein-like molecules of SPOM and REOM	NEOM, Fluorescence zones from complex molecules (humic-like and fulvic-like, lignocellulose-like) of SPOM and REOM
µ	SPOM, SEOM, Fluorescence zone from protein-like molecules of SPOM, REOM and SEOM	NEOM, PEOM, Fluorescence zones from complex molecules (humic-like and fulvic-like, lignocellulose-like) of SPOM, REOM and SEOM

In contrast, the less accessible fraction NEOM has a negative impact on the organic carbon mineralization. Besides, the most complex fluorescence zones of the 3D spectra of SPOM and REOM represented by humic-like and fulvic-like acids and lignocellulose-like molecules have also a high negative impact on carbon mineralization. These results are consistent: the most accessible and simplest molecules provide an increase of the biodegradability whereas the least accessible

[1] *The remained samples are still incubated with cropped soil*

and complex molecules provide a decrease of it. Moreover, concerning the kinetics prediction, the most accessible fractions SPOM and SEOM provide an increase of the rate whereas the least accessible fractions PEOM and NEOM provide a decrease of it.
This new methodology based on chemical extractions and 3D fluorescence spectroscopy highlights at the same time the complexity, the accessibility of organic matter and the humification process occurring during soil aerobic incubation.

3.2 Applications for such as diagnosis tool and dynamic modelling of organic waste treatment

Organic carbon stability and potential efficiency at increasing soil organic matter is usually estimated with the indicator of residual organic carbon (IROC) calculated as a function of soluble, cellulose- and lignin-like fractions provided by the Van Soest fractionation and a 3 days carbon mineralization test [1]. However, as mentioned by [1], this indicator was calibrated on samples which contain mainly fiber and carbohydrate, and is not suitable for protein-like samples such as digested sludge, sludge or manures as shown by [2] and [4]. Moreover, as previously mentioned, the humified fraction is missing in the Van Soest fractionation, in contrast to this new fractionation procedure. Indeed, thanks to the SEOM fraction and the 3D fluorescence spectroscopy, it is possible to identify and to include humification in the organic wastes characterization. Besides, as PLS models have shown, the SEOM fraction is a key variable of the mineralization kinetics prediction. Humification is a key process of the organic carbon stabilization that should explain more efficient capacity at increasing soil organic carbon stocks after spreading.
Contrary to the IROC indicator [1], this new methodology is able to better take into account the chemical nature of the extracted fractions and their evolution during treatment in the prediction of the potential efficiency at increasing soil organic matter after residue application. Further works are ongoing to complete the database of PLS models, and thus predicting the potential contribution to the soil organic carbon stocks.

4. Conclusion and outlook

These results present a first step towards improved quality control of organic residues (depending on the treatment processes and the quality of the incoming organic waste). Besides, this characterization methodology will be applied to a wider range of organic residues in order to enlarge the applicability domain of the usual IROC in soil [1] and to precise not only the size of the most influent fraction but also its composition. Moreover, the new fractionation method developed can be used as input variables of dynamic biological treatment models in order to optimize the whole treatment line according to both objectives, energetic and agronomic valorization of organic wastes. Only 5 days are needed to make this chemical characterization. However, the use of 3D fluorescence spectroscopy is a limitation to use it in practical terms. But the next step would be to normalized this protocol and propose it to analytical laboratories.

Acknowledgements

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References

[1] Lashermes, G., Nicolardot, B., Parnaudeau, V., Thuriès, L., Chaussod, R., Guillotin, M.L., Linères, M., Mary, B., Metzger, L., Morvan, T., Tricaud, A., Villette, C., Houot, S. (2009): Indicator of potential residual carbon in soils after exogenous organic matter application. European Journal of Soil Science, 60, 297–310.
[2] Peltre C., Dignac M.F., Derenne S., Houot S. (2010) : Change of the chemical composition and biodegradability of the Van Soest soluble fraction during composting: A study using a novel extraction method. Waste Management, 30, 12, 2448-2460.
[3] Peltre C., Christensen B.T., Dragon S., Icard C., Kätterer T., Houot S. (2012): RothC simulation of carbon accumulation in soil after repeated application of widely different organic amendments. Soil Biology and Biochemistry, 52, 49-60.
[4] Mottet, A., François, E., Latrille, E., Steyer, J.P., Déléris, S., Vedrenne, F., Carrère, H. (2010) : Estimating anaerobic biodegradability indicators for waste activated sludge. Chem. Eng. J. 160, 488–496
[5] Jimenez, J., Gonidec, E., Cacho Rivero, J.A., Latrille, E., Vedrenne, F., Steyer, J.-P. (2014) : Prediction of anaerobic biodegradability and bioaccessibility of municipal sludge by coupling sequential extractions with fluorescence spectroscopy: towards ADM1 variables characterization. Water Res. 50, 359–72
[6] Jimenez, J., Aemig, Q., Doussiet, N., Feurgard, I., Steyer, J.-P., Patureau, D., Houot, S. (2015): Towards a unified organic waste characterization of organic matter accessibility and complexity. Bioresource Technology 194, 344-353.

TB-O_04 Effect of food waste biochar usage in farmland on carbon sequestration and vegetable growth

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1. Introduction

For global warming prevention, carbon sequestration in the soil is effective with using biochar into the farmland. Carbon contained in the biochar carbonized with various biomass materials such as wood, bamboo, agricultural materials and food garbage can be stored in the soil for long period. Simultaneously, as the soil property is improved with using the biochar in the farmland, the vegetable growth promotion is also expected. The improvement of the soil property, water holding ability, water permeability and air permeability, comes partly from the addition of the biochar granule and powder in the soil, because there are a lot of micro voids and pores in and on the biochr granule and powder which bring the soil softness and the well-aerated environment of the soil. Symbiosis microorganisms, such as root nodule bacteria and mycorrhiza, in the soil also proliferate with addition of the biochar to the soil. It is also known that the soil aggregation is developed with microorganisms in the soil.

In this study, the microorganisms' proliferation is studied in farmland soil where the biochar and /or compost were used. Finally, the effect of the biochar addition in the farmland soilon the spinach growth was studied.

2. Materials and Method

2.1 Preparation of biochar

Figure 1 shows flow chart of preparation process of the biochar from food waste as a raw material from about 50 supermarkets. Part of food waste was pressed in order to decrease the moisture content under 65 wt%. The biochar was prepared from the food waste which was carbonized with outside-heating type furnace in 550°C - 600°C for several hours as shown in Figure 2. Finally, the biochar was pulverized into the grain size of under 3 mm.

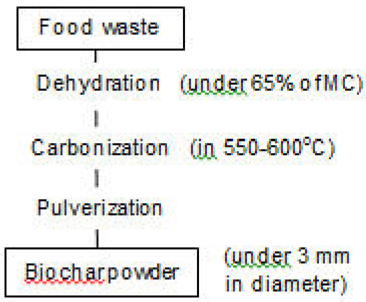

Figure 1: Flow chart of preparation of food waste biochar.

Figure 2: Bach-typed carbonizing furnace

2.2 Characterization of biochar

The ash content (%) of the food waste biochar was obtained by measuring the ash amount after burning at 815oC in the air [1]. The volatile matter (%) of was obtained by measuring the mass decrease after heating at 900oC for 7 min [1] in the air without oxygen. The fixed carbon (%) was estimated by deducting the ash content, the volatile matter and the water content from the total biochar wight [1].

The specific surface area of the biochar was measured with the BET method.

a. ATP concentration

As an index to the microorganisms amount in the soil, the adenosine tri-phosphate (ATP) concentration extracted from mitochondria in the cytoplasm of microorganisms was measured. The ATP concentration was measured with the Luminometer UPD-4000 (Meidensha Corp.,). When ATP, to which d-luciferin has been added, changes to adenosine monophosphate in the presence of luciferase and Mg^{2+}, light at a wavelength of 560 nm is emitted. Distilled water (20 ml) was added to 2 g of the sample and stirred with a tube mixer at 2500 rpm for 1 min. Then 250 µl of this suspension was withdrawn with a micropipette and an ATP measuring kit (Meidensha Corp., Lucifer AS) was added to it.

b. Vegetable growth

The biochar powder and the cow manure compost were added at the rate of 20 t/ha and 10 t/ha, respectively, to the red-clayey soil farmland. In comparison, the farmland compartments with the compost and without additives were used. The weight distribution of the leaves and roots of the spinach (*Spinacia oleracea*) was measured.

3. Results and discussion

3.1 Characterization of biochar

Table 1 shows ingredients of the pulverized biochar samples A, B and C, which were picked up in the different part of the biochar, because inhomogeneity is high in the food waste as a raw material. Some variability is recognized in measured data of the water content, the ash content and volatile matter. The ash content is higher than those of the biochar made from wood and bamboo. The average fixed carbon 68.0 wt% is finally estimated.
Table 2 shows some characteristics of the biochar. The specific surface area of 205 m^2/g, the bulk density of 0.2 g/ml and the pH value of 8.1.
The data in Tables 2 and 3 show that the carbonization of the food waste progresses to a sufficient extent.

Table 1: Ingredient amount % in the some biochar samples.

Ingredient	Amount (%)		
	Sample A	Sample B	Sample C
Water content	3.5	2.7	2.6
Ash content	3.6	4.2	4.0
Volatile matter	29.5	24.4	21.5
Fixed carbon	63.4	68.7	71.9

Table 2: Some characteristics of ther biochar.

Characteristics	
Specific surface area	205 m^2/g
Bulk density	0.2 g/ml
pH	8.1
Grain size	under 3 mm

3.2 Soil microorganisms

Figure 3 shows time dependence of ATP concentration of the farmland soil. The ATP concentration increased drastically after using both the biochar and the compost, and about the half value of which was observed in the soil with the compost and without the biochar, and then the ATP concentration of both of the soils gradually decreased. There no change of the ATP concentration was observed in both of the soils with the biochar and without additives. It was suggested that this increase of the ATP concentration in the soil with both the biochar and the compost came from the accelerated proliferation of the microorganisms on the surface of the biochar. Tanaka et al. [2] and Yoshizawa et al. [3, 4] found that the proliferation of composting microorganisms was enhanced on and in bamboo charcoal as a medium to which rice bran had been added as a nutrient.

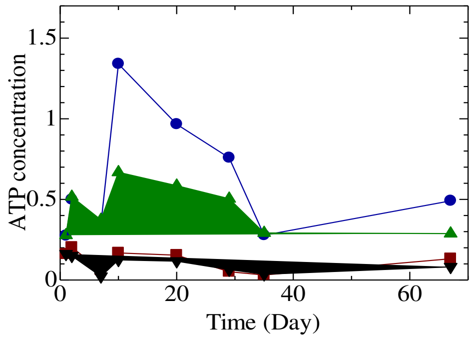

Figure 3: Time dependence of ATP concentration of the soil.
● : with charcoal and compost, ▲ : with compost, ▼ : with charcoal and ■ : no additives.

3.3 Vegetable growth

Figure 4 shows photographs of the appearance of spinach grown in the farmland. It was observed that the plants grew largely in the soil with both the charcoal and the compost rather than that in the soil with the compost. Small plants were cropped in the soil without additives, the charcoal and the compost.

Quantitative comparisons of the leaf and root fractions of spinach growth are shown in Figures 5 and 6, respectively. In the weight distribution of the leaf part, the spinach leaf grown in the soil with both the charcoal and the compose yielded 10% in weight ratio on 30 – 40 g and 40 – 50 g fractions, respectively, where is no existence of the spinach leaf grown in the soil with the compost and without additives. In the weight distribution of the root part, 2% and 13% of the spinach root were in 1.5 – 2 g and 1 – 1.5 g fractions, respectively. With the compost, 22% of the root was in 0.5 – 1 g fraction. There only exists the root without additives in the 0 – 0.5 g fraction.

It was suggested that the stimulation of the spinach growth in the soil used with both the charcoal and the compost came from the improvement of the physical and biological property of the soil [3].

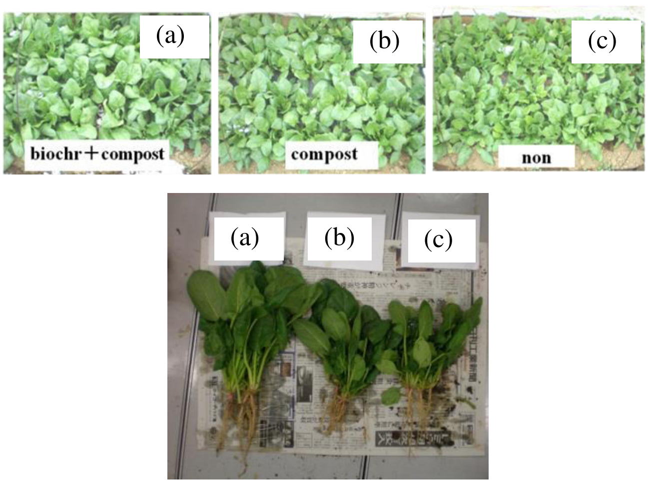

Figure 4: Effect of biochar and compost on spinach growth. (a) with biochar and compost, (b) with compost and (c) no additives.

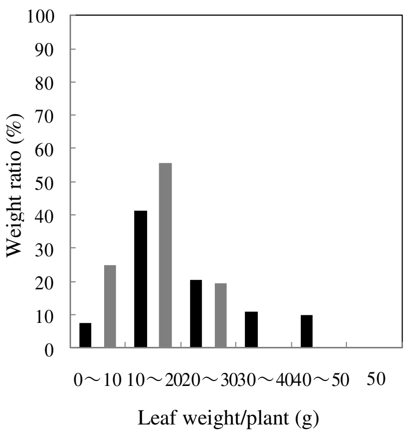

Figure 5: Weight distribution of the leaf part per plant.
■ : with biochar and compost,
▧ : with compost and ■ : no additives.

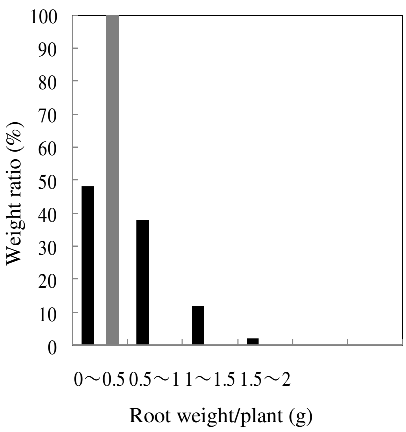

Figure 6: Weight distribution of the root part per plant.
■ : with biocharl and compost,
▧ : with compost and ■ : no additives.

4. Conclusion and outlook

The ATP concentration increased in the soil added with both the food garbage biochar and the cow manure compost. This suggests that the proliferation of the microorganisms was accelerated on the surface of the biochar.

Growth stimulation of the spinach in the farmland used with both the biochar and the compost was observed.

References

[1] Japanese Industrial Standards (JIC) M 8812

[2] Tanaka, S.; Ohata, M.; Yoshizawa, S.; Mineki, S.; Fujioka, K.; Kokubun, T. (2005): Proliferation of Microorganisms in Compost by Addition of Various Charcoals. Proc. Inter. Conf. on Carbon (Carbon 2005). Gyeongju, Korea, July 3-7. S04-06

[3] Tanaka, S.; Onozawa O.; Yoshizawa, S. (2011): Promotion Effect of Various Charcoals on the Proliferation of Composting Microorganisms. Extended Abstracts of Asia Pacific Biochar Conference (APBC 2011), Kyoto, Japan, Sept. 15-17

[4] Yoshizawa, S.; Tanaka, S.; Ohata, M.; Mineki, S.; Goto, S.; Fujioka, K.; Kokubun, T. (2006): Promotion Effect of Various Charcoals on the Proliferation of Composting Microorganisms. TANSO, no. 224, 261-265

TB-O_05 Simulation of carbon and nitrogen mineralization after application of ammonium sulphate, pig slurry and maize stalks to agricultural soil

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1. Objectives

This research aimed at improving the simulation of nitrogen (N) turnover when maize stalks (ST), ammonium sulphate (AS), and pig slurry (PS) are applied to the soil. In order to better quantify the fluxes of applied N, we conducted an incubation study involving ^{15}N labelled materials; results of the incubation experiments were then used to calibrate three different dynamic simulation models.

2. Methodology

2.1. Incubation experiment

A loam soil (pHW 7.9; Organic C 1.98%; Total N 0.17%) was added with labelled (15N) or unlabelled materials (Table 1) with the following treatments: 1) unfertilised soil (CON); 2) ST15N + ASUN + PSUN; 3) STUN + AS15N + PSUN and 4) STUN + ASUN + PS15N. Labelled ST were obtained from plants grown in sand and fertilized with labelled urea (≈10% AT% 15N). Labelled PS was a mix of faeces and urines collected from pigs feed with grain from the labelled maize plants.

Table 1: Characteristics of the materials used in the incubation experiment.

Material	^{15}N labelled	DM %	Org. C %	Tot. N %	NH_4-N/ Tot. N	AT% ^{15}N Tot. N
ST	No	94.00	40.55	0.76	n.d.	0.369
	Yes	94.00	30.16	0.67	n.d.	4.745
AS	No	0.00	0.00	0.21	1.00	0.367
	Yes	0.00	0.00	0.21	1.00	11.236
PS	No	2.87	1.13	0.18	0.57	0.385
	Yes	4.26	1.80	0.26	0.56	2.477

DM-dry matter; n.d.-not determined.

Twelve sets of experimental units have been prepared allowing 12 destructive measurements at the end of 12 incubation intervals. Each set consisted of 16 experimental units (four materials x four replications) arranged in a completely randomized design. Experimental units consisted of pre-incubated soil (equal to 100 g dry wt) amended with water (CON) or the three fertilizers (applied at a rate of 19, 35 and 48 mg N kg^{-1} soil for ST, AS and PS, respectively). Incubation was carried out in the dark at 25°C, and soil humidity was kept constant by periodical additions of distilled water to compensate for evaporation losses. During 180 days, we measured CO_2 emissions, microbial biomass C, N and ^{15}N, and soil NH_4-N, NH_4-^{15}N, NO_3-N and NO_3-^{15}N content. Soil mineral N and ^{15}N content (SMN and $SM^{15}N$) were calculated as the sum of NH_4-N and NO_3-N (SMN) or NH_4-^{15}N and NO_3-^{15}N ($SM^{15}N$).

2.2 Simulation models

The three models –2 modifications of ICBM 2B/N (Kätterer and Andrén, 2001) and CN-SIM (Petersen et al., 2005)– represent soil organic matter turnover through the interactions among native, added and microbial soil organic pools with a different degree of detail (Figure 1). In MOD2 and CN-SIM, ST and PS carbon is partitioned into a resistant (fC_{AOM1}) and a labile fraction (1-fC_{AOM1}); this happens also for N with the parameter fN_{AOM1}. MOD-1 uses only one pool to represent the addition of organic materials. Decomposition of organic pools is simulated with first order kinetics, and N availability regulates C decomposition. Models were calibrated using measured C, N and ^{15}N. Optimal parameters values were found using the downhill simplex (Nelder and Mead, 1965), minimising the relative root mean square error (RRMSE) between simulated and measured variables. In order to better ensure the achievement of the *true* optimum, this procedure was repeated using many (500) simplexes.

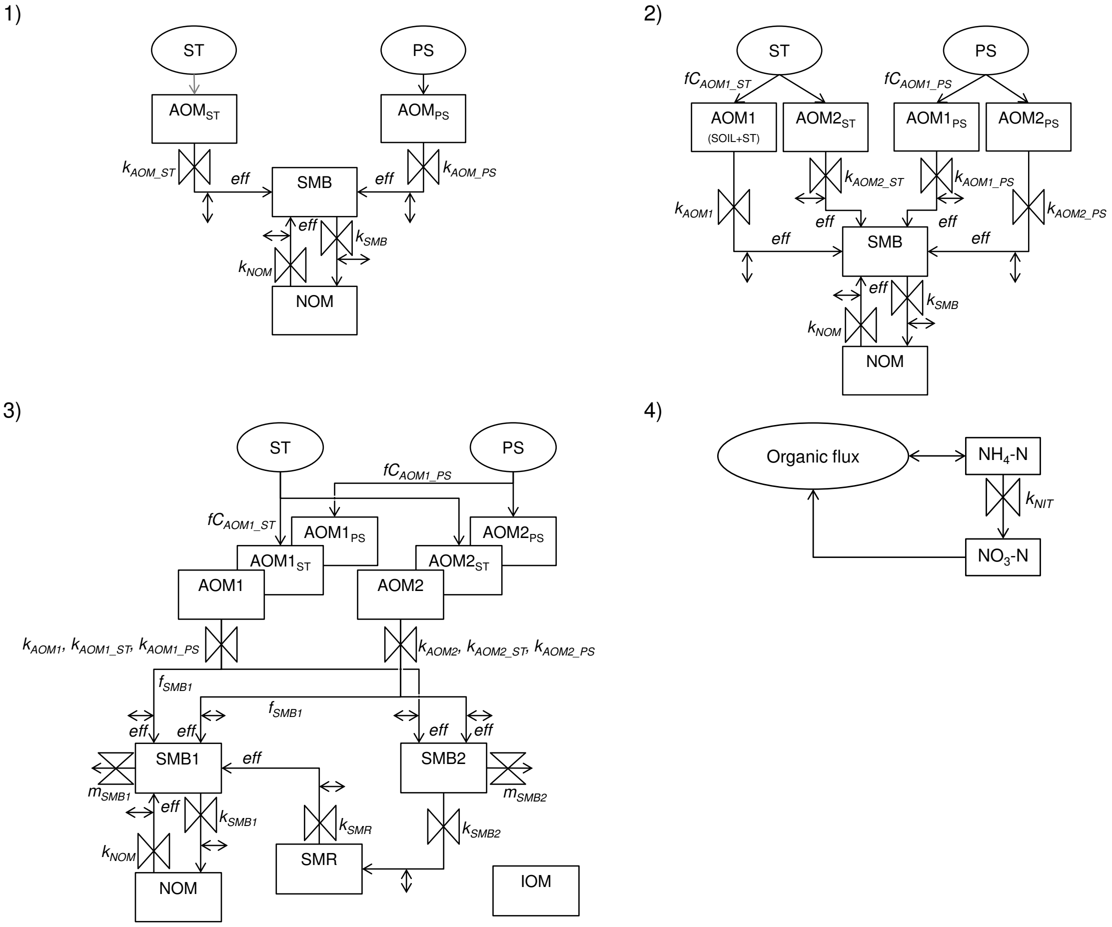

Figure 1: Relational diagrams of the three model used in the study. 1) MOD1, derived from Kätterer and Andrén, 2001; 2) MOD2, derived from Kätterer and Andrén, 2001; 3) CN-SIM (redrawn from Petersen et al., 2005); 4) scheme of N immobilisation, mineralization and nitrification, common to all models. Rectangles are state variables (organic pools), valves are rate variables, and continuous lines are flows of N. Flows of C and ^{15}N follows those of N. AOM: Added Organic Matter (AOM1, resistant; AOM2, labile); SMB: Soil Microbial Biomass (SMB1, 'Autochthonous'; SMB2, 'Zymogenous'); NOM: Native Organic Matter; SMR: Soil Microbial Residues; IOM: Inert Organic Matter.

3. Results and discussion

3.1. Carbon and nitrogen dynamics

In all amended treatments, CO_2 emission rates ranged from 6% to 1% of applied C at Day 1 and Day 15, respectively. In the subsequent weeks, respiration rates decreased and reached values close to those measured in CON. After two weeks, an average of 25% of applied C was mineralised in all amended treatments. Accumulated respired C at the end of incubation was on average half of the added C with ST+PS. Microbial biomass C increased until Day 11, when it represented from 7 to 20% of applied C, depending on the treatment; thereafter, microbial biomass C decreased, and reached, at the end of the incubation, values close to those measured at the beginning of the experiment (about 1.8% of soil C). In all fertilised treatments, net N immobilisation occurred immediately after materials addition. At Day 11, on average, microbial biomass N accounted for 24% of applied N. In the last days of incubation SMN increased, indicating that mineralization prevailed over N immobilisation; at Day 188, about 43% of applied N was recovered in SMN, while 12% was recovered in microbial biomass N (on average for the three fertilised treatments). Results of ^{15}N showed that the availability of applied N was in the order ST<PS<AS. At the end of incubation, 32%, 69% and 82% of applied ^{15}N was recovered in the $SM^{15}N$+microbial biomass ^{15}N in the labelled ST, PS and AS treatments, respectively.

3.2 Model simulation

Parameter calibration: The optimised values of the calibrated model parameters describing added materials are reported in Table 2. During calibration, some parameters reached the imposed boundaries, suggesting that simulation improvements could be achieved adopting larger calibration ranges; however, wider ranges would result in unrealistic parameter values, degrading model simulation to a simple data interpolation. This also highlighted difficulties of these models to represent the studied system. Results of MOD1 showed that ST decomposed faster than PS (kAOM); this was probably an artifact because, in this experiment, ST contributed more than PS to CO2 emissions and thus, the optimization algorithm tried to match CO2 measurements by adjusting kAOM of ST rather than kAOM of PS. In the two other models, there was a good agreement between the decomposition constants of the labile (kAOM2) and resistant (kAOM1) pools of ST and PS, with both the labile and recalcitrant pools of PS decomposing faster than ST. Parameter calibration resulted in a similar representation of ST in MOD2 and CN-SIM. The fraction of ST carbon allocated to the recalcitrant pool AOM1 (fCAOM1) ranged from 0.55 to 0.58, while the partitioning coefficient of N (fNAOM1) ranged from 0.51 to 0.69, with a good agreement between labelled and unlabeled ST. Differently from ST, the recalcitrant fraction of PS carbon (fCAOM1) was about three times higher for PS15 than PSUN, both in the MOD2 and CN-SIM models.

Table 2: Calibrated values of model parameters used for the partitioning of C and N added to the soil with ST and PS in AOM pools and decomposition constants of the AOM pools.

Parameter	Description	Units	Material	Optimised value		
				MOD1	MOD2	CN-SIM
fC_{AOM1}	Fraction of input C allocated to AOM_1 pool	mg C mg^{-1} C	ST^{UN}	—	0.55	0.56
			ST^{15}	—	0.58	0.56
			PS^{UN}	—	0.24	0.28
			PS^{15}	—	0.75	0.68
fN_{AOM1}	Fraction of input N allocated to AOM_1 pool	mg N mg^{-1} N	ST^{UN}	—	0.51	0.61
			ST^{15}	—	0.69	0.57
			PS^{UN}	—	0.64	0.80
			PS^{15}	—	0.90	0.80
k_{AOM}	Decomposition constant of AOM pool	day^{-1}	ST^{UN} and ST^{15}	0.0522	—	—
			PS^{UN} and PS^{15}	0.0027	—	—
k_{AOM1}	Decomposition constant of AOM_1 pool	day^{-1}	ST^{UN} and ST^{15}	—	0.0035	0.0036
			PS^{UN} and PS^{15}	—	0.0057	0.0299
k_{AOM2}	Decomposition constant of AOM_2 pool	day^{-1}	ST^{UN} and ST^{15}	—	0.17	0.16
			PS^{UN} and PS^{15}	—	0.89	0.70

Simulation errors: All the three models were able to simulate rather well CO_2 emissions and the dynamics of NH_4-N and NO_3-N, with average RRMSEs of 13%, 25% and 23% for the three variables, respectively (Table 3). On the contrary, simulation of the other variables was often unsatisfactory, with average RRMSEs of 68% (microbial biomass C), 117% (microbial biomass N), 65% (microbial biomass ^{15}N) and 47% (NO_3-^{15}N). This fact could be partially ascribed to a higher experimental variability associated with microbial biomass C and N measurements and to the lack of some processes not implemented in the models that make it difficult to simulate those dynamics. With the exception of few cases, higher errors were obtained with MOD1 compared to MOD2 and CN-SIM, and with MOD2 compared to CN-SIM. These results clearly highlighted the inaccuracy of MOD1 for the simulation of short-term C and N dynamics in amended soils. Improvements in the simulation with CN-SIM compared to MOD2 can be due to a more detailed representation of the soil organic matter turnover in the former model.

As an example, Figure 2 reports the simulation results of the ST^{UN}-AS^{UN}-PS^{15} treatment.

All models were able to reproduce higher respiration rates in the first weeks of incubation and slower rates thereafter, due to the exhaustion of the labile and part of the resistant ST and PS pools. However, calibration of MOD2 resulted in an underestimation of CO_2 emission after Day 11. Concurrently with C decomposition, MOD1 strongly overestimated N immobilisation until Day 40 and thereafter overestimated N mineralization, while the other two models were able to reproduce an initial phase of net N immobilisation and a subsequent phase of very slow net mineralization.

Table 3: Relative root mean squared errors between simulated and observed variables obtained for the three models after parameter calibration. All variables are net of CON and are expressed as percentage of applied C or N.

Variable	Treatment								
	ST^{15}-AS^{UN}-PS^{UN}			ST^{UN}-AS^{15}-PS^{UN}			ST^{UN}-AS^{UN}-PS^{15}		
	MOD1	**MOD2**	**CN-SIM**	**MOD1**	**MOD2**	**CN-SIM**	**MOD1**	**MOD2**	**CN-SIM**
CO_2-C	0.16	0.15	0.07	0.12	0.15	0.1	0.11	0.25	0.08
Microbial biomass C	0.93	0.38	0.19	0.63	0.23	0.36	1.66	0.93	0.85
NH_4-N	0.47	0.24	0.25	0.27	0.16	0.17	0.27	0.19	0.20
NO_3-N	0.23	0.23	0.23	0.27	0.21	0.15	0.30	0.25	0.21
Microbial biomass N	0.81	0.98	0.9	1.96	1.48	0.95	1.75	0.91	0.8
NO_3-^{15}N	1.04	0.69	0.64	0.47	0.34	0.32	0.27	0.23	0.23
Microbial biomass ^{15}N	1.78	0.71	0.64	0.44	0.37	0.46	0.53	0.44	0.49

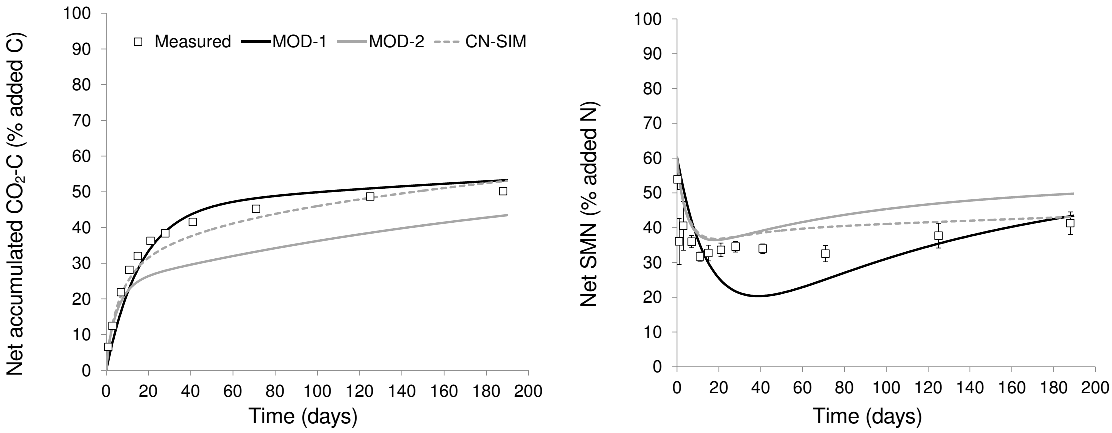

Figure 2: Net accumulated CO_2-C and SMN measured and simulated in the ST^{UN}-AS^{UN}-PS^{15} treatment.

4. Conclusion and outlook

These results confirm the importance of performing model calibration using both C and N measurements. In fact, simulation errors obtained in this work may have been much lower if we considered only C or N dynamics. However, optimizing models using both C and N measurements enables their use for a wider number of purposes. We suggest that further model modifications, such as variable substrate use efficiency and variable C to N ratios of microbial biomass, may give a better agreement between measured and simulated dynamics.

Acknowledgements

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References

[1] Kätterer, T.; Andrén, O. (2001). The ICBM family of analytically solved models of soil carbon, nitrogen and microbial biomass dynamics descriptions and application examples. Ecol. Model. 136, 191–207.
[2] Nelder, J.A., Mead, R. (1965). A simplex method for function minimization. Computer J. 7, 308-313.
[3] Petersen, B.M.; Jensen, L.S.; Hansen, S.; Pedersen, A.; Henriksen, T.M.; Sørensen, P.; Trinsoutrot-Gattin, I.; Berntsen, J. (2005). CN-SIM: a model for the turnover of soil organic matter. II. Short-term carbon and nitrogen development. Soil Biol. Biochem. 37, 375–393.

TB-O_06 Determining the mechanisms of nitrous oxide emission under contrasting soil disturbance levels and organic amendments

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1. Objectives

Soil tillage and fertilization practices can affect soil properties (e.g., pH, temperature, water saturation, nitrate and labile organic carbon contents) and consequently the abundance of nitrifying and denitrifying bacteria communities that are known to regulate N_2O efflux from soils [1-3]. The objective of this study was to determine mechanisms regulating soil N_2O emissions from a Nitisol under contrasting soil disturbance levels and N sources. This was achieved by correlating bacteria communities harboring specific catabolic nitrifying-ammonium monooxygenase (*amoA*), and denitrifying nitrate- (*narG*), nitrite- (*nirS*), nitric oxide- (*qnorB*) and nitrous oxide reductases (*nosZ*) genes with soil abiotic factors and N_2O emissions.

2. Methodology

Short-term N_2O emissions from a Rhodic Nitisol (0-10 cm soil layer characteristics: clay, silt and sand content of 250, 460, and 290 g kg^{-1}, pH-$H_2O_{(1:1)}$ 5.3, soil organic C content of 3.03 g kg^{-1}) under contrasting soil disturbance [undisturbed (US) and disturbed soil (DS)] and different N sources [140 kg N ha^{-1} as urea, raw swine slurry (RS), anaerobically digested swine slurry (ADS), composted swine slurry (CS) and in the absence of N (control)] were evaluated [4]. Organic fertilizers were obtained from a fattening swine farm (Table 1). The experiment was carried out for 64 days. Soil N_2O emissions were assessed using static chambers according to standard protocols [5]. Gas samples were analyzed by photoacoustic infrared spectroscopy (INNOVA 1412, Lumasense Technologies, Denmark)[6]. Soil samples were collected (0-0.1 m depth) and N_2O emissions correlated with abiotic factors [temperature, water-filled pore space (WFPS), dissolved organic carbon (DOC), ammonium (NH_4^+-N) and nitrate (NO_3^--N) contents] and specific catabolic genes involved in nitrification and denitrification bioprocesses. Real-time quantitative PCR (qPCR) was used to quantify bacteria communities harboring nitrifying-ammonium monooxygenase (*amoA*), and denitrifying nitrate- (*narG*), nitrite- (*nirS*), nitric oxide- (*qnorB*) and nitrous oxide reductases (*nosZ*) genes [7].

Table 1: Application rate and characteristics of the organic fertilizers used in this study.

Material	Characteristics								Application rate	TOC input	N input			
	DM	VS	TOC	TN	Org-N	NH_4-N	NO_3-N	C/N			TN	Org-N	NH_4-N	NO_3-N
	%	---------------------- kg m^{-3} ----------------------							m^3 ha^{-1}	---------------- kg ha^{-1} ----------------				
RS[1]	7.4	45.9	29.0	4.4	1.7	2.7	ND	6.6	31.7	919	140	54	86	ND
ADS[1]	6.5	38.4	17.7	5.2	2.6	2.6	ND	3.4	27.1	480	140	70	70	ND
		----------------------- g kg^{-1} ----------------------							Mg ha^{-1}					
CS[2]	29.1	ND	317.0	16.6	15.1	1.2	0.3	19.1	29.0	2,675	140	127	10	3

RS: raw swine slurry; ADS: anaerobically digested swine slurry; CS: composted swine slurry; WS: wheat straw.[1]Results are expressed on a fresh matter basis; [2]Results are expressed on a dry matter basis; ND: not determined; DM: dry matter; VS: volatile solids; TOC: total organic carbon; TN: total nitrogen; Org-N: organic nitrogen; NH_4-N: ammonium-nitrogen; NO_3-N: nitrate-nitrogen; C/N: total organic carbon/total nitrogen ratio.

3. Results and discussion

Negligible interaction between soil disturbance (Fig. 1a) and fertilization (Fig. 1b-c) on soil N_2O-N efflux were observed. N_2O-N emissions from soil varied from 1.5 to 320.0 g ha^{-1} day^{-1} and were markedly affected by soil WFPS (Fig. 1a). Lower N_2O-N emissions (< 100 g ha^{-1} day^{-1}) were registered during the first 19 days of the experiment during low rainfall periods and WFPS < 0.6 cm^3 cm^{-3}. Conversely, higher N_2O-N emissions coincided with WFPS > 0.6 cm^3 cm^{-3}. Daily N_2O-N emissions from US were consistently higher than DS during most of the evaluation period. Among the fertilization treatments, higher N_2O-N emissions were registered in the first 30 days after RS amendment. Insignificant variations in N_2O-N emissions from ADS, UR, CS and CTR treatments were observed during the experimental time frame.

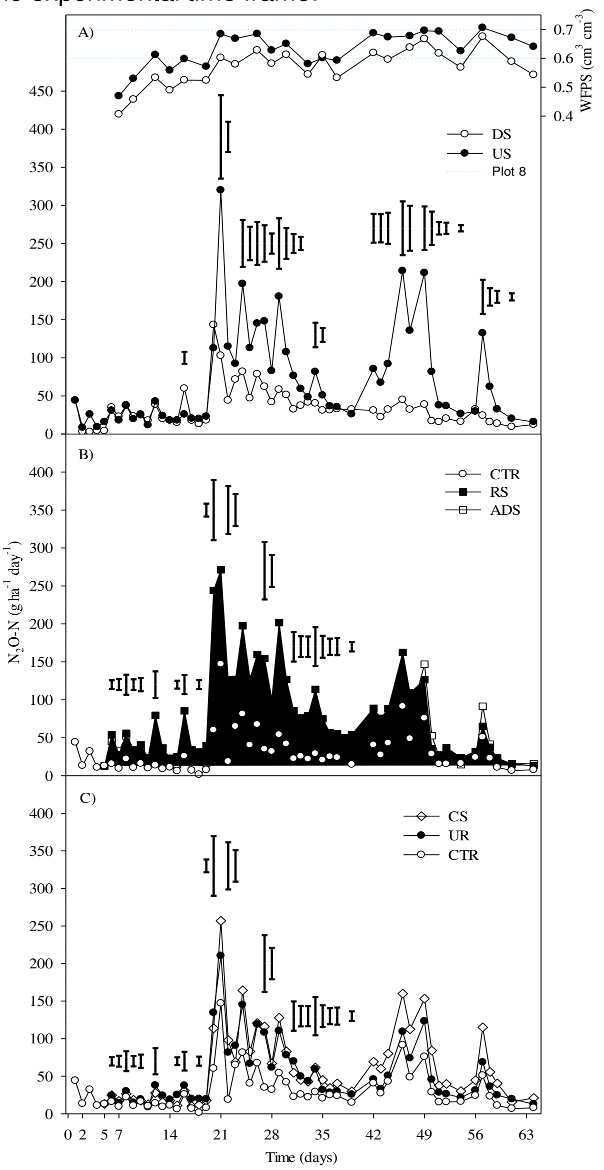

Figure 1: Soil water-filled pore space (A) and daily N_2O-N emissions according to soil disturbance (A) and fertilization treatments (B,C). DS: disturbed soil; US: undisturbed soil; CTR: negative control without N; UR: urea; RS: raw swine slurry; ADS: anaerobically digested swine slurry; CS: composted swine slurry. Bars denotes Fisher's test LSD ($p<0.05$).

N_2O emissions were remarkably affected by soil disturbance (US and DS)(Fig 1a). Possible interactions between soil abiotic and microbial communities that regulates N_2O-N emissions were investigated. The results were presented and discussed based on Pearson's correlation analyses (Table 2). For during most of the evaluated experimental period WFPS in the US was much higher than in the DS (Fig. 1a). N_2O emissions seemed to be mostly regulated by soil WFPS (r=0.753, $p<0.001$) in the US. This is in agreement with previous studies that reported an exponential increase of N_2O emissions for soils with WFPS > 0.6 cm^3 cm^{-3} [1,9]. Contrariwise, soil NO_3^--N and DOC contents (r=0.667, $p<0.01$ and -0.572, $p<0.01$) as well as DOC/NO_3^--N ratio (r=-0.459, $p<0.05$) showed better correlations with N_2O emissions from DS. Soil temperature and NH_4^+-N content had no significant correlations with N_2O emissions regardless of the treatments tested.
Nitrification was reported as the main process contributing to N_2O emissions at WFPS < 0.6 cm^3 cm^{-3} while denitrifying activity seems to be the dominant process regulating N_2O emissions when soil WFPS = 0.7 cm^3 cm^{-3} [1]. Pearson's correlation analyses revealed that denitrification was the main processes contribution to soil N_2O emissions in this study (Table 2). Significant correlations between *narG* (*narG/16S rDNA*) which is a gene encoding for NO_3^- reductase, and soil N_2O emissions were noticed for both US (r=0.528, $p<0.05$) and DS (r=0.505, $p<0.05$) treatments. The in-

creasing abundance of *nosZ* (*nosZ/16S rDNA*) that encodes for N_2O reductase had negative correlation with soil N_2O emissions in the US. Thus, it is plausible to assume that the higher N_2O emission from the US was associated with incomplete denitrification processes under high soil WFPS and NO_3^--N contents [3].

No correlation was observed between the fertilization treatments (CTR, UR and ADS) and soil abiotic or biotic factors. However, soil amended with RS showed N_2O emissions was significantly correlated with either soil WFPS (r=0.776, p<0.05) as well as *narG/16S rDNA* concentrations (r=0.789, p<0.05). RS application increased soil NH_4^+-N content which was promptly reduced to NO_3^--N (data not shown). Additionally, RS also provided a high labile C input that can enhance denitrification rates and soil N_2O emission [3]. Therefore, it is plausible to assume that incomplete denitrification played an important role on regulation of soil N_2O emissions from RS. In contrast, soil N_2O emissions from the CS treatment showed significant correlation with both *amoA/16S rDNA* (r=0.751, p<0.05) and *narG/16S rDNA* (r=0.870. p<0.01) genes. The *amoA* gene encodes for the reduction of NH_4^+-N during both autotrophic and heterotrophic nitrification, suggesting that either of these processes may have been occurring concomitantly and contributing for N_2O emissions. Whereas not previously demonstrated as a significant N_2O source in agricultural soils, heterotrophic nitrification (organic N oxidation) seems to be the main pathway for N_2O emission from forest soils with low pH and high organic matter content [1, 8,10]. Thus, the recalcitrant profile of the CS with high org-N content could have enhanced soil heterotrophic nitrification and N_2O emission when soil WFPS ranged from 0.6 to 0.7 $cm^3\ cm^{-3}$ in the US. The contribution of both heterotrophic nitrification and denitrification processes augmented N_2O emission from the CS amended soil, contrarily to what would be expected due to the low mineral N input by CS.

Table 2: Correlations between N_2O-N emissions, soil abiotic factors and the abundance of nitrifying and denitrifying catabolic genes according to Pearson´s coefficient.

Parameter	Soil disturbance[1]		Fertilization[2]				
	US	DS	CTR	UR	RS	ADS	CS
Soil temperature	-0.362	-0.055	-0.229	-0.227	-0.159	-0.341	-0.171
WFPS	0.753[a]	0.302	0.553	0.452	0.776[c]	0.552	0.637
NH_4^+-N	0.372	0.017	0.250	-0.137	0.255	-0.186	0.130
NO_3^--N	0.438	0.667[b]	0.420	0.371	0.567	0.398	0.409
DOC	0.253	-0.572[b]	0.280	0.276	-0.197	-0.634	-0.065
DOC / NO_3^--N	-0.103	-0.459[c]	-0.081	-0.178	-0.366	-0.103	-0.035
amoA/16S rDNA ratio	0.113	-0.059	0.341	-0.273	-0.104	0.336	0.751[c]
narG/16S rDNA ratio	0.528[c]	0.505[c]	0.143	0.671	0.789[c]	0.328	0.870[b]
nirS/16S rDNA ratio	-0.018	-0.132	0.258	0.049	-0.263	-0.170	0.420
qnorB/16S rDNA ratio	0.242	-0.057	0.371	0.082	0.511	-0.390	0.626
nosZ/16S rDNA ratio	-0.604[b]	-0.303	0.277	-0.640	-0.363	-0.438	-0.333
amoA/nosZ ratio	0.311	0.058	0.391	0.102	-0.009	0.507	0.770[c]
narG/nosZ ratio	0.730[a]	0.465[c]	0.609	0.823[c]	0.760[c]	0.518	0.816[c]
nirS/nosZ ratio	0.160	0.174	0.334	0.568	0.031	-0.021	0.803[c]
qnorB/nosZ ratio	0.619[b]	0.246	0.563	0.677	0.510	0.146	0.942[a]

a: p<0.001; b: p<0.01; c: p<0.05; CTR: control without fertilization; UR: urea; RS: raw swine slurry; ADS: anaerobically digested swine slurry; CS: composted swine slurry; [(1)]n=20; [(2)]n=8.

Principal component analysis (PCA) revealed that two ordination axes were able to explain up to 62% of the variability factors affecting soil N_2O emissions (Fig. 2). The PC1 explained 38.4% of the observed variability showing higher correlation with soil N_2O emissions (r=0.347), NO3--N (r=0.342), WFPS (r=0.346), and the ratio of amoA/nosZ (r=0.390), narG/nosZ (r=0.384), and qnorB/nosZ (r=0.431). The ratio of amoA/nosZ, narG/nosZ or qnorb/nosZ seems to better indicate soil N_2O emissions than the NO_3--N and WFPS as abiotic factors. Is worth noting that N_2O emissions are not exclusively dependant on nitrification/ denitrification metabolism only but also with the intrinsic interactions with soil NO_3--N and WFPS. The PC2 explained 23.5% of the variability and was correlated with DOC (r=0.369), NO_3--N (r=-0.348), DOC/NO_3--N ratio (r=0.521), nirS/nosZ ratio (r=0.336), WFPS (r=0.319) and temperature (r=-0.453). The PC2 was more influenced by changes in environmental conditions and soil abiotic factors regulating soil N_2O emissions. PCA analysis revealed that from day 7 to 21, the increasing abundance of nitrifying and denitrifying bacteria communities under high NO_3--N and WFPS favored soil N_2O emissions. From day 21 to 35, soil N_2O emissions were restrained by lower soil NO_3--N and WFPS. Following 35 days, the increasing DOC/NO_3--N ratios under high WFPS favored complete denitrification to N_2, thus diminishing N_2O emissions.

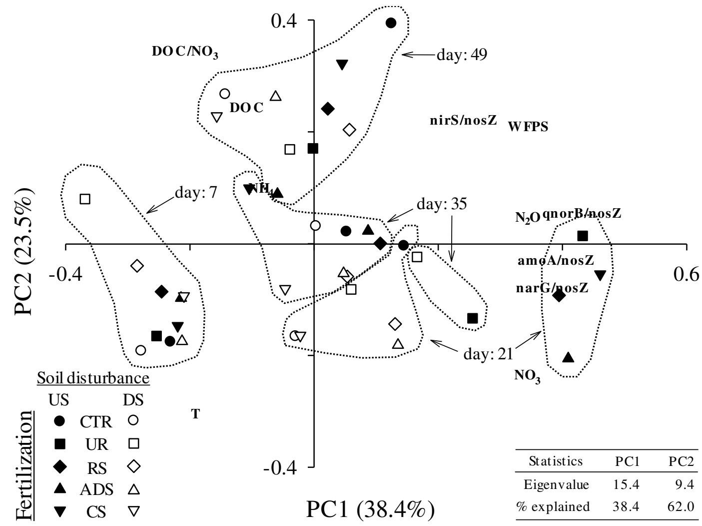

Figure 2: Principal component analysis (PCA) for N_2O-N emissions, soil abiotic factors (NH_4^+-N, NO_3^--N, DOC, DOC/ NO_3^--N, WFPS, and temperature) and nitrifying and denitrifying catabolic genes ratios (*amoA/nosZ, narG/nosZ*, *nirS/nosZ*, and *qnorB/nosZ*) according to soil disturbance and fertilization treatments. CTR: control without fertilization; UR: urea; RS: raw swine slurry; ADS: anaerobically digested swine slurry; CS: composted swine slurry.

4. Conclusion and outlook

Soil N_2O emissions are regulated by complex interactions between soil abiotic factors and abundance of nitrifying and denitrifying bacteria communities. Denitrification was the main bioprocess regulating soil N_2O emissions in both DS and US. The low soil WFPS encountered in the DS impaired N_2O emissions as compared to US. The application of a recalcitrant org-N source (CS) contributed to the proliferation of heterotrophic nitrifying bacteria communities and N_2O emissions in relation to other fertilizers with higher proportions of NH_4^+-N.

Acknowledgements

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References

[1] Bateman, E.J., Baggs, E.M. (2005): Contributions of nitrification and denitrification to N_2O emissions from soils at different water-filled pore space. Biol. Fertil. Soils 41:379-388.

[2] Bahl, K.B-., Baggs, E.M., Dannenmann, M., Kiese, R., Boltenstern, S.Z-. (2013): Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Phil. Trans. R. Soc. B 368: 20130122

[3] Giles, M., Morley, N., Baggs, E.M., Daniell., T.J. (2012): Soil nitrate reducing processes – drivers, mechanisms for spatial variation, and significance for nitrous oxide production. Front. Microbiol. 3:407

[4] Grave, R.A., Nicoloso, R.S., Cassol, P.C., Aita, C., Corrêa, J.C., Dalla Costa, M., Fritz, D.D. (2015): Short-term carbon dioxide emission under contrasting soil disturbance levels and organic amendments. Soil Till Res. 146:184-192.

[5] Rochette, P., Chadwick, D.R., de Klein, C.A.M., Cameron, K. (2012): Global Research Alliance on Agricultural Greenhouse Gases: Nitrous oxide chamber methodology guidelines, Version 1.0. de Klein, C.A.M., Harvey, M. (Eds.) Chapter 3 (Deployment protocol) (www.globalresearchalliance.org/research/livestock/activities/nitrous-oxide-chamber-methodology-guidelines/)

[6] Nicoloso, R.S., Bayer, C., Denega, G.L., Oliveira, P.A.V., Higarashi, M.M., Correa, J.C., Lopes, L.S. (2013): Gas chromatography and photoacoustic spectroscopy for the assessment of soil greenhouse gases emissions. Ci. Rural 43:262-269.

[7] Angnes, G.; Nicoloso, R.S.; Silva, M.L.B.; Oliveira, P.A.V.; Higarashi, M.M.; Mezzari, M.P.; Miller, P.R.M. (2013): Correlating denitrifying catabolic genes with N_2O and N_2 emissions from swine slurry composting. Bioresource Technology 14, 368-375.

[8] Zhang, J., Cai, Z., Zhu, T. (2011): N_2O production pathways in the subtropical acid forest soils in China. Environ. Res. 111:643-649.

[9] Dobbie, K.E., Smith, K.A. (2003): Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling variables. Glob. Change Biol. 9:204-218.

[10] Zhang, J.; Müller, C.; Cai, Z. (2015): Heterotrophic nitrification of organic N and its contribution to nitrous oxide emissions in soils. Soil. Biol. Biochem., 84:199-209.

TB-O_07 Evolution of zinc concentration in soil in *Pinus radiata D.* – Don silvopastoral systems limed and fertilized with sewage sludge

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1. Objectives

The objective of this study was to evaluate during 14 years the effect of three doses of sewage sludge (160, 320, and 480 kg total N ha^{-1}) combined with lime (2.5 t $CaCO_3$ ha^{-1}) or without lime on the total Zn concentration in the soil compared to control treatments (no fertilisation and mineral fertilisation) in a silvopastoral system established with *Pinus radiata* D. Don in an acid soil of Galicia (NW Spain).

2. Methodology

The experiment was conducted in Pol (Lugo, Galicia, northwestern Spain, European Atlantic Biogeographic Region) at an altitude of 748 m above sea level. The experiment was established in 1997 and used a five-year old *Pinus radiata* D. Don planted in 1993 with a density of 1667 trees ha^{-1}. The experimental design was a randomized complete block with three replicates. In autumn of 1997, the soil was cleared and ploughed, and the experimental plots were established. Each plot had a square of 5×5 trees and occupied 96 m^2, and plots were sown in autumn of 1997 with a mixture of 25 kg ha^{-1} of *Lolium perenne* var. Brigantia, 10 kg ha^{-1} of *Dactylis glomerata* var. Artabro and 4 kg ha^{-1} of *Trifolium repens* cv. Huia after ploughing. All cell plots were initially fertilised with 120 kg P_2O_5 ha^{-1} and 200 kg K_2O ha^{-1} in autumn 1997 to initially improve pasture establishment. The established nine treatments were no fertilisation (NF) and three sewage sludge doses based on N application (S1: 160 kg total N ha^{-1}; S2: 320 kg total N ha^{-1}; and S3: 480 kg total N ha^{-1}), with or without liming applied in 1997 before sowing (2.5 t $CaCO_3$ ha^{-1}). A no fertilisation (NF) treatment was also established as a control in the limed and unlimed plots. A control mineral treatment (MIN) in the unlimed plots was also included because the combination of lime and the MIN treatment is not usually applied in the area. The MIN treatment consisted of the application of 500 kg ha^{-1} of 8% N – 24% P_2O_5 – 16% K_2O ha^{-1} in accordance with conventional practice for fertilising pastures from 1998 to 2003. Sewage sludge was applied in 1998, 1999 and 2000. To evaluate the residual effect of these treatments, mineral fertiliser was added in 2001, 2002 and 2003 in the plots previously fertilised with sewage sludge, initially because in the higher doses the sludge was not easily incorporated (some unincorporated sewage sludge rests were visually visible) and later to improve pasture production.

The anaerobically digested sludge came from a municipal waste treatment plant in Lugo. Following the U.S. Environmental Protection Agency (EPA) recommendations, the doses were based on the percentage of total N (35.5 g kg^{-1}) and the dry matter content (24%) of the sewage sludge. The EPA established that approximately 25% of the total applied N is mineralised during the first year when sewage sludge is anaerobically digested [1]. The EU Directive 86/278/CEE [2] and the Spanish regulation R.D. 1310/1990 [3] regarding heavy metal concentrations in the application of sewage sludge to soil were also considered. The Zn concentrations in the sewage sludge used in the fertilisation in 1998, 1999 and 2000 were 821, 446 and 1320 mg kg^{-1}, respectively. These values were lower than the limit established by the legislation (2500 mg kg^{-1}) when the sewage sludge is used as fertiliser in acid soils. Therefore, the quality of sewage sludge used as fertiliser in this study could be considered good.

A composite soil sample per plot was randomly taken at a depth of 25 cm every December from 1998 to 2012 as described in R.D. 1310/1990 [3]. In the laboratory, the soil samples were air-dried, passed through a 2 mm sieve and ground with an agate mortar. The total Zn concentration in the soil was analysed with the VARIAN 220FS spectrophotometer using atomic absorption [4] after a nitric acid digestion made in a CEM MDS-2000 microwave [5].

Data were analysed with repeated measures ANOVA (proc glm procedure), and Mauchly's criterion was used to test for sphericity. If sphericity assumption was met then univariate approach output was used, otherwise multivaritae output (Wilks' Lambda test was taken into account). The statistical model used was $Y_{ij} = \mu + A_i + T_j + TA_{ji} + \varepsilon_{ij}$, where Y_{ij} is the dependent variable, μ is the mean of the variable, A_i is the year i, T_j is the treatment j, TA_{ji} is the year-treatment interaction (year*treatment) and ε_{ij} is the error. The LSD test was used for subsequent pairwise comparisons ($p < 0.05$; a = 0.05) if the ANOVA was significant. The statistical software package SAS (2001) was used for all analyses [6].

3. Results and discussion

The concentration of total Zn in the soil of this study (12.45-32.44 mg kg^{-1}) (Figure 1) can be considered low compared with the intervals found by several authors, such as Barber [7] (10-300 mg kg^{-1}) and Kabata-Pendias and Pendias [8] (10-105 mg kg^{-1}). Moreover, all values were very low compared to the Spanish regulation limits (150 mg kg^{-1}) (R.D.1310/1990) [3]. This may be explained by the fact that this study was located in an area without nearby pollution sources, the low soil pH increasing Zn availability, the high amount of precipitation in the area and that the soil had initial low levels of this element (17.2 mg Zn kg^{-1}).

In this study, it was observed a significant effect of the treatment ($p<0.001$), the year ($p<0.001$) and the interaction year*treatment ($p<0.05$) on the concentration of total Zn in the soil. In 1999, the concentration of this cation increased compared with 1998 probably due to the fertilisation with sewage sludge because Zn is the most abundant heavy metal in sewage sludge [9]. From 2000, the concentration of total Zn in the soil gradually decreased, until 2008, when we found the lowest concentration of this element in the soil. The reduction of levels of total Zn in the soil over time could be explained by the increase of cation extractions performed by *Pinus radiata* D. Don and pasture but also by the Zn leaching through the sandy soil profile. Similar results were also observed in a silvopastoral system established with *Fraxinus excelsior* L. in a sandy soil as the soil of our study [10] in which it was shown the important roles that clay minerals have in the immobilisation of heavy metals through their highly specific surface [11]. However, from 2009 to 2012, in general, the concentration of total Zn in the soil increased again compared with 2008, being the concentration of this cation in 2012 similar to that found in 1999 after the fertilisation. This result indicates a high residual effect of the sewage sludge and the need for long-term studies in soils fertilised with sewage sludge to avoid an environmental risk. The increase of total Zn concentration in the soil in the last years of the study could be due to the application of mineral fertiliser in all plots previously fertilised with sewage sludge which may have favoured the incorporation of patches of sewage sludge in the soil observed in 2001 [12] and, therefore, the Zn release to the soil. Many other studies have also shown an increase in Zn concentration in soil as a result of sewage sludge applications [10] [13]. Moreover, in the last years of the experiment, the treetop was less dense due to the fall of needles to the soil which probably allowed the input of more light to the understory thus favouring the incorporation of sewage sludge to the soil due to the increase in soil temperature and therefore microbial activity.

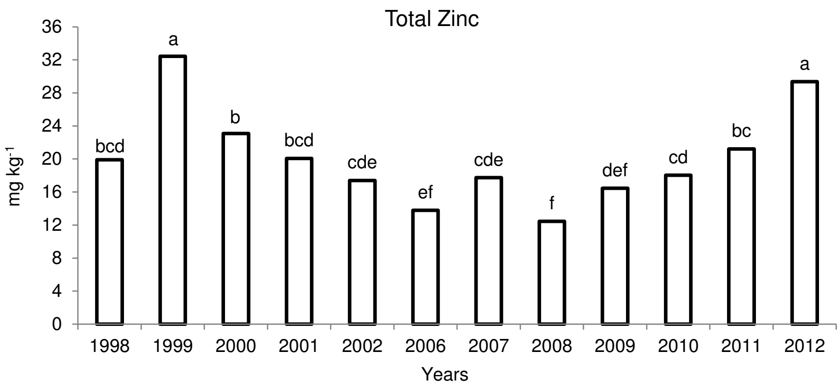

Figure 1: Mean concentration of total Zn in the soil (mg kg^{-1}) from 1998 to 2012. Different letters indicate significant differ ences between years.

On the other hand, Figure 2 shows the mean concentration of total Zn in the soil under each treatment in 1998, 2002, 2009 and 2012. The results found in the other years studied were similar to obtained in these years and therefore they are not shown. In 1998 and 2002, it was observed a positive effect of lime on the total Zn concentration in the soil when medium doses of sewage sludge were applied (S2). This result could be attributed to the increase in the exchangeable Ca [14], which tends to increase the incorporation and the mineralisation of the sewage sludge to the soil, causing the release of nutrients such as zinc. The positive effect of lime on the incorporation of heavy metals to the soil from fertilisation with sewage sludge was also observed in other silvopastoral systems established in the area with *Pinus radiata* D. Don [14]. However, in the last years of the study, the total Zn concentration was not modified by the input of lime to the soil probably due to lime inputs is characterised because its effect gradually diminishes in the years following its application.

Regarding to the effect of the doses of sewage sludge on the concentration of total Zn in the soil, in the limed plots, the medium doses of sewage sludge (S2) increased more the concentration of this cation in the soil than the low doses (S1) and the no fertilisation treatment (NF) in 1998 and 2002. In 2009, the levels of total Zn were higher when the sewage sludge was applied independently of the doses (S1, S2 and S3) compared with the no fertilisation treatment (NF). In 2012 the high doses of sewage sludge (S3) increased more the concentration of total Zn in the soil than the low doses (S1) and the NF treatment. In unlimed plots, it was also observed a higher concentration of total zinc in the soil when the high doses of sewage (S3) were applied compared with S1 and NF in 2002 and 2009 and compared with S1, S2 and NF in 2012. The improvement of the levels of total Zn in the soil associated to the highest doses of sewage sludge could be explained by the fact that the amount of Zn applied to the soil during 1998, 1999 and 2000 with the high (38.4 kg Zn ha^{-1}) doses of sewage sludge was higher than with the low (12.8 kg Zn ha^{-1}) and medium (25.6 kg Zn ha^{-1}) doses. Finally, in general the lowest concentration of total Zn in the soil of this study was associated to the control treatments (No Lime NF and MIN) compared with the treatments in which the high doses of sewage sludge (S3) were combined with lime or without lime probably because the control treatments did not imply an input of Zn to the soil. Several authors as Egiarte et al. [15] in northwestern Spain or Bettiol and Ghini [16] in Brazilian tropical soils have also found in their studies that the increase of the total Zn concentration in the soil was directly proportional to sewage sludge doses applied.

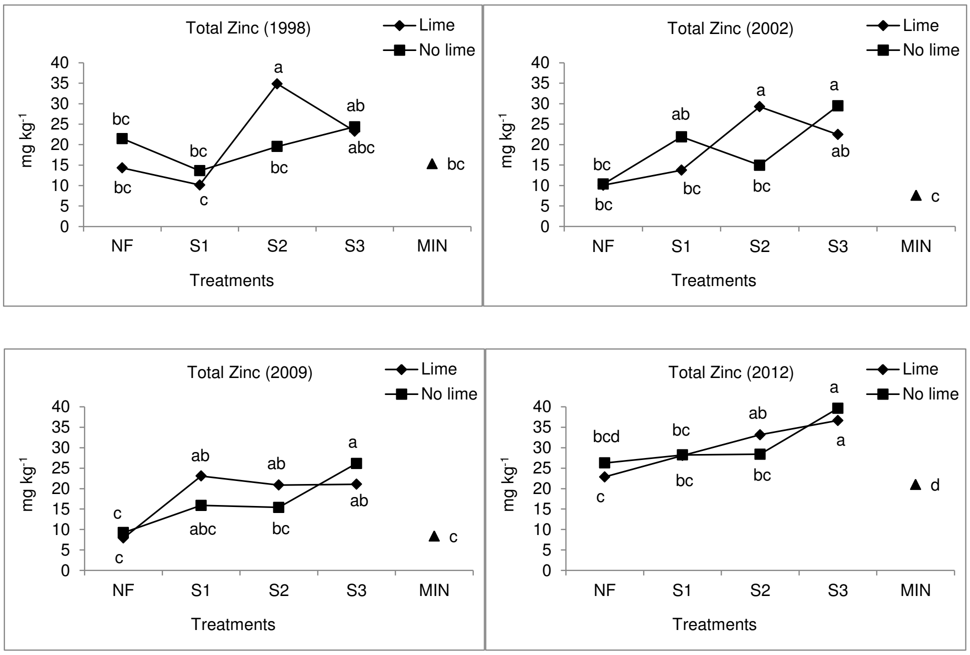

Figure 2: Mean concentration of total Zn in the soil (mg kg^{-1}) under each treatment in 1998, 2002, 2009 and 2012. NF: no fertiliser; S1: low sewage sludge dose (160 kg total N ha^{-1}); S2: medium sewage sludge dose (320 kg total N ha^{-1}); S3: high sewage sludge dose (480 kg total N ha^{-1}) and MIN: mineral fertiliser. Different letters indicate significant differences between fertiliser treatments.

4. Conclusion and outlook

The results obtained showed a large residual effect of the fertilisation with sewage sludge on the total Zn concentration in the soil, mainly when the high doses of sewage sludge were applied. Therefore, in acid soils such as those in this study it is necessary to evaluate during long periods the effect of sewage sludge on the Zn concentration in the soil to avoid an environmental risk.

Acknowledgements

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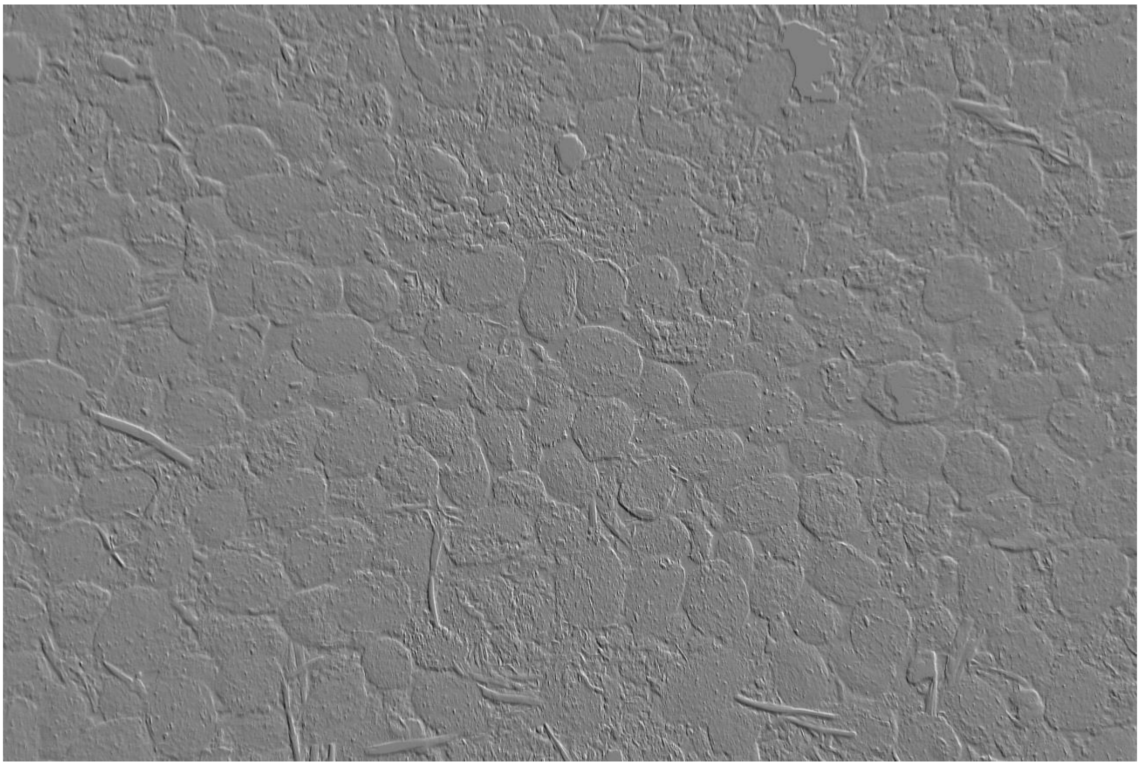
References

[1] EPA (Environmental Protection Agency) (1994): Land application of sewage sludge. A guide for land appliers on the requirements of the federal standards for the use of disposal of sewage sludge, 40 CFR Part 503. Environment Protection Agency, Washington, USA.
[2] EU (European Union) (1986): DOCE n_ L 181 04/07/1986.Council Directive 86/278/EEC of 12 June 1986 on the Protection of the Environment and, in Particular of the Soil, when Sewage Sludge is Used in Agriculture. http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri¼CELEX:31986L0278:ES. HTML (verified 10 May 2015).
[3] BOE (Spanish Official Bulletin) (1990): Real Decreto 1310/1990 29 de Octubre de 1990, que regula la utilización de los lodos de depuración (Royal Decree 1310/1990 29th October 1990, that regulates the use of sewage sludge). Ministerio Agricultura, Pesca y Alimentación, Madrid, Spain. http://www.boe.es/boe/dias/1990/11/01/pdfs/A32339-32340.pdf (verified 10 May 2015).
[4] VARIAN (1989): Analytical methods. VARIAN, Mulgrave, Australia.
[5] CEM (1994): Microwave digestion applications manual. CED, Matthews, NC, San Francisco, USA.
[6] SAS (2001): SAS/Stat User`s Guide: Statistics. SAS Institute Inc, Cary, NC, USA.
[7] Barber, A.C. (1995): Soil nutrient bioavailability. A mechanistic approach. John Wiley and Sons, Inc., New York, USA.
[8] Kabata-Pendias, A.; Pendias, H. (2001): Trace elements in soils and plants. CRC Press, Florida, USA.
[9] Mosquera-Losada, M.R.; Muñoz-Ferreiro, N.; Rigueiro-Rodríguez, A. (2010): Agronomic characterization of different types of sewage sludge: policy implications, Waste Management 30, 492–503.
[10] Rigueiro-Rodríguez, A.; Ferreiro-Domínguez, N.; Mosquera-Losada, M.R. (2010): The effects of fertilisation with anaerobic, composted and pelletized sewage sludge on soil, tree growth, pasture production and biodiversity in a silvopastoral system under ash (*Fraxinus excelsior* L.), Grass and Forage Science 65, 248–259.
[11] Zubillaga, M.S.; Bressan, E.; Lavado, R.S. (2008): Heavy metal mobility in polluted soils: effect of different treatments, American Journal of Environment Sciences 4, 620–624.
[12] López-Díaz, M.L.; Mosquera-Losada, M.R.; Rigueiro-Rodriguez, A. (2007): Lime, sewage sludge and mineral fertilisation in a silvopastoral system develop in very acid soils, Agroforestry Systems 70, 91–101.
[13] Rigueiro-Rodríguez, A.; Mosquera-Losada, M.R.; Ferreiro-Domínguez, N. (2012): Pasture and soil zinc evolution in forest and agriculture soils of Northwest Spain three years after fertilisation with sewage sludge, Agriculture, Ecosystems and Environment 150, 111–120.
[14] Rigueiro-Rodríguez, A.; López-Díaz, M.L.; Mosquera-Losada, M.R. (2011): Organic matter and chromium evolution in herbage and soil in a *Pinus radiata* silvopastoral system in northwest Spain after sewage sludge and lime application, Communications in Soil Science and Plant Analysis 42, 1551–1564.
[15] Egiarte, G.; Pinto, M.; Ruíz-Romera, E.;Camps Arbestain, M. (2009): Changes in heavy metal concentrations in acid soils under pine stands subjected to repeated applications of biosolids, Soil Science 174, 372-379.
[16] Bettiol, W.; Ghini, R. (2011): Impacts of sewage sludge in tropical soil: A case study in Brazil, Applied and Environmental Soil Science, doi:10.1155/2011/212807

Thematic Area TB – Sustainable soils
(Poster presentations)



Ina Körner: Animal excrements, processed photograph

TB-P_01 Fast estimation of the labile carbon fraction of organic waste products by FTIR photoacoustic spectroscopy

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1. Objectives

Soil incubation is the most commonly used laboratory method for determining the labile fraction of C in various materials [1]. However, this method is time consuming and expensive. Therefore, the main objective of this study was to test whether Fourier transform infrared photoacoustic spectroscopy (FTIR-PAS) can be used for predicting the mineralisable fraction of C in various organic waste products and identifying functional groups associated with it.

2. Methodology

The sample set consisted of 385 composted and non-composted organic waste products (OWP), more specifically: 65% urban or industrial waste (municipal solid waste composts, sewage sludge, agro-industrial wastes and biowastes), 21% animal manure and slurries (cattle, chicken, pig and horse manure) and 14% anaerobically digested wastes, plant materials and organic amendments. All samples were oven dried at 40˚C and finely ground (<1mm).

- Soil incubations: The mineralisable C fraction of OWP was determined by 91-day incubations of soil-OWP mixtures in hermetically sealed jars under controlled conditions of temperature (28 ˚C) and water content (75-100% of the soil water holding capacity). CO_2-C evolved was trapped in alkali traps (10 mL 0.5 M NaOH), which were replaced after 1, 3, 7, 14, 21, 28, 49, 70 and 91 days, and titrated by colorimetry. The total mineralised C was expressed as the fraction of total organic C (TOC) respired after 3 (Cm3d), 7 (Cm7d), 28 (Cm28d) and 91 days (Cm91d).
- FTIR-PAS – Prediction of C mineralisation: The FTIR-PAS spectra of the OWP were recorded in the mid-infrared region (4000-600 cm^{-1}) at a resolution of 4 cm^{-1} by co-adding 64 scans. Spectra were smoothed using the Savitzky-Golay algorithm on four points each side (a total window of nine smoothing points) and a zero order polynomial. Furthermore, the smoothed spectra were normalised by the mean. Partial least square regression (PLSR) analysis was then used to correlate the obtained OWP spectra with the measured mineralised fraction of C and develop calibration models predicting this fraction.

3. Results and discussion

3.1 Prediction of mineralisable C fraction

The PLSR analysis led to calibration models predicting an acceptable fraction of the variance in the mineralisable C fraction. The coefficient of determination (R^2) for this calibration sample set was 0.78, 0.78, 0.75 and 0.70 for the prediction of the total mineralisable C after 3, 7, 28 and 91 days respectively (Figure 1). The root mean square error (RMSE) was 2.15 (Cm3d), 3.92 (Cm7d), 5.94 (Cm28d) and 8.07 (Cm91d). Afterwards, a test set including randomly selected samples from all OWP types was adopted in order to validate the robustness of this model and its predictive power on unknown samples. Using the previously developed calibration model on the test set resulted in a prediction of the mineralisable fraction of C which was still good. More specifically, the coefficient of determination for total C mineralised after 3, 7, 28 and 91 days was 0.77, 0.75, 0.70 and 0.72, respectively, and the RMSE was 2.31 (Cm3d), 4.15 (Cm7d), 6.78 (Cm28d) and 7.71 (Cm91d). These results show that the model was performing similarly or better with the model developed on the same sample set using near infrared spectroscopy (NIRS) by Peltre et al. [2].

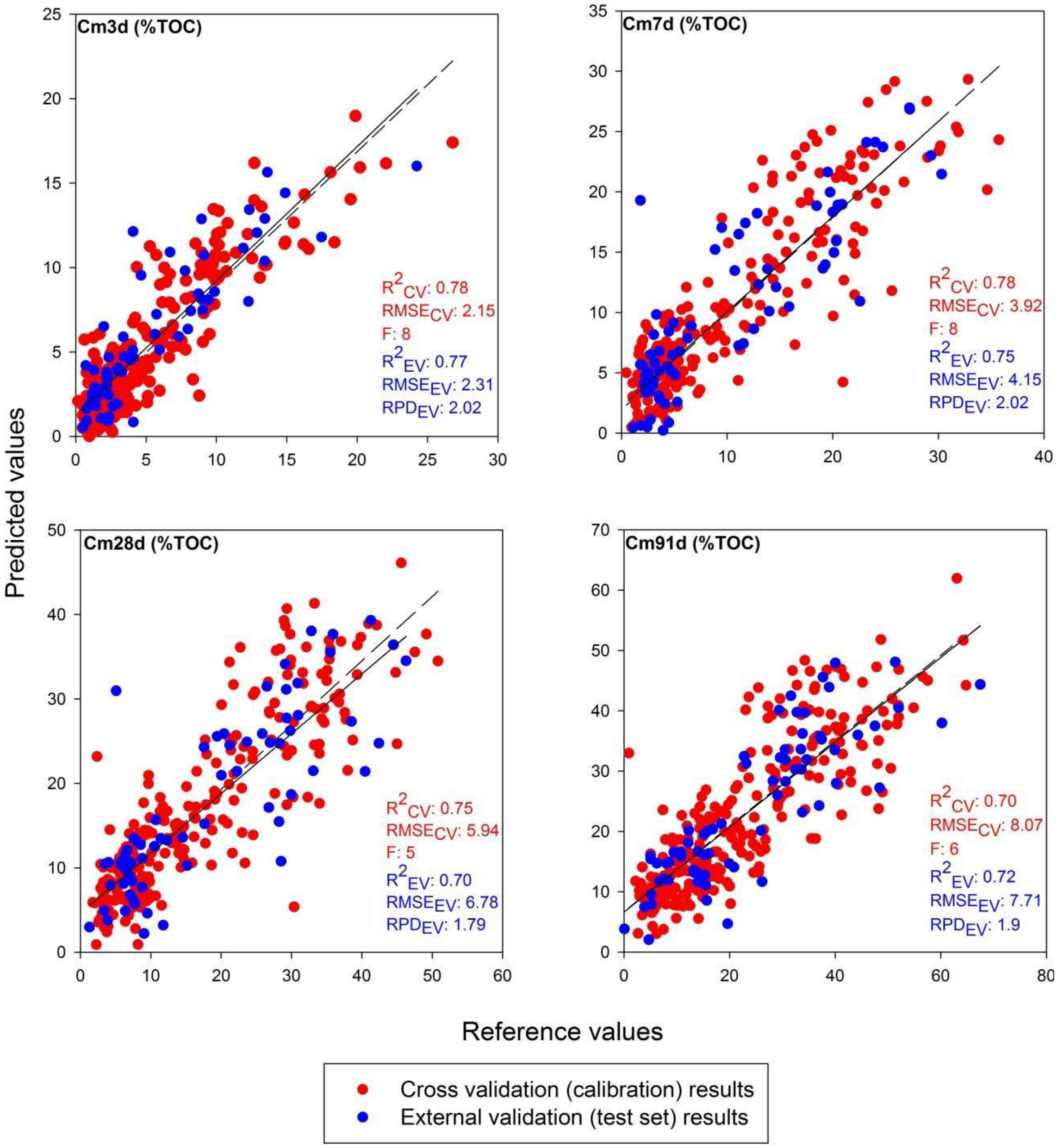

Figure 1: Correlation between reference (measured) and predicted percentage of total organic carbon (%TOC) that was mineralised after 3 (Cm3d), 7 (Cm7d), 28 (Cm28d) and 91 (Cm91d) days of incubation [cross validation results: red dots, dashed regression line, external validation results: blue dots, solid regression line) (R^2: r-square value, RMSE: root mean square value, CV: cross-validation (calibration) data set, EV: external validation (test) data set, RPD: ratio of performance to deviation, F: number of factors used in calibration]

3.2 Interpretation of regression coefficients

The regression (or beta) coefficients of the models predicting the total carbon fraction that mineralised after 3, 7, 28 and 91 days (i.e. the spectral regions used for this prediction) were similar with small differences between the coefficients. The interpretation of these coefficients allowed the identification of spectral regions that had a positive or negative association with the mineralisable fraction of C. A positive association (Figure 2) was observed with the spectral regions centered at 3700, 3300, 2950, 1740, 1660, 1438, 1400, 1160 and 876 cm^{-1}. The region between 3700 and 3000 cm^{-1} is characterised by overlapping vibrations of various compounds and moisture, so the interpretation of coefficients from this region is difficult. The positive regression coefficients at 2950, 2850 and 1400 cm^{-1} correspond to the C-H vibration in aliphatic compounds which are characterised as easily degradable compounds [3]. The coefficient at 1740, 1660 and 1160 cm^{-1}, are assigned to the C=O stretching vibration in ketones, C=O stretching vibration in carboxylates and C-O stretching vibration in polysaccharides respectively. All these compounds are known as easily degradable and therefore the positive regression coefficients for C mineralisation in these spectral regions were not surprising. The coefficients at 2520, 1435 and 875 cm^{-1} are corresponding to the CO_3^{2-} vibration in carbonates [4]. Many of the spectra of the OWP revealed high absorption intensity in these regions while at the same time they had the highest mineralisation values but not the highest total organic C content (data not shown). Therefore the positive coefficients in these spectral regions could result from covariance between the high absorption intensity at these regions and the high mineralisation values; however, alternatively they could also indicate that some of the CO_2 originated from the carbonates and released upon addition to soil which is typically more acidic than the waste materials. A negative association of the mineralisable C fraction with the spectral regions centered at 2300, 1509, 1325, 1270, 1220, 950 and 825 cm^{-1} was observed (Figure 2). The negative regression coefficient at 2300 cm^{-1} can be assigned to the –COH stretching vibration in polysaccharides and more specific in crystalline form of cellulose [5]. Cellulose is more resistant to enzymatic hydrolysis in the crystalline form and therefore its presence in the negative associated coefficients could be expected. The regression coefficient at 1510 cm^{-1} is assigned to the C=C stretching vibration in lignin [3]. Lignin is a recalcitrant compound that also inhibits the hydrolysis of cellulose. The regression coefficients at 1325 and 825 cm^{-1} correspond to the C-N and C-H vibrations in aromatic compounds, which form recalcitrant compounds, while the coefficients at 1270 and 1220 cm^{-1} are assigned to the C-N vibration in amide [3].

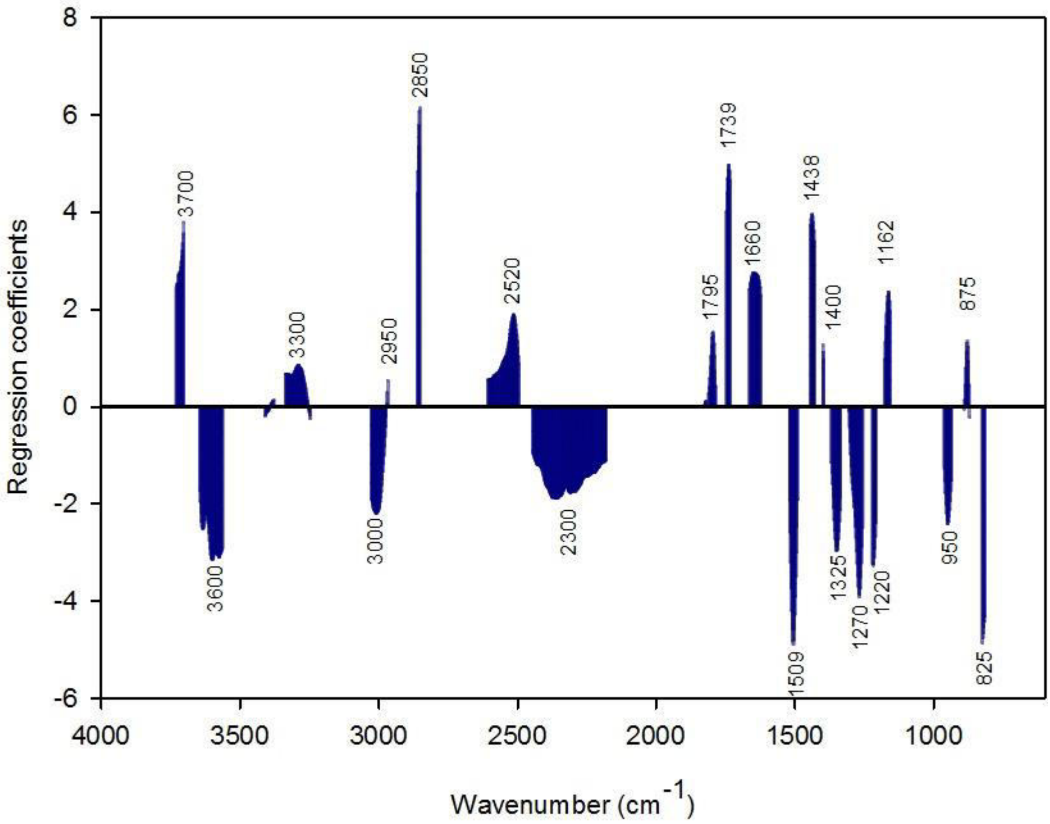

Figure 2: Regression coefficients used in the prediction of the mineralisable C fraction after 91 days.

4. Conclusions

The application of FTIR-PAS provided a fast and reliable technique for accurate prediction of the mineralisable fraction of C. In comparison with traditional FTIR, photoacoustic spectroscopy offered the advantage of directly measuring the absorbed IR radiation, which is advantageous when measuring dark and opaque samples such as these OWP. The coefficient of correlation between predicted and measured values of the mineralisable fraction of C ranged between 0.70 and 0.78, while the RMSE ranged between 2.31 and 7.71. This was similar to or better than earlier NIR predictions [2], but with the advantage that the spectral regions associated with labile carbon can be interpreted in terms of the chemical components responsible for the lability. The interpretation of these spectral regions revealed a positive correlation to aliphatic compounds, carboxylic acids, carboxylates and polysaccharides, and a negative correlation to more stable compounds such as lignin, crystalline cellulose and aromatic compounds. This produced a good evaluation of the robustness of the calibration model, which makes FTIR-PAS a powerful characterisation tool.

Acknowledgements

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References

[1] Khalil, M.I., Hossain, M.B. & Schmidhalter, U. (2005). Carbon and nitrogen mineralization in different upland soils of the subtropics treated with organic materials. *Soil Biology and Biochemistry*, *37*(8): 1507–1518.
[2] Peltre, C., Thuriès, L., Barthès, B., Brunet, D., Morvan, T., Nicolardot, B., Parnadeau, V. & Houot, S. (2011). Near infrared reflectance spectroscopy: A tool to characterize the composition of different types of exogenous organic matter and their behaviour in soil. *Soil Biology and Biochemistry*, *43*(1): 197–205.
[3] Smidt, E. & Schwanninger, M. (2005). Characterization of waste materials using FTIR spectroscopy: Process monitoring and quality assessment. *Spectroscopy Letters*, *38*(3): 247–270.
[4] Nguyen, T., Janik, L. & Raupach, M. (1991). Diffuse reflectance infrared Fourier transform (DRIFT) spectroscopy in soil studies. *Soil Research*, (1982): 49–67.
[5] Calderón, F., Haddix, M., Conant, R., Magrini-Bair, K. & Paul, E. (2013). Diffuse-reflectance fourier-transform mid-infrared spectroscopy as a method of characterizing changes in soil organic matter. *Soil Science Society of America Journal*, *77*(5), 1591.

TB-P_02 Soil amendment using fresh and stabilised organic materials – Effects during a wheat-maize cropping sequence

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1. Objectives

In this work, the use of a fresh organic amendment (sewage sludge) was compared to a stabilized organic material (compost from animal manures) during a wheat-maize cropping sequence with two main objectives:

- To study the effects on the soil physico-chemical, chemical and biological properties and on the yield of the two crops considered.
- To evaluate the influence of the different treatments on gaseous emissions of CO_2, N_2O and CH_4.

2. Methodology

The field experiment was conducted during one year, with two cropping cycles, at an agricultural area located at Orihuela (Alicante, 38°07'38.4"N 0°48'57.2"W). The climate in this area is semi-arid continental Mediterranean with an average cumulative annual precipitation of 300 mm and a mean annual temperature of 18 ºC. The soil at the site was a clay textured soil, moderately alkaline (pH = 8.3), with low salinity (0.40 dS/m) and low contents of oxidizable organic C (1.25%) and total Kjeldahl N (0.18%). The experimental field was divided into 6 experimental plots (4.5 m x 9.5 m), arranged in a randomized complete block design with two replicates per treatment. The treatments considered in this study were: control soil without amendment (control), soil amended with aerobic sewage sludge (SS) and soil amended with animal waste (goat and rabbit manure) derived compost (CS). CS was prepared using as raw materials a mixture 50:50 (v:v) of rabbit and goat manure, wheat straw and grass clippings in the proportion 85:8:7(on a fresh weight basis), respectively. SS and CS had pH values slightly alkaline (pH = 7.5 and 7.8, respectively); different dry matter contents (21% and 43%, respectively), similar electrical conductivity values (2.42 dS/m and 2.74 dS/m), notable organic matter contents, higher in SS (66.5% and 62.1%) and high total N contents, especially SS (6.03% and 3.10%). The treatments were established in January 2014, prior sowing, at a single application rate of 210 kg N/ha. No extra fertilisation during the crops was applied. The treatments wer applied manually and immediately incorporated to a soil depth of 20cm by light rototilling. The first crop (wheat, *Triticum aestivum* cv. Galera) was sown one week after the incorporation of the organic amendments. Five months later (at the end of May 2014), wheat was harvested and after one month (in July 2014), the following crop (maize, *Zea mays* cv. Pioneer P1758Y) was sown and harvested in December 2014. The effects of the treatments on soil physico-chemical, chemical and biological properties were evaluated throughout the experiment by collecting topsoil samples (0–25 cm) from all plots on four occasions: immediately after the incorporation of the organic amendments (M1) and at the end of the wheat cycle (M2), at the beginning of the maize cycle (M3) and after harvesting the maize crop (M4). The analytical methods used were the following:

- Method 1: The pH in the soil and organic amendments (SS and CS) samples was measured on 1:2.5 and 1:10 water soluble extracts (w/v), respectively. Electrical conductivity (EC) was determined on 1:5 and 1:10 water extracts of soil and organic amendment, respectively.
- Method 2: The oxidizable organic C (Corg) in soil samples was determined by the Walkley and Black modified method described by Bustamante et al. (2007); organic matter in SS and CS was assessed by determining the loss-on ignition at 430 °C for 24 h.
- Method 3: The total N in the soil samples was determined by the Kjeldahl method, while in SS and CS was determined by automatic microanalysis.
- Method 4: The soil respiration was determined by titration with 0.1 M HCl after trapping total CO_2 in 0.1 M NaOH, according to the method described by Bustamante et al. [1]. Briefly, 10 g aliquots of the collected soil samples were incubated in the dark for 5 days at 25 ºC in closed

500 ml glass vessels, trapping the CO_2 evolved with 10 ml of 0.1 M NaOH, after adding BaCl2, in small tubes placed inside the incubation vessels. Empty vessels were used as blanks.

Fluxes of CO_2, N_2O and CH_4 were measured throughout the experiment with manually operated circular static chambers (diameter 0.36 m and height 0.18 m), using three chambers per plot for gas sampling. Gas samples (20 ml) were taken by syringe at 0 and 60 minutes after the closure of the chamber to measure the increase or the decrease of concentration inside the chamber, and stored in chromatography vials. The linearity of gas diffusion into the headspace over this closure period (60 min) had previously been estimated and each flux could be calculated from a single determination at the end of closure by considering the chamber volume and soil surface [2]. Concentrations of GHGs in the gas samples were determined by gas chromatography, using an HP-6890 gas chromatograph (GC) equipped with a headspace autoanalyzer (HT3), both from Agilent Technologies. Also, the yields of each crop studied were determined after harvesting. All the analyses were made in triplicate and the data obtained were analyzed using a two-way analysis of variance (ANOVA) for repeated measures; to compare the differences obtained, the Tukey-b test was used ($P < 0.05$).

3. Results and discussion

3.1 Effect of the treatments on soil parameters

The incorporation of the organic amendments produced slight differences in the soil pH values and only during the first crop period (Table 1), probably due to the initial values of the organic materials used, closed to neutrality, and especially due to the buffering effect of this type of soil [1]. The electrical conductivity (EC) values were not clearly affected by the treatments, only showing a significant increase during the wheat period crop with the incorporation of SS, probably due to these organic amendments had not high values of EC (lower than 3 dS/m).

Table 1: Effect of the different treatments on soil physico-chemical, chemical and biological parameters during each crop period.

Treatment	pH	EC (dS/m)	Corg (%)	TKN (%)	Soil respiration (mg CO_2-C/kg day)
			Wheat crop		
Control	8.32ab	0.45a	1.61a	0.19a	23.5a
Sewage Sludge	8.24a	0.51b	1.75b	0.20b	30.9b
Compost	8.34b	0.45a	1.83c	0.22c	29.6b
			Maize crop		
Control	8.31a	0.58b	1.57a	0.18a	34.0b
Sewage Sludge	8.31a	0.48a	1.63a	0.18a	27.5a
Compost	8.35a	0.42a	1.75b	0.19b	26.3a

EC-electrical conductivity; Corg-oxidizable organic C; TKN- total Kjeldahl N.

Average values in a column followed by the same letter are not significantly different at $P < 0.05$ (Tukey-b test).

Regarding the concentrations of oxidizable organic C (Corg) and total Kjeldahl N (TKN), clear and significant differences were observed between the concentrations of these parameters related to soil fertility with the different treatments, especially during the development of wheat. The incorporation of the organic materials increased the contents of Corg compared to the soil without amendment (Control), but this increase only remained until the end of the wheat-maize cropping sequence in the soil amended with CS.

For the TKN, the effect was similar, observing the highest concentrations with the treatment CS, also maintaining only this treatment the effect at the end of the two-crop period. Khorramdel et al. [3] also observed during maize cropping a higher C sequestration (4,1 t/ha) in low input management systems using compost and manures in a dose of 30 t/ha and hand weeding, compared to other systems of medium or high management input, in which the dose of amendment is reduced and the presence of inorganic fertilizers and field practices (tillage, etc.) is increased (0.01 t/ha in the high input management system).

The application into soils of the organic amendments also enhanced the activity of the soil microbiota, increasing the rates of the soil respiration, especially during the first crop period. This reactivation of the activity of the soil microorganisms was also reported in other studies using fresh and stabilized organic materials as soil amendments [4].

3.2 Emissions of CO_2, N_2O and CH_4

The addition of SS and CS to the soils produced a clear effect on the cumulative fluxes of N_2O, CH_4 and CO_2 during the wheat-maize sequence (Figs.1a, b, c, d, e, f and g). These emissions were, in general, higher for all the gases during the first crop, increasing immediately after the incorporation of the organic materials. During the development of wheat, the soils amended with SS showed the highest cumulative emissions for all the gases studied.

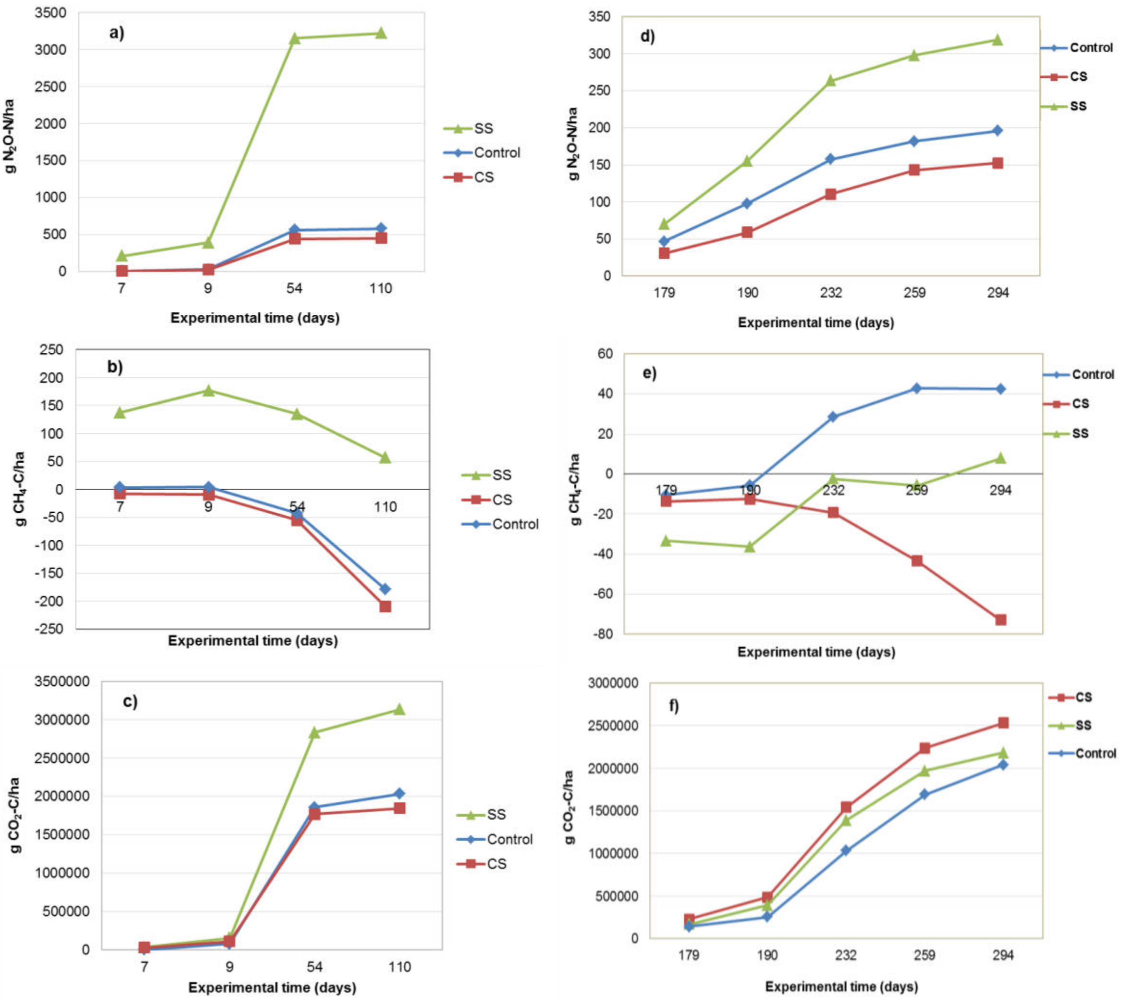

Figure 1: Cumulative emissions in the soil during the wheat crop of N_2O–N (a), CH_4–C (b) and CO_2–C (c) and during the maize crop of N_2O–N (d), CH_4–C (e) and CO_2–C (f). CS: animal waste derived-compost; SS: sewage sludge.

The incorporation of SS produced a notable increase of the N_2O-N emissions during the wheat period, compared to the effect observed in the soil amended with CS, being similar to that of the control soil (Fig. 1a). The addition of organic materials implies a supply of labile organic C, which accelerates the activity of soil microbes, thus increasing nitrification and denitrification rates [5]. All the soils displayed a similar trend during the maize period, although the soil amended with SS showed the highest N_2O-N emissions.

Increases in CO_2-C cumulative emissions were observed after approximately two weeks from the incorporation of the organic amendments in all the soils and especially in those amended with SS (Fig. 1c), reflecting the higher mineralization rate of this material compared to compost. Soil CO_2-C emissions are the result of a combination of heterotrophic and autotrophic respiration, and both could have been stimulated by the addition of organic materials [6]. The magnitude of these emissions was related to the decomposition rates (associated to the amount of labile C) of the materials used [7]. However, during the maize period, CO_2 emissions were similar in all the treatments (Fig. 1f), showing increases only as a consequence of common events (e.g. tillage) in all the soils.

Regarding CH_4-C emissions, only the soils amended with SS showed increasing trends during the wheat period. In contrast, the control soils and the soils amended with CS acted as soil CH_4-C sinks (Fig. 1b). The initial moisture content of the sludge might have favored the emergence of an anaerobic environment, and the higher amount of inorganic N might have promoted the positive emissions from the soils amended with SS [7].

3.3 Effect of the treatments on crop yields

There was not a significant treatment effect on the biomass parameters and the yield of wheat (Table 2). In this sense, it is worth noticing that a higher height of plant was observed in the amended soils with SS (data not shown), which produced wheat lodging in a great part of these treatments, probably due to an overfertilisation of available N in this treatment. On the other hand, for the maize crop, significant differences were observed, with higher values for the CS and the control soil, than for the soil with SS, probably due to the higher N consumption in the SS amended soils during the first crop.

Table 2: Effect of the different treatments on the yield and biomass parameters of the wheat and maize crops.

Treatment	Total biomass[1]		Aerial biomass		Yield (t FW/ha)
	FW (t/ha)	DW (t/ha)	FW (t/ha)	DW (t/ha)	
Wheat crop					
Control	30.5a	21.3a	29.1a	20.6a	13.8a
Sewage Sludge	30.1a	20.4a	29.1a	19.8a	13.3a
Compost	28.6a	19.2a	27.3a	18.7a	12.4a
Maize crop					
Control	51.4b	22.3b	45.1b	20.4b	20.1b
Sewage Sludge	21.7a	11.0a	20.0a	10.4a	11.7a
Compost	49.1b	22.2b	44.7b	20.8b	21.5b

FW-fresh weight; DW-dry weight. [1]Total biomass: root + aerial part;

Average values in a column followed by the same letter are not significantly different at $P < 0.05$ (Tukey-b test).

4. Conclusion and outlook

The incorporation of fresh and stabilized organic amendments to the soils produced a clear increase in the soil organic matter and nitrogen contents, also leading, in the short-term, to an increase of the soil microbial activity, observing only in the second cropping period a clear effect of the treatments over crop yields. In addition, the use of SS induced higher emissions of N_2O, CH_4 and CO_2 than CS (only considering the crop sequence in the GHG balance), this effect being more noticeable immediately after the addition of the organic amendments, during the wheat period.

Acknowledgements

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References

[1] Bustamante, M.A.; Pérez-Murcia, M.D.; Paredes, C.; Moral, R.; Pérez-Espinosa, A.; Moreno-Caselles, J. (2007): Short-term carbon and nitrogen mineralisation in soil amended with winery and distillery organic wastes. Bioresour. Technol. 98, 3269–3277.

[2] Sanz-Cobena A, García-Marco S, Quemada M, Gabriel JL, Almendros P, Vallejo A. (2014): Do cover crops enhance N_2O, CO_2 or CH_4 emissions from soil in Mediterranean arable systems? Sci Total Environ 466–467,164–74.

[3] Khorramdel, S.; Koocheki, A.; Nassiri Mahallati, M.;,Khorasani, R. (2013): Evaluation of carbon sequestration potential in corn fields with different management systems, Soil and Tillage Research 113, 25-31.

[4] Bastida, F.; Kandeler, E.; Moreno, J.L.; Ros, M.; García, C.; Hernandez, T. (2008): Application of fresh and composted organic wastes modifies structure, size and activity of soil microbial community under semiarid climate, Appl. Soil Ecol. 40, 318-329.

[5] Jones, S.K.; Rees, R.M.; Skiba, U.M.; Ball, B.C. (2005): Greenhouse gas emissions from a managed grassland. Glob. Planet. Change 47, 201–211.

[6] Ryals, R.; Silver, W.L. (2013): Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. Ecol. Appl. 23, 46–59.

[7] de Urzedo, D.; Pires-Franco, M., Machado-Pitombo, L.; Braga do Carmo, J. (2013): Effects of organic and inorganic fertilizers on greenhouse gas (GHG) emissions in tropical forestry. Forest Ecol. Manage. 23, 46–59.

TB-P_03 Effect of repeated soil application of organic waste amendments on draught force and fuel consumption for soil tillage

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1. Objectives

One of the effects of repeated soil application of organic waste products (OWP) is the increase in soil porosity and therefore the decrease in soil bulk density. This is also expected to result in a potential decrease in draught force for soil tillage and therefore to a potential decrease in tractor fuel consumption. The objectives of this study were to determine relationships that will facilitate estimation of draught force and tractor fuel consumption for soil tillage from more easily measurable soil parameters such as soil texture, carbon content and OWP amendment history.

2. Methodology

The Crucial experiment is a sandy loam with on average 26.2 % coarse sand, 43.6 % fine sand, 14.3 % silt and 12.6 % clay. It was initiated in 2003 and has mainly been cropped with spring cereals. The treatments are organised in a randomised block design with three blocks and include: household waste compost (normal rate: CH; accelerated rate: CHA), sewage sludge (normal rate: S; accelerated rate: SA), cattle manure (accelerated rate: CMA), cattle slurry (CS), animal deep litter (DL, very straw-rich manure), human urine (HU), NPK fertiliser (NPK) and unfertilised (U, no mineral or organic fertiliser applied). The normal rate of OWP or mineral fertiliser was based on N-input equivalent to approximately 100 kg available N ha-1 year-1, depending on the crop grown. The accelerated rates of application aimed at applying approximately three times the normal available N levels [1].

- Draught force for soil tillage was measured in all the plots of the Crucial experiment on 28 March 2014. The measurement setup consisted of three spring tines operating at a working depth of 12 cm. Each tine was connected to a transducer measuring the horizontal and vertical forces [2]. Specific draught (kPa) and specific fuel consumption were derived from draught force measurements [3].
- The soil was sampled simultaneously with the draught force measurements. Two independent samplings were performed to determine (i) texture and total organic carbon content and (ii) bulk density and water content (in all treatments except HU, CS and DL).

3. Results and discussion

3.1 Relationship between bulk density and soil organic carbon

The soil organic carbon content significantly differed across the treatments ($p<0.001$), covering a wide range of values from the lowest 1.2±0.1 % in the unfertilised (U) to the highest 3.5±0.4 % in the accelerated compost (CHA). Soil bulk density (BD) differed significantly across treatments ($p<0.001$), ranging from 1.23±0.09 g cm^{-3} (CHA) to 1.55±0.02 g cm^{-3} (U).). Bulk density could be predicted from the SOC content with good accuracy (R^2=0.75) according to the equation (Fig. 1):

$$BD = 1.690 – 0.1342 * SOC \quad \text{Eq. 1}$$

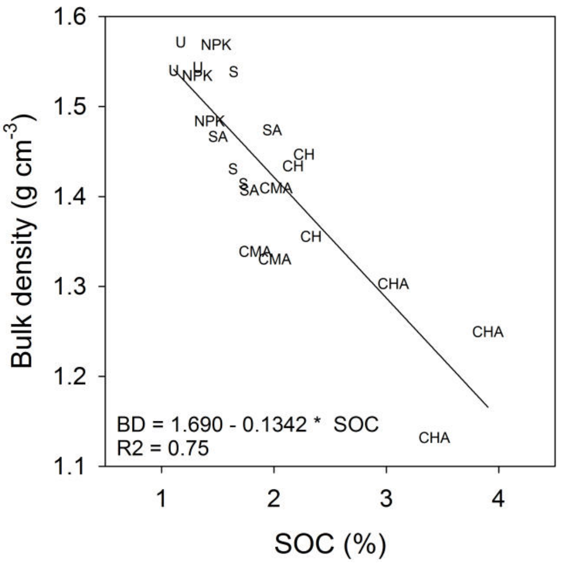

Figure 1: Relationship between soil organic carbon content (SOC) and soil bulk density for the different treatment of the field experiment. Household waste compost at normal (CH) and accelerated (CHA) rates, cattle manure at an accelerated rate (CMA), NPK mineral fertiliser (NPK), sewage sludge at normal (S) and accelerated (SA) rates [4].

3.2 Relationship between specific draught and soil characteristics

The draught force varied across treatments from 2.05 ± 0.40 kN (CHA) to 2.93 ± 0.24 kN (HU), which resulted in a specific draught (SDFr) ranging from 87.45 ± 17.28 kPa (CHA) to 125.21 ± 10.20 kPa (HU), with an average of 111.8 kPa. The ANOVA revealed a significant effect of clay content ($p<0.01$) and bulk density ($p<0.001$) on SDFr (Table 5). The effect of SOC was significant ($p<0.05$) after accounting for the variance explained by clay and bulk density. The effect of soil water content was not significant after subtracting the variance explained by clay, BD and SOC content, whereas soil cohesion had a significant effect ($p<0.05$) and the treatment and block effects were subsequently non-significant (Table 1). This likely indicates that the change in specific draught was controlled in the first place by the amount of SOC accumulated and its effect on bulk density rather than its composition.

Table 1: ANOVA table for testing the effect of clay content, bulk density (BD), SOC content, soil water content, soil cohesion, treatments and block effect on specific draught for soil tillage [4].

	Df	Sum of squares	Mean squares	F value	P-value
Clay	1	1082.1	1082.1	17.6183	0.004049**
BD	1	2257.4	2257.4	36.7537	0.00051***
SOC	1	586.5	586.5	9.5493	0.017562*
Water content	1	227.5	227.5	3.7039	0.095686
Cohesion	1	523.2	523.2	8.5182	0.022385*
Treatment	6	358.5	59.7	0.9727	0.505323
Block	2	477.2	238.6	3.8847	0.073294
Residuals	7	429.9	61.4		

Significance levels: *: $p<0.05$, ** : $p<0.01$, ***: $p<0.001$

Multiple linear regressions were performed in order to relate specific draught from easily measurable soil parameters (texture and SOC). A stepwise regression procedure revealed that the inclusion of more than one texture class did not improve the regression accuracy and that the combination of SOC and clay content in the linear regression gave the best results. Specific draught was related to the SOC and clay content, explaining 67% of the variance on the full dataset, 59% of the variance in cross validation (R2CV) and with a RMSECV of 10.4 and RPD of 1.6 according to the equation:

$$SDFr = 58.85 + 6.76 * Clay - 17.35 * SOC \qquad \text{eq. 2}$$

where clay and SOC content are expressed as % (g $100g^{-1}$ bulk soil) (Fig. 1). The linearity of this relationship was tested for SOC effect by adding the square effect of SOC, which proved to be non-significant. It should be noted that this relationship is valid for this field site at this specific time of the year but can hardly be extrapolated to other situation. It is presented to explore the relationships between SOC and specific draught.

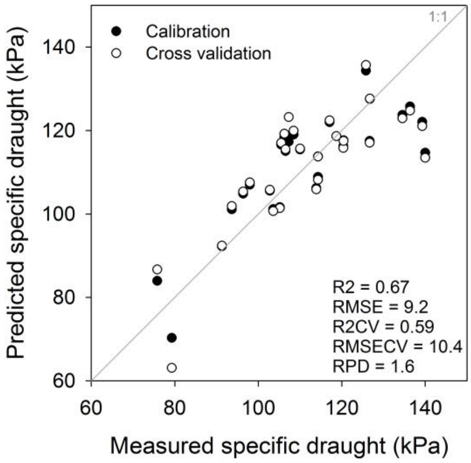

Figure 2: Multiple linear regression between specific draught and clay and SOC content according to eq. 2. Calibration predictions and cross-validated predictions (leave-one-out cross validation) [4].

3.3 Fuel consumption for soil tillage

Fuel consumption based on the draught force measured in the field experiment was calculated as ranging between 3.4 ± 0.5 L h^{-1} (CHA) and 4.7 ± 0.1 L h^{-1} (HU) (Table 7). Fuel consumption was also high in the NPK treatment compared with the other fertiliser treatments (4.5 ± 0.4 L h^{-1}). When expressed as a percentage difference in fuel consumption compared with the NPK treatment, taken as a standard reference fertilisation, fuel savings ranged from 5 ± 13 % (SA) to 25 ± 11 % (CHA). The household waste compost applied at the normal rate also led to quite a large fuel saving of 14 ± 11 %. Conversely, the application of human urine (HU) led to slightly greater fuel consumption than in the NPK treatment (+ 5 ± 3 %). Fuel savings for soil tillage therefore appear to be an important environmental benefit of soil application of organic waste that should be taken into consideration. This should however be placed in the context of the overall chain of waste recycling, taking into account other emissions related to soil application of organic wastes. This can only be done by using integrative environmental assessment methods such as life cycle assessment.

Table 2: Tractor fuel for the tine tillage operation, and fuel consumption savings or increases as a percentage of the fuel consumption in the NPK treatment. Standard deviations presented in brackets.[4]

	Fuel for operation (L h^{-1})	**Fuel savings (% of difference compared to NPK treatment)**
CHA	3.4 (0.5)	-25 (11)
CH	3.8 (0.5)	-14 (11)
DL	4.0 (0.4)	-11 (10)
U	4.1 (0.3)	-7 (8)
CMA	4.1 (0.7)	-7 (16)
S	4.2 (0.5)	-5 (12)
CS	4.2 (0.5)	-5.2 (12)
SA	4.2 (0.6)	-5 (13)
NPK	4.5 (0.4)	
HU	4.7 (0.1)	5 (3)

Household waste compost at normal (CH) and accelerated (CHA) rates, cattle manure at an accelerated rate (CMA), cattle slurry (CS), deep litter (DL), human urine (HU), NPK mineral fertiliser (NPK), sewage sludge at normal (S) and accelerated (SA) rates, unfertilized (U)

4 Conclusion and outlook

This study showed that a change in specific draught for soil tillage was significantly explained by clay content, bulk density, SOC content and soil cohesion. However, no evidence was found that the quality of SOC accumulated for different organic wastes influenced specific draught. Specific draught could be related to clay and SOC content, explaining 67 % of the variance. The change in specific draught may result in fuel savings of up to 25 % on soil tillage for compost applied at an accelerated rate (reaching 3.5 % SOC) and up to 14 % for compost applied at a normal rate (reaching 2.3 % SOC). This could represent an important environmental benefit of the soil application of organic waste. This benefit should therefore be included in environmental assessments of waste recycling chains, such as life cycle assessment, for comparison with the other environmental emissions related to waste recycling, and provide a more comprehensive environmental balance of waste recycling through soil application.

Acknowledgements

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References

[1] Poulsen, P.H.B., Magid, J., Luxhøi, J., de Neergaard, A. (2013): Effects of fertilization with urban and agricultural organic wastes in a field trial – Waste imprint on soil microbial activity, Soil Biology and Biochemistry 57, 794-802

[2] Nyord, T., Kristensen, E.F., Munkholm, L.J., Jorgensen, M.H. (2010): Design of a slurry injector for use in a growing cereal crop. Soil & Tillage Research 107, 26-35.

[3] ASABE, 2011. Agricultural Machinery Management Data. D497.7. American Society of Agricultural and Biological Engineers, St. Joseph, MI, USA, p. 8.

[4] Peltre, C., Nyord, T., Bruun, S., Jensen, L.S., Magid, J. (2015): Repeated soil application of organic waste amendments reduces draught force and fuel consumption for soil tillage. Agriculture, Ecosystems & Environment 211, 94-101.

TB-P_04 Use of *Phoenix dactylifera* pruning biomass for the development of more fibrous and recalcitrant composts

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1. Objectives

The main objective of this work was to study the role of pruning biomass obtained from date palm (*Phoenix dactylifera)* as an ingredient in several co-composting scenarios. These scenarios were designed to obtain more fibrous and recalcitrant composts to be used with specific agronomic purposes, linked to soilless uses and to the recovery of the physical properties in degraded soils.

2. Methodology

Four ternary composting experiments, in triplicate, were carried out. The mixtures were established using 17% of sewage sludge (dry weight basis, d.w.) in all the scenarios, with a combination of date palm pruning biomass (rachis + leaf) (DPPB) and grass clippings (GC) in the proportions 17:66, 42:41, 67:16 and 83:0 (d.w.) of DPPB and GC, to obtain the mixtures A-B-C-D, respectively. Table 1 present the characteristics of the raw materials used. The mixtures were composted in an enclosed, passively aerated composting vessel (350L), according the system used by Bustamante et al. [1]. All the piles were turned 3 times during the composting process (7, 20 and 30 days) in order to improve both the homogeneity of the material and the biodegradation process. After the last turning, the temperature of the piles were stable and near to that of the surrounding atmosphere, and therefore the bio-oxidative phase of composting was considered finished. Then, the composts were left to mature in the same vessel for a further one month period, approximately. The moisture of the piles was controlled weekly by adding the necessary amount of water to maintain moisture content not below 50%. The piles were sampled in four occasions (at 0, 15, 30 and 60 days, coinciding with the initial phase, the thermophillic phase, the end of bio-oxidative phase and maturity. The samples were taken by mixing seven sub-samples from seven sites of the pile, from the whole profile (from the top to the bottom of the pile). Throughout the composting process, the temperature evolution was monitored and different chemical parameters were determined. In the mature composts, several parameters related to recalcitrance (Chemical stability degree (SD) according to Lopez et al. [2]) and physical properties were also evaluated.

Table 1: Main characteristics of the raw materials.

Parameter	Sewage sludge, SS	Date palm pruning biomass, DPPB	Grass clippings, GC
Water content (%)	80.9	36.4	70.6
pH	6.38	6.25	7.76
Electrical conductivity (dS/m)	5.40	6.43	7.84
Total organic matter (%)	56.6	91.6	73.5
Total organic carbon (%)	40.5	56.4	48.8
Total nitrogen (%)	6.35	1.39	3.63
C/N	6.40	40.7	13.5
P (%)	2.53	0.41	1.79
Na (g/kg)	3.70	4.50	12.2
K (g/kg)	3.45	25.7	62.5

3. Results and discussion

Thermal profiles of the mixtures were negatively affected by DPPB compared to GC biomass (figure 1), according to the EXothermic Index (EXI). This index was calculated as the daily summation of the temperature increment in the pile compared to the ambient temperature in the bioactive period, 30 days; (EXI = -3.66 DPPB (%) + 584, R^2 = 0.85). The mixtures that reached higher temperature values were A and B which also are the ones that contains more GP in the mixture and less DPPB. Moreover, these composting mixtures showed temperature values that assured the sanitisation, on the contrary than the mixtures C and D.

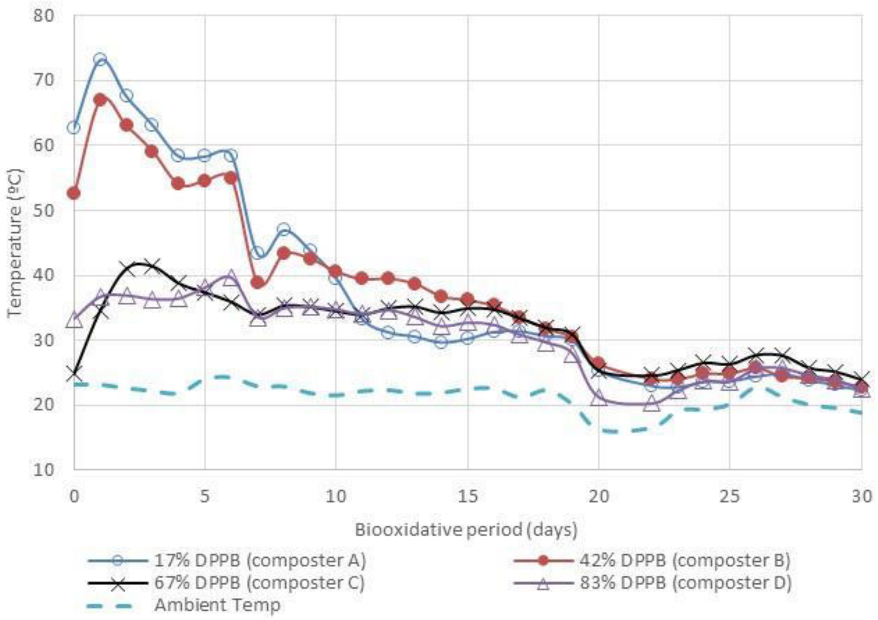

Figure 1: Thermal profile of the mixtures (17% date palm pruning biomass (DPPB)=composter A, 42% DPPB=composter B, 67% DPPB=composter C, 83% DPPB=composter D).

The evolution of the main physico-chemical and chemical characteristics of the mixtures is summarized in Table 2. The pH values remained close to neutrality throughout the process for all the mixtures, as it has been observed by Bustamante et al. [3] in an experiment of co-composting of sewage sludge and winery-distillery wastes. On the other hand, the salinity, determined with the electrical conductivity (EC) was directly related to GC content in the mixture, this aspect being one of the main constraints for compost intended for soilless use purposes. In addition, EC values coincided with the degree of mineralisation of the materials, being lower in composter D (5.51 dS/m), followed by composter C (6.81 dS/m), while mixtures A (9.73 dS/m) and B (8.65 dS/m) showed higher values, in parallel with the low dissolved organic matter contents DOM (below 40%) and higher stabilisation degree (SD) (46 and 49%, respectively) observed.

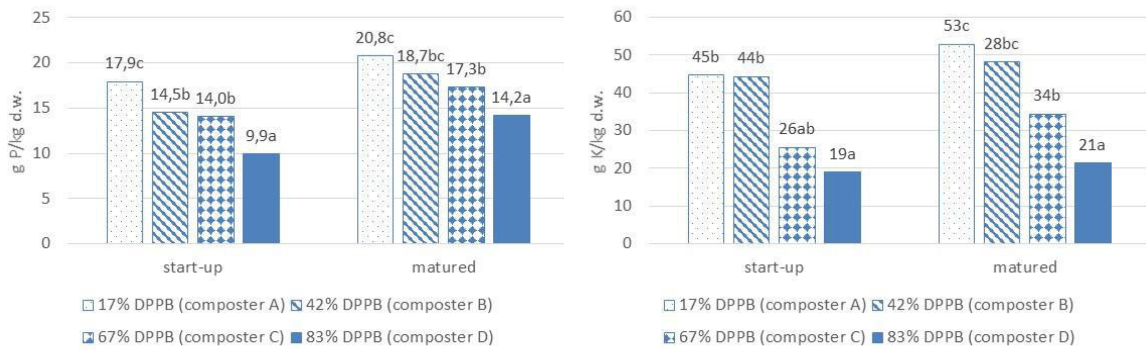

Figure 2: Phosphorus and potassium concentrations in initial and mature composting samples, DPPB: date palm pruning biomass. Average values in columns followed by the same letter are not significantly different at $P < 0.05$ (Tukey-b test), considering start-up and matured separately.

However, the presence of GC presence also increased the NPK contents at the end of the process (figure 2), as it can be expected due to mineralisation, and especially Na and K contents compared to DPPB. In general, the increasing proportion of DPPB in the initial mixtures implied higher values of the total organic matter and C/N ratio, as well as lower contents of total nitrogen in the mature composts.

Comparing the loss of organic matter (OM) among the mixtures, mixture D showed a higher loss (66.85 %) than mixtures B (52.64 %), C (45.49 %) and A (44.69 %). Considering each mixture through time, the main losses were observed during the first 15 days and decreased with time, except for the mixture D, where a larger OM reduction of 34.42 % was observed during the last period (30-60 days). The higher loss in composter D can be related to a higher initial OM content rather than stabilisation, because the SD value at the end of the process (39.74%) was the lowest observed and the content of degradable organic matter (DOM), was still high (51.28 %). The high proportion of DPPB in the mixture D together with the high initial C/N ratio, could contribute to slow down the composting process, with a final OM content of 75.75 %, which can be considered high. However, mixture C also presented high proportion of DPPB and similar initial C/N ratio, but the mixture was balanced with grass clippings, more degradable and with higher N contents than DPPB, which could accelerate the process. This fact could be observed in the slightly lower final OM content (73.72 %) and the higher SD (42.42 %). Mixtures with lower initial C/N ratio showed more intense degradation (lower final OM contents of 63.21 % and 68.86 %, respectively for A and B mixtures) and allowed higher final SD (49.31 % and 46.85 %, respectively), which were close to mature compost (at least, 50 %). In case of mixture D, the lack of a nitrogen source implied a lower degradation, producing a compost with high DOM, even recalcitrant o resistant organic matter (ROM, Lopez et al. [2]) was quite similar in all the composts (32.12%, 34.18%, 33.26% and 33.83%, respectively for mixtures A, B, C and D).

Table 2: pH, electrical conductivity (EC), total organic matter (TOM) and total N (TN) evolution in the four ternary composting experiments and stability degree (SD) and proportion of resistant nitrogen (nHN/orgN) for mature composts.

SS: DPPB:GC Ratio (d.w.)	Sampling (days)	pH	EC (dS/m)	% TOM dw	% SD (DOM/TOM)	% TN dw	% nHN/orgN	C/N
17:17:66	0	7.08	6.17	75.65		2.56		16.51
Mixture A	15	7.05	9.33	68.73		3.01		12.71
	30	6.84	9.52	65.37		3.31		10.88
	60	6.83	9.73	63.21	49.31	3.51	43.10	10.21
17:42:41	0	7.13	6.69	82.36		2.34		19.18
Mixture B	15	7.07	7.86	72.07		3.14		13.09
	30	6.93	8.34	70.30		3.16		12.69
	60	6.83	8.65	68.86	46.85	3.01	42.40	13.06
17:67:16	0	6.98	4.80	83.73		1.39		37.38
Mixture C	15	7.02	6.76	77.70		2.01		24.82
	30	6.96	7.24	76.23		2.20		20.17
	60	6.98	6.81	73.72	42.42	2.51	47.51	16.37
17:83:0	0	6.60	5.06	90.41		1.35		36.14
Mixture D	15	6.45	6.16	84.51		2.00		23.66
	30	6.35	5.96	82.65		2.33		19.69
	60	6.43	5.51	75.75	39.74	2.46	40.56	17.74

DOM: dissolved organic matter; SS: sewage sludge, DPPB: date palm pruning biomass (rachis + leaf), GC: grass cli pings. 0 days: initial phase; 15 days: thermophilic phase; 30 days: end of bio-oxidative phase, 60 days: maturity.

Regarding nitrogen, this nutrient should be conserved during the process, in order to avoid gas emissions and nutrient losses. The variation during composting indicated that in mixtures B and D nitrogen losses were about 28%. In case D, even if a high initial C/N was proposed, probably the carbon contents were not available enough for microorganisms and due to this, the material was not properly transformed and the nitrogen from sewage sludge was lost rather than conserved. On the other hand, in mixture C, which also presented a high initial C/N ratio, GC could supply part of the available carbon, demanded in the earlier stages, increasing the final nitrogen content in an 11 %. Mixture A, which showed the lowest DPPB proportion, had a loss of N about 9 %, despite the higher content in degradable nitrogen (1.91 %). Nevertheless, in mixtures B, C and D easy degradable nitrogen (the opposite to nhN), is about 60 % of the organic nitrogen, while in mixture C is 52 %. Considering this, it can be assessed that far less than 50 % of organic nitrogen is stabilised, which could contribute to nitrogen losses once applied the compost to soil, but far less than non-stabilised organic waste, such as sewage sludge.

Finally, cation exchange capacity (CEC) was negatively correlated to DPPB presence (figure 3). As it was observed in SD, the higher is the proportion of GC in the mixture, the higher CEC value (mixtures A and B) was found. Mixture D, which did not contain grass clippings, presented the lowest CEC value at the end of the process.

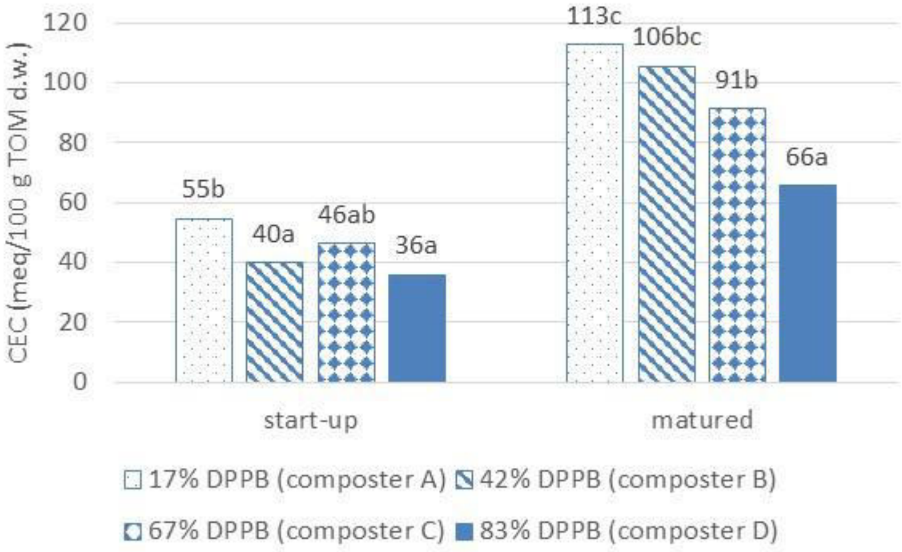

Figure 3: Cation exchange capacity in the four composting experiments at the initial and mature stages. DPPB: date palm pruning biomass (rachis + leaf); TOM: total organic matter.

4. Conclusion and outlook

Co-composting strategies can vary strongly the nature and uses of derived compost. In our experiment, the use of DPPB biomass in composting induced significant increases of total organic matter but this does not corresponds to higher stability degree, which is achieved in the mixtures with higher content in grass clippings. However, more NPK fertilizing capacity and salinity were observed in GC enriched compost. In relation to organic amendment capacity, high content in DPPB can need larger periods to be adequately transformed and achieve a SD over 50 % to guarantee that organic matter applied is going to be mineralized step by step. On the other hand, when DPPB is high and complemented with source of available carbon, transformation can be improved and nitrogen conserved.

Acknowledgements

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References

[1] Bustamante, M.A.; Restrepo, A.P.; Alburquerque, J.A.; Pérez-Murcia, M.D.; Paredes, C.; Moral, R.; Bernal, M.P. (2013). Recycling of anaerobic digestates by composting: effect of the bulking agent used. J. Clean. Prod. 47, 61–69.

[2] Bustamante, M.A.; Paredes, C.; Moral, R., Moreno-Caselles, J.; Pérez-Murcia, M.D.; Pérez-Espinosa, A.; Bernal, M.P. (2007). Co-composting of distillery and winery wastes with sewage sludge. Water Science & Technology 56, 187-192.

[3] López, M.; Huerta-Pujol, O.; Martínez-Farré, F.X.; Soliva, M. (2010): Approaching compost stability from Klason lignin modified method: Chemical stability degree for OM and N quality assessment. Resources, Conservation and Recycling 55, 171–181.

TB-P_05 Manure-derived biochars behave also as fertilizer

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1. Objectives

A plant growth experiment was carried out in a climatic chamber in order to test the following hypotheses: (i) pyrolysis of manure feedstock would produce biochar with higher fertilizer value than standard wood-derived biochar; (ii) manure-derived biochars would eventually alter the soil chemical and biological properties in a manner modulated by the biochar process conditions (i.e. pyrolysis temperature and feedstock type).

2. Methodology

The experiment was carried using a randomized block design with three replicates in a climatic growth chamber with a controlled environment (20 °C). Two types of soil (NW Italy) with contrasting physico-chemical properties were used: (1) a silt-loam soil (pH: 6.2, O.C.: 1.2%, N: 0.15%, Olsen P: 23 mg kg^{-1}, K: 42 mg kg^{-1}) and (2) a sandy soil (pH: 8.3, TOC: 0.52%, N: 0.057%, Olsen P: 14 mg kg^{-1}, K: 28 mg kg^{-1}). Treatments included in the experiment were: four different types of manure-derived biochars based on the combination of different feedstock (poultry litter and swine manure) and different pyrolysis temperatures (400 or 600 °C) (PL400, PL600, SM400, SM600); a readily available standard biochar from wood chip (WC) produced at high temperature (1000 °C) and a control without amendment (Control). Soils were biochar-amended at 2% w/w (~20 Mg ha^{-1}) and soil moisture was adjusted to 60% water filled pore space (WFPS). Seeds of Italian ryegrass (*Lolium multiflorum L.*) were sown (0.3 g pot^{-1}) in plastic pots (13.5 x 13.5 cm) filled with the soil-biochar mixture (1.5 kg dry soil pot^{-1}). Soil moisture was adjusted every 3 days. No extra fertilization was applied after sowing but all treatments had been fertilized with 170 kg ha^{-1} of NH_4NO_3 during a previous experiment [1]. A total of five harvests were completed over the entire growth period of 150 days and dry matter (DM) yield was calculated. Roots were extracted at the end of the experiment for further analysis. Above ground biomass at each harvest was analyzed for NPK concentration in order to estimate their uptake by the plants. Both chemical (TOC, TN, pH, CEC, P and cations) and microbial properties (microbial biomass C and soil enzymes: dehydrogenase and β-glucosidase) of biochar-amended soils were investigated at the end of the experiment. All data were analyzed separately using a one-way ANOVA for each soil type after checking the assumptions of normality and homoscedasticity. A post-hoc Tukey test at 5% was used to check significant differences among treatments.

3. Results and discussion

3.1 Biochar production and characterization

The technical and economic feasibility of producing biochar from swine and poultry manures has been modeled and simulated based on experimental data and is of scientific interest [2,3]. The results of these modeling exercises suggest that the moisture content of the feedstock plays a key role in determining the economic viability. In the case of swine manure the cost of solids separation (to 30% solids) followed by drying was found to be prohibitive whereas producing biochar from poultry litter (25% moisture) in an organic rankine cycle combined heat and power plant is an option. It has been estimated that with a gate fee of €13 Mg^{-1} the break-even selling point of the biochar is €90 Mg^{-1} [3].

Results from the biochar characterization showed that C, ash content, pH, surface area and nutrient (both macro and micro) contents of manure-derived biochars increased with increasing pyrolysis temperatures while the opposite was the case for the volatile matter content and CEC (Table 1). The C content, pH and surface areas of manure-based biochars were lower compared to standard wood chip biochar while the opposite was observed in relation to their CEC and nutrient contents. The higher CEC and nutrient contents of manure- derived biochars could be used to improve the crop growth through improved soil CEC plus increased nutrient retention into soil-biochar matrix [4].

Table 1: Physico-chemical characteristics different biochars utilized in the experiment

Biochar type	C	N	VM	Ash	pH	P[a]	Ca[b]	Mg[b]	K[b]	CEC	Surface area
	%					g kg^{-1}				cmol$_c$ kg^{-1}	m^2 g^{-1}
PL400	52.1	5.85	44.9	25.3	9.5	12.25	28.3	17.3	38.8	30.2	5.4
PL600	52.8	4.01	24.7	35.4	10.4	15.40	35.9	24.0	58.8	27.5	6.3
SM400	54.9	2.23	29.9	27.5	10.0	9.70	20.3	15.7	16.2	52.5	5.8
SM600	57.9	1.79	17.8	34.5	10.4	15.55	28.9	21.3	35.3	18.6	10.6
WC	89.3	0.27	15.3	7.8	11.0	0.73	13.6	3.2	2.6	14.8	178.3

PL400, PL600- poultry litter biochars at 400 and 600 °C, SM400, SM600- swine manure biochars at 400 and 600 °C, WC- wood chip biochar at 1000 °C.
VM- volatile matter; CEC- cation exchange capacity; [a] available P, [b] total.

3.2 Crop yield and nutrient uptake

Results from ryegrass growth demonstrated that the low temperature manure-derived biochars significantly increased above-ground biomass yield compared to the control and other biochar treatments in both soils (Table 2). All biochars increased root mass compared to the control on silt-loam soil and for PL400 and SM400 treatments in the case of sandy soil. Both above and below ground biomass yield showed a significantly positive correlation with the N content of the biochars ($n=5$, $P<0.05$, $r=0.867$ on average) on both soils. The highest shoot DM yield, which increase by 47.5% compared to the control was recorded in PL400 treated silt-loam soil, followed by a 34.4% increase for the SM400 treatment. Similarly, an increase of root biomass (DM) of up to 127.2% compared to the control was observed for PL400 treated silt-loam soil followed by a 93.8% increase for the SM400 treatment. The increased biomass production on manure-derived biochar amended soils were the result of direct nutrient additions from these chars plus an increase in nutrient retention as well as availability on soil-biochar matrix [5].

Table 2: Effects of biochar on above and underground biomass yield of ryegrass and nutrient uptake on two soils

Treatments	Shoot yield / g DM pot^{-1}	Root yield / g DM pot^{-1}	Aboveground uptake N / mg pot^{-1}	P / mg pot^{-1}	K / mg pot^{-1}
			Silt-loam soil		
Control	12.2 c	0.81 e	480.3 b	47.1 b	302.5 d
PL400	18.0 a	1.84 a	670.7 a	109.6 a	825.6 a
PL600	14.8 abc	1.39 bc	483.2 b	91.3 a	763.4 ab
SM400	16.4 ab	1.57 b	647.8 a	99.2 a	600.8 bc
SM600	13.0 bc	1.31 c	457.9 b	86.8 a	450.2 cd
WC	11.53 c	1.03 d	454.6 b	38.2 b	372.5 d
SE	1.12	0.06	47.5	10.5	57.9
P	0.01	< 0.001	0.017	0.002	< 0.001
			Sandy soil		
Control	5.97 b	0.39 b	198.4 b	23.7 b	206.4 d
PL400	8.59 a	0.91 a	263.5 a	44.6 a	471.0 a
PL600	7.26 ab	0.66 ab	247.3 a	33.4 ab	436.6 a
SM400	8.11 a	0.68 ab	282.4 a	44.3 a	337.3 b
SM600	7.01 ab	0.56 ab	239.2 ab	32.9 ab	300.9 bc
WC	5.94 b	0.47 b	193.6 b	21.3 b	248.3 cd
SE	0.53	0.13	15.4	4.39	24.8
P	0.019	0.174	0.009	0.01	< 0.001

SE- standard error of the mean, DM- dry matter.
Mean values followed by the same letter are not significantly different ($P<0.05$).

A significant N uptake increase in ryegrass with PL400 and SM400 treatments provided further proof that the N content of these chars are readily available to the plants (Table 2). This validates the proposition that such biochars have a fertilizer value [5]. The N uptake did not differ significantly relative to the control for PL600, SM600 and WC treatments in the silt-loam soil, while it increased significantly for all manure-derived biochar (except SM600) treatments in the poorer sandy soils. The highest increase in N uptake, 42.3% compared with Control, was recorded in SM400 treated sandy soil followed by a 39.6% increase in PL400 treated silt-loam soil. All manure-derived biochars greatly enhanced P uptake compared with the control and WC treatments in silt-loam soil, which was also the case for PL400 and SM400 treatments in sandy soil. The low P uptake in sandy soil compared to silt-loam may be due to the P fixation in calcareous soils as the sandy soil used in this study had significant amount of $CaCO_3$ (16% on average) [6]. Enhanced uptake of biochar P by plants provided further evidence that a large portion of the P is available for the crops and these materials can also act as P fertilizer. Similarly K uptake was significantly increased in manure-derived biochar treatments in both soils.

3.3 Soil chemical and biological properties

Biochar amendment to both soils significantly modified both soil chemical and microbial properties (Table 3). All biochars significantly increased TOC compared to Control, as expected. The highest increase in TOC, by 112.4% compared to the control, was observed for the standard biochar (WC) treated silt-loam soil followed by a 90.0% increase with the SM400 treatment. Such a large increase in TOC can be explained by the increased root biomass turnover associated with increased biomass production as result of soil biochar addition [7]. All manure-derived biochars significantly increased total N and available P content compared to the control and WC treatments in both soils and this is the result of the direct release of these elements into the soil-biochar matrix [8]. Exchangeable K content was rather low, except for PL600 and SM400 treatments, in silt-loam soil and was not noticeable different from the control, while all manure biochars significantly increased K availability in sandy soils. The lower exchangeable K in silt-loam than in sandy soils is due to low soil pH (6.5-7.5 recorded pH among treatments). The CEC among biochar treatments did not vary significantly in either soil. The organic matter in a biochar feedstock often loses its functional groups during pyrolysis but this could be restored again over time as result of biochar oxidation in soils. However it may take several years before any effect on soil CEC would be observed [8].

Table 3: Effects of biochar amendment on chemical and microbial properties of silt-loam and sandy soils at the end of the ryegrass growth experiment

Treatment	TOC / %	TN / %	Olsen P / mg kg^{-1}	Exch. K / mg kg^{-1}	CEC / cmol$_c$ kg^{-1}	MBC / µg g^{-1}	Dehydrogenase[a] / µg INTF g^{-1} h^{-1}	β-glucosidase[b] / µg PNP g^{-1} h^{-1}
				Silt-loam soil				
Control	1.29 b	0.134 d	20.8 d	10.8 c	16.1	83.9 a	48.6 c	187.7 a
PL400	2.29 a	0.22 a	95.1 b	37.8 c	16.5	85.1 a	81.1 ab	127.0 b
PL600	2.28 a	0.213 a	92.4 b	119.4 a	14.6	49.1 b	53.7 bc	149.5 b
SM400	2.45 a	0.177 b	76.9 c	57.5 b	16.4	20.3 c	92.1 a	92.2 c
SM600	2.43 a	0.169 bc	106.4 a	22.2 c	15.9	26.3 c	80.9 ab	152.1 b
WC	2.74 a	0.138 cd	20.7d	24.7 c	16.3	75.2 a	18.4 d	215.0 a
SEM	0.26	0.01	2.91	6.14	0.90	7.01	8.00	9.21
P	0.031	< 0.001	< 0.001	< 0.001	0.710	< 0.001	< 0.001	< 0.001
				Sandy soil				
Control	0.91 c	0.06 c	12.6 e	18.1 cd	6.92	30.4 d	48.4 ab	288.2 a
PL400	1.67 b	0.15 a	108.1 c	104.6 b	6.89	83.4 a	57.7 a	190.8 b
PL600	1.89 b	0.14 ab	136.5 a	143.7 a	6.75	62.2 bc	38.7 bc	177.6 b
SM400	1.97 b	0.092 bc	129.3 b	82.7 b	7.38	51.6 c	44.1 abc	187.3 b
SM600	1.75 b	0.13 ab	83.7 d	88.1 b	7.78	78.9 ab	27.2 c	315.9 a
WC	2.93 a	0.081 c	14.2 e	9.4 d	6.49	67.8 abc	4.881 d	287.9 a
SE	0.183	0.02	2.21	8.44	0.50	5.67	5.99	25.2
P	< 0.001	0.008	< 0.001	< 0.001	0.533	< 0.001	0.001	0.005

SE- standard error of the mean, TOC- total organic carbon, TN- total nitrogen, MBC- microbial biomass carbon; [a] expressed as produced level of iodonitrotetrazolium formazan –INTF and [b] as produced level of p-nitrophenol –PNP. Means followed by the same letter are not significantly different (*P*<0.05).

The microbial biomass carbon (MBC) values were significantly lower compared with the control for PL600, SM400 and SM600 treatments in silt-loam soil, while the values were significantly higher for all biochar treatments in sandy soil. The lower MBC values of SM400, SM600 treatments in silt-loam soil could be due to reduced microbial consumption of volatile compounds [9]. However, this does not hold true in sandy soil for the same treatments. The reason behind this may be due to the release of some biochar compounds from biochar produced at 600 °C leading to higher K_2SO_4 extractable C after fumigation as the method used (fumigation-extraction) has its own limitation and drawbacks [10]. Dehydrogenase activity (DA) significantly increased compared with the control for PL400, SM400, SM600 treatments in silt-loam soil while the values did not differ significantly for manure biochar treatments, except for SM600, in sandy soil. A significantly lower DA was found for WC treatment compared to control and other biochar treatments. The DA showed a positive correlation with volatile matter content of the chars and is often associated with microbial respiration [9]. The β-glucosidase activity was found to be significantly lower in the manure-derived biochar treatments than in the control, while the values showed no significantly difference for WC treatments in either soil and for the SM600 treatment in sandy soil. The activity of this enzyme over all treatments, on average, was 1.5 times less in silt-loam soil than in the sandy soil and is often related to the nature of C and VM present in the biochars in addition to native SOC [9]. These two enzymes (DA and β-glucosidase) thus provide insights into the degree of resistance of organic matter against microbial degradation in soils amended with biochar. Further experimental verification of these enzyme responses plus microbial biomass on soil biochar additions is necessary.

4. Conclusion and outlook

The total dry matter yields of ryegrass (both above and belowground biomass) were related to both positive nutrients content of biochars and enhanced soil characteristics. Low temperature manure derived biochars (PL400 and SM400) showed a good potential to be utilized as NPK-fertilizers with a significant value. This unique property of manure-derived biochar could fulfill the additional NPK demand and can be benefitted by the crops. The differences in physico-chemical properties between manure-derived and easily available standard wood chip biochars were well reflected with observed differences in soil chemical properties as well as enzymatic activities. The results from this study could thus be helpful in setting a new biochar guidelines based on biochar quality composition.

Acknowledgements

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References

[1] Subedi, R.; Taupe, N.; Pelissetti, S.; Petruzzelli, L.; Bertora, C.; Leahy, J.J.; Grignani, C. (2015): Greenhouse gas emissions and physico-chemical properties of soils amended with manure-derived biochars: influence of pyrolysis temperature and feedstock type. Submitted to Journal of Environmental Management.

[2] Wnetrzak. R.; Kwapinski, W.; Peters K.; Sommers, S.G.; Jensen, L.S.; Leahy, J.J. (2013): The influence of the pig manure separation system on the energy production potentials. Bioresource Technology 136, 502-508.

[3] Huang, Y.; Anderson, M.; McIlveen-Wright, D.; Lyons. G.A.; McRoberts, W.C.; Wang, Y.D.; Roskilly, A.P.; Hewitt, N.J. (2015): Biochar and renewable energy from poultry litter: a technical and economic analysis based on computational simulations (article in press). Applied Energy, http://dx.doi.org/10.1016/apenergy.2015.01.029.

[4] Hossain, M.K.; Strezov, V.; Chan, K.Y.; Ziolkowski, A; Nelson, P.F. (2011): Influence of pyrolysis temperature on production and nutrient properties of wastewater sludge biochar. Journal of Environmental Management 92, 223–228.

[5] Cantrell, K.B.; Hunt, P.G.; Uchimiya, M.; Novak, J.M.; Ro, K.S. (2012): Impact of pyrolysis temperature and manure source on physicochemicalcharacteristics of biochar.Bioresource Technology 107, 419–428.

[6] Cui, H.; Wang, M.K.; Fu, M.L.; Ci, E. (2011): Enhancing phosphorus availability in phosphorus-fertilized zones by reducing phosphate adsorbed on ferrihydrite using rice straw-derived biochar. Journal for Soil Sediments 11, 1135–1141.

[7] Kammann, C.; Ratering, S.; Eckchard, C.; Muller, C. (2012): Biochar and hydrochar effects on greenhouse gas fluxes from soils. Journal of Environmental Quality 41, 1052–1066.

[8] DeLuca, T.H.; MacKenzie, M.D.; Gundale, M.J. (2009): Biochar effects on soil nutrient transformation. Chapter 14. In: Lehmann, J., Joseph, S. (Eds.), Biochar for Environmental Management: Science and Technology. Earthscan, London, U. K., pp. 251–270.

[9] Ameloot, N.; De Neve, S.; Jegajeevagan, K.; Yildiz, G.; Buchan, D.; Funkuin, Y.N.; Prins, W.; Bouckaert, L.; Sleutel, S. (2013): Short-term CO_2 and N_2O emissions and microbial properties of biochar amended sandy loam soils. Soil Biology and Biochemistry 57, 401–410.

[10] Kuzyakov, Y.; Subbotina, I.; Chen, H.Q.; Bogomolova, I.; Xu, X.L. (2009) Black carbon decomposition and incorporation into soil microbial biomass estimated by C-14 labeling. Soil Biology and Biochemistry 41, 210–219.

TB-P_06 The impact of rock mineral wool on water retention in a conventional growth medium, and development of zonal pelargoniums

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1. Objectives

The greenhouse and containerised nursery industries utilise artificial root substrates composed of organic and inorganic components. Sphagnum peat is usually the main component during the formulations of artificial substrates due to its many properties that make it an ideal constituent. Environmental concerns (1, 2) and increasing prices have generated interest in the development of alternatives to peat.
Rock mineral wool was investigated, as used as a component of the plant growth media. Using an appropriate proportion of rock mineral wool, we wanted to improve the water retention capacity of the soil growth media in order to ensure enough macro-pores for draining the excess water away whist maintaining an adequate level of oxygen.

2. Methodology

The presented study includes a conventional growth medium for pot plants and rock mineral wool. The rock mineral wool was used as 2x2cm cubes, and 2 and 4cm thick discs (Figure 1).

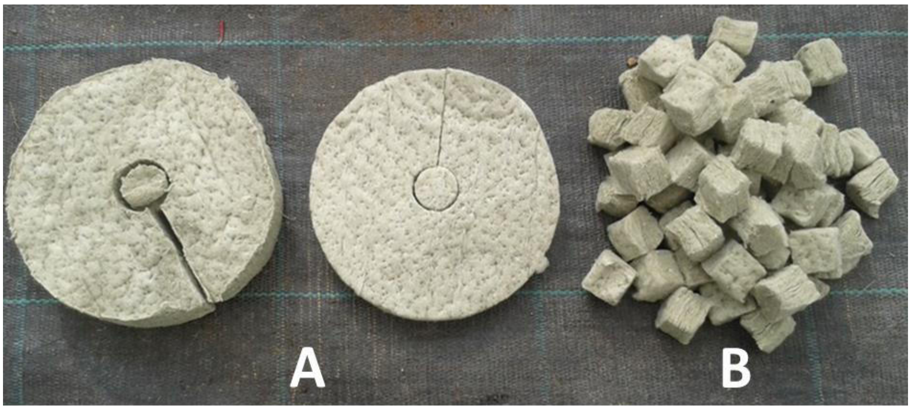

Figure 1: Rock mineral wool; A-2 and 4cm thick discs; B- 2×2cm cubes.

The experiment was conducted under controlled conditions within a greenhouse. We studied 6 different treatments (Table 1) over two separate experiments. Treatments within the experiments were divided according to the proportion of rock mineral wool, which was mixed with conventional growth media. We used plastic pots (15 × 15.5cm) with volumes of 3.5 litres. The experiments were allocated according to the coverage of substrates within the vegetation:

- Method 1: During the first experiment, the water retention capacity of the growth media was tested in pots without plants. The water retention capacities of the media were determined weekly using the classical approach based on weighing. The weighing of the substrates was always performed before watering.

- Method 2: During the second experiment, we tested the growth responses of the zonal pelargonium cultivar 'Tango® Dark Red' to different growth media with rock mineral wool.

Table 1: Ratio between rock mineral wool and industrial substrate per individual treatments

Treatment	Acronym	Ratio Rock mineral wool : substrate
1	4cm KV + 100% S	4cm rock mineral wool disc at the bottom of the pot + 100% conventional growth media
2	2cm KV + 100% S	2cm rock mineral wool disc at the bottom of the pot + 100% conventional growth media
3	100% S (control)	0% rock mineral wool + 100% conventional growth media (control treatment)
4	60% KV + 40% S	60% rock mineral wool + 40% conventional growth media
5	40% KV + 60% S	40% rock mineral wool + 60% conventional growth media
6	20% KV + 80% S	20% rock mineral wool + 80% conventional growth media

Watering of individual treatments was performed every week from the 5th of July until the 30th September 2013. Each pot was watered with 500ml of water. In order to determine the differences in the amounts of water retained within the various substrates after a prolonged deficit of water, we did not water the pots during the last four weeks of the experimental period. The experiment was designed as three replications. We prepared 10 pots for each treatment.

3. Results and discussion

3.1 Analysis of the quantity of retained water in substrates without the plant

Weekly dynamics of the residual retained water in the substrates is presented in Figure 2.The graph shows that the substrate with 60% of the rock mineral wool had the highest ability for retaining water during the whole trial period. Good results were also observed for those substrates with 40% of the rock mineral wool, whilst the substrates with lower content of rock mineral wool in the form of cubes (20%) were less valuable.

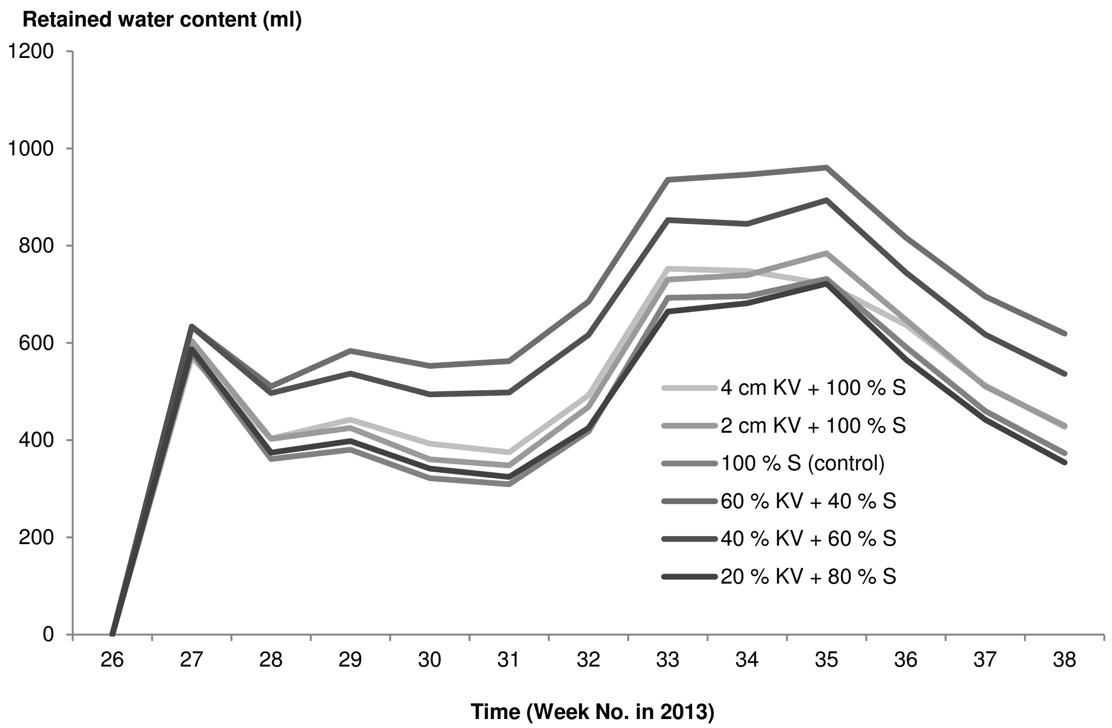

Figure 2: Retained water content (ml) in the tested substrates during the experimental time

At the end of the experimental period, the highest amount of retained water was measured in the substrate with 60% of rock mineral wool and was significantly higher in comparison with the substrate with 20% of rock mineral wool and the conventional growth media without the rock mineral wool (Table 2). The substrate with 60% of rock mineral wool retained 618.6ml of water, which represented 12.4% of the total added water. In addition, the quantity of water retained in the substrate with 40% mineral wool was significantly greater statistically in comparison with the substrate with 20% of rock mineral wool. 535.5ml of water was retained in the substrate with 40% of rock mineral wool, which represented 10.7% of the total added water. Substrate with rock mineral wool in the form of discs was not significantly different from the conventional growth media regarding water retention capacity, although the values for the amounts of retained water were higher.

Table 2: Retained water in individual substrates at the end of experimental period

Parameter	Treatments					
	4 cm KV + 100 % S	2 cm KV + 100 % S	100 % S (control)	60 % KV + 40 % S	40 % KV + 60 % S	20 % KV + 80 % S
RWS± sd (ml) [1]	426.7±147 bc	429.2±88 bc	372.6±60 c	618.6±43 a	535.5±34 ab	353.7±6 c
% RWS[2]	8.5	8.6	7.5	12.4	10.7	7.1

[1]RWS = retained water in the soil in millilitres ± sd = standard deviation; [2]% RWS = % of retention of water; [3]Statistic = treatments followed by the same letter are not significantly different (P < 0.05)

3.2 Impact of rock mineral wool on the morphological characteristics of zonal pelargoniums

The analysis of variance regarding morphological traits indicated that there were significant differences amongst the studied substrates (Table 3).

Table 3: Analysis of variances of the more important morphological traits of zonal pelargonium cv. 'Tango® Dark Red' used to differentiate between the studied treatments. Values represent means for traits ± standard deviation.

Treatment	Number of shoots	Number of inflorescences	FWS[2] (g)	FWR[3] (g)	DWS[4] (g)	DWR[5] (g)
4cm KV disc +100 % S	7.9±1.4a[1]	6.8±1.8 a	79.98±13.19 a	7.68±1.36 a	11.37±1.77 a	1.49±0.29 a
2cm KV disc +100 % S	7.9±1.1 a	6.8±1.7 a	79.60±12.55 ab	7.95±1.56 a	10.54±2.24 b	1.46±0.41 a
100 % S (Control)	7.1±1.4 a	6.1±1.7 ab	72.16±10.43 b	7.25±1.99 a	10.05±1.33 b	1.43±0.50 a
60 % KV+ 40 % S	7.2±1.4 a	6.6±1.8 ab	73.92±11.21 ab	7.50±1.55 a	9.82±1.42 b	1.59±0.45 a
40 % KV+ 60 % S	7.1±1.1 a	6.0±1.5 ab	76.42±10.62 ab	7.44±1.82 a	10.32±1.50 b	1.48±0.36 a
20 % KV+ 80 % S	7.8±1.5 a	4.8±2.3 b	76.48±13.50 ab	8.37±1.11 a	10.42±1.68 b	1.62±0.35 a

[1]Means (n=30) ± sd (standard deviation); a,b – identical letter in columns indicate no significant difference based on Tukey's multiple range test (P < 0.05); [2]FWS - fresh weight of shots; [3]FWR - fresh weight of roots; [4]DWS - dry weight of shots; [5]DWR - dry weight of roots; S = conventional growth media; KV = rock mineral wool (disc or cubes)

Analyses of the number of shoots between substrates indicated that the highest numbers of shoots were obtained in substrates with 2 and 4cm discs of rock mineral wool at the bottom of the pot (7.9 shoots per plant). In comparison with the plants that grew in the conventional growth media substrate, plants in substrate with 2 and 4cm discs of rock mineral wool formed 11.3% more shoots, however the differences amongst treatments were insignificant.

The number of inflorescences was significantly different according to individual treatments. Plants that were grown in the substrates with 2 and 4 cm rock mineral wool discs at the bottom of the pots, formed significantly more inflorescences (6.8 inflorescences per plant) than the plants grown in the substrate with 20% of the rock mineral wool in the form of cubes (4.8 inflorescences per plant). Plants that were grown within the conventional growth media without rock mineral wool had 10.3% less inflorescences compared to those plants grown in the substrate with 2 or 4cm discs of rock mineral wool.

Differences were observed between the fresh and dry weights of shoots. Plants that were grown in the substrates with 4cm discs of rock mineral wool at the bottom of the pots had significantly higher fresh and dry weights of shots per plant than those plants grown within the conventional growth media without rock mineral wool.

Analyses of root weight indicated that plants grown in the substrate with 20% of rock mineral wool obtained the highest fresh and dry root weights but the differences amongst treatments were insignificant. Plants grown within the conventional growth media without rock mineral wool were observed to have the lowest fresh and dry weights of roots.

4. Conclusion and outlook

The types and sizes of growth media components are crucial factors for water retention capacities (3). The results showed that rock mineral wool as a component in growth media can have a positive effect on water retention capacity and plant development. By improving the water retention capacity of the growth media, lower amounts of water for plant watering are needed. By increasing the volume fraction of the mineral component in growth media, the quantity of peat used as a primary ingredient can also be reduced. The reduction of water and peat may have positive effects on the economy and the ecological aspects of production.

References

[1] Robertson R.A. (1993): Peat, horticulture and environment, Biodiversity & Conservation 2(5), 541-547

[2] Blievernicht, A.; Irrgang S.; Zander M.; Ulrichs C. (2011): Produktion von Torfmoosen (*Sphagnum* sp.) als Torfersatz im Erwerbsgartenbau, Gesunde pflanzen 62(3-4) 125- 131

[3] Handreck K.A.; Black N.D. (2010): Growing Media for Ornamental Plants and Turf (4th ed.), UNSW Press, Australia

Thematic Area TC – Advances in emission prevention (Oral presentations)

Christiane Lüdtke: Art from Tetrapak

TC-K The ALFAM2 project – Predicting ammonia loss from field-applied manure

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1. Objectives

The overall goal of this project is to develop tools for predicting ammonia emission from field-applied manure. Specific objective are to:

1. Construct a database of ammonia emission measurements,
2. Quantify relationships between management practices, environment, manure characteristics, and emission,
3. Develop a mathematical model for predicting ammonia emission,
4. Create software tools for predicting ammonia emission

In this paper, we focus on 1 and 2.

2. Methodology

Ammonia emission measurements and associated weather, manure, and application data were solicited from researchers in Europe and North America. The response variable was average emission rate over a single interval within a sequence of measurements (Table 1). Two possible observational units are present in these data: plot (the physical location which received a particular type of manure under some known conditions) and measurement interval (a period of time over which emission was measured for a single plot). In the database, each unique plot is associated with multiple measurement intervals.

Table 1: A summary of the types of data included in the database.

Identification	Location	Soil	Weather	Slurry	Application	Crop	Emission
Project	Latitude	Texture	Temperature	Type	Timing	Type	Interval time
Publication	Longitude	Moisture	Radiation	Bedding	Method	Height	Method
Experiment	Topography	pH	Wind speed	Dry matter	Rate	Coverage	Plot size
Field code		Tillage	Rain	Total N	Incorporation		Background
Plot code		Temperature	Humidity	Ammonia			Emission
Treatment				pH			

Notes: Identification, location, soil, slurry, application, and crop data were collected on a plot basis. Weather and emission were collected on an interval (shift) basis. Not all observations included data for all variables.

Emission in each measurement interval was calculated by summing the product of reported average flux (kg N/ha-hr) and interval duration (hr). Cumulative emission was the sum of prior emission. Relative emission and flux were calculated by dividing by the total ammonia nitrogen (TAN) applied in slurry (kg N/ha). Cumulative emission was estimated at regular durations (e.g., 48 hr) by cubic interpolation.

3. Results and discussion

3.1 Database overview

Researchers from ten countries contributed data, which were combined with an earlier database (ALFAM [1]). In total, 20 institutes from 12 countries contributed >30000 observations, from >1700 plots and 300 experiments. Most plots received either cattle or pig slurry by trail hose application, and other types are not discussed here (Table 2). Grass was the most common crop, followed by no crop (bare soil). Measurements are available for all combinations of manure type and application method for grass only, yet even within this subset observations are strongly unbalanced, with cattle manure (particularly broadcast application) best represented (Table 2).

Table 2: Plot counts (number of separate plots in database) for each combination of manure type, application method, and crop.

Manure type	Application method	Crop type Grass	Bare soil	Cereal	Maize	Stubble	Other
Cattle	Broadcast	446	58	0	0	19	13
Cattle	Trailing shoe	119	0	0	0	0	7
Cattle	Trailing hose[a]	105	55	27	25	9	11
Cattle	Closed slot	4	2	0	0	0	0
Cattle	Open slot	109	1	0	0	0	0
Cattle	Pressurized injection	2	1	0	0	0	0
Pig	Broadcast	90	95	0	0	115	9
Pig	Trailing shoe	5	0	0	0	0	0
Pig	Trailing hose[a]	20	22	70	16	11	48
Pig	Closed slot	2	5	14	0	4	0
Pig	Open slot	3	6	6	0	5	0

[a] Included band spreading.

Reported emission rates varied by >6 orders of magnitude among plots and over time. Emission rate generally showed a rapid decline, approximately first-order, in the first day and a slower decline after (Fig. 1). Apparent first-order rate constants from the first day of emission were almost all negative (96%) with a median of −1.3 d^{-1}, indicating a > 10-fold decline in flux over the first day on average. Two day cumulative emission ranged from near zero to much greater than applied TAN (>6-fold). However, only 35 plots (2.4%) showed cumulative emission greater than applied TAN.

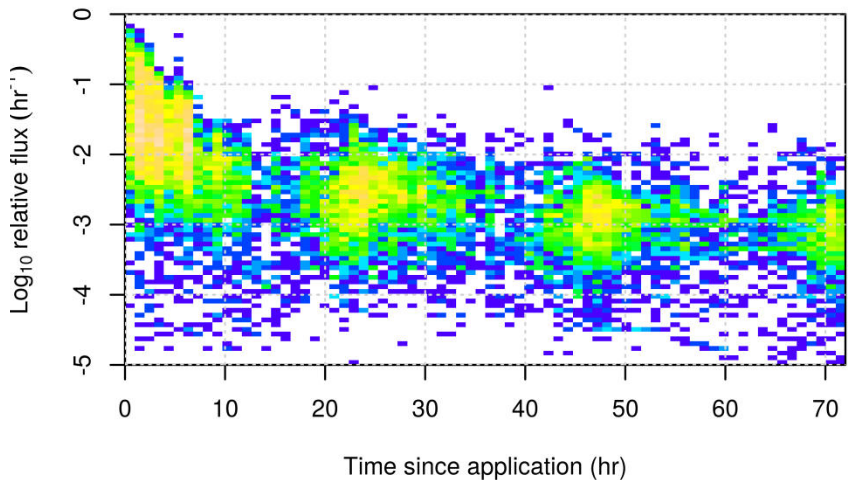

Figure 1: All observations of ammonia emission rate (flux) in the complete database, normalized by total ammonia application rate. The color of each cell is related to the number of observations within it, with blue colors having the least (≥1), and yellow and tan having the greatest (≤63).

3.2 Effects on ammonia emission from manure applied to grass

Data on application of cattle and pig manure to grass were used to demonstrate the utility of the database and to quantify effects of management and weather on ammonia emission. This subset was selected because grass was the only crop with observations for all application methods for both cattle and pig slurry. A backward selection procedure was used to select model variables. The starting model included the following predictors: institute, application method, manure type, manure dry matter (DM), manure pH, grass height, air temperature, wind speed, and solar radiation, with no interactions. Weather data were averaged over the first 24 hr following application.

Relative cumulative emission at two days (48 hours) was selected as the response variable. This duration was long enough for most emission to occur (median emission was 89% of final value, and 82% of plots had reached 80% of final emission by 48 hr) but avoided eliminating many plots with measurement periods <3 days. Multiple linear regression was used with the lm function in R

for model fitting, and predictor significance was assessed using single term deletion tests with the drop1 function [2].
Selection of an appropriate transformation for emission is challenging. Relative cumulative emission is generally bounded (0-1) and effects are not additive, at least near the extremes. Here, we used a new approach which may prove useful: a logit transformation. The transformed response variable was the log odds of emission (or retention) of a single ammonia molecule, where odds is defined as the ratio of probability of emission to probability of retention. Application of this model therefore assumes that predictors have fixed effects on the odds of emission, regardless of the total amount. Additional work will be needed to more completely evaluate the approach and compare it to alternatives.

Emission appeared to be strongly dependent on application method and slurry type (Table 3). All application methods designed to reduce emission appear effective, with trailing hose and trailing shoe providing similar reductions (≥70%, or an odds reduction ≥85%) and open slot injection providing a larger reduction. In a literature review, Webb et al. [3] found similar a similar average reduction for trailing shoe (65%), but smaller reductions for trailing hose (35%) and open slot injection (80%). Pig manure had lower emission than cattle, even when adjusted for DM. This response has been observed by others [1, 4], although both of these studies were based on data now included in the complete ALFAM database.

Wind speed and air temperature had clear positive effects on emission overall. Temperature will affect emission through an effect on the activity of free ammonia (NH_3 (aq)), and wind affects transfer from the manure surface to the atmosphere. Considering typical variation in weather (in the data used for model construction, standard deviation was 5.8°C and 1.7 m s^{-1}), the importance of wind was slightly larger than temperature. Conversely, there was no clear effect of solar radiation. Emission also appeared to be related to DM, with a 1% increase in DM associated with a 10% increase (odds increase of 22%). Effects of wind, temperature, and DM on ammonia emission have been observed in some previous studies (e.g., [5, 6]).

Table 3: Apparent effects of predictors on two day cumulative relative ammonia emission from manure applied to grass, based on linear regression with a logit transformed response variable and inclusion of institute as a predictor.

Term	Relative effect (%)	Estimate	Standard error	*t* value	*P*
Trailing shoe	-74	-1.9	0.23	-8.4	<0.0001
Trailing hose[a]	-70	-1.7	0.24	-7.3	<0.0001
Open slot	-92	-3.2	0.2	-16	<0.0001
Pig	-32	-0.67	0.22	-3.1	0.0021
Dry matter (%)	9.8	0.20	0.034	5.7	<0.0001
pH	20	0.40	0.18	2.2	0.025
Air temperature (°C)	1.4	0.027	0.0097	2.8	0.0051
Wind speed (m s^{-1})	10	0.2	0.039	5.1	<0.0001

Values in estimate and standard error columns are directly from summary.lm ouput, and are in transformed units, i.e., they represent the effect of the predictor level (or one unit increase in the predictor) on the log odds of ammonia emission. The reference scenario was broadcast-applied cattle manure. The relative effect column shows the effect on emission as a fraction of emitted ammonia. This value depends on emission in the reference scenario, which was set at 50% of applied total ammonia nitrogen for this calculation. Model R^2 was 0.67.

The initial and final models included a factor for institute, i.e., each individual institute was assumed to have a fixed effect on the odds of ammonia emission. Lumped into the levels of this factor are variables not included in the model (and possibly not included in the database), as well as any biases inherent in, e.g., manure application or emission measurement methods. For example, soil properties have been shown to influence ammonia emission, but were not included in the model. A model that includes all important predictors, and is based on data without significant biases would show no institute effects, but here institute is a highly significant predictor ($P < 10^{-15}$). Apparent institute effects are shown in Fig. 2. Correction for predictors shown in Table 3 does reduce apparent differences between institutes, but the remaining effects complicate development of a universal model for predicting ammonia emission. A better understanding of the causes of these differences among institutes would be useful.

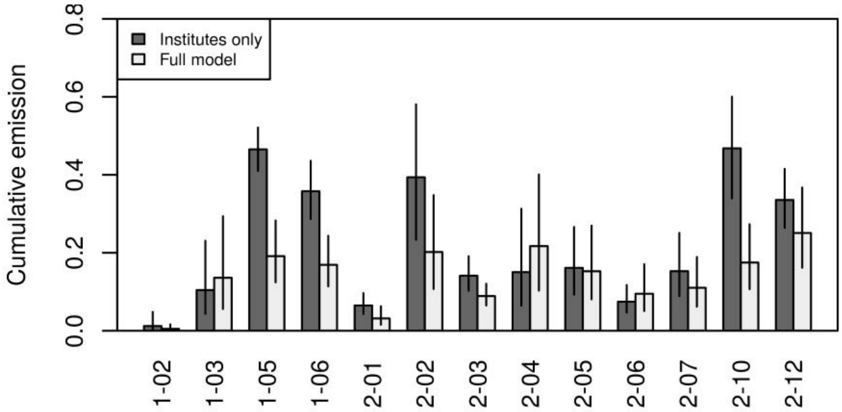

Figure 2: Apparent effects of institutes on measured ammonia emission from cattle and pig manure applied to grass. All values are model predictions and 95% confidence intervals, and x axis categories are institutes included in the model. The “institutes only” model is a single-factor analysis of variance model, and plotted values are identical to institute means (with no correction for application method or any other possible predictor). The “full model” values are based on the model described above for trailing shoe application of mean cattle manure under mean weather conditions.

4 Conclusion and outlook

The database summarized here is based on data collected under a wide range of conditions and locations. Once complete, the database will be made publicly available online, where it may be useful for multiple purposes, including evaluation or determination of emission factors for particular conditions. Future work includes development of a mathematical model and software tools for predicting ammonia emission.

Acknowledgements

Data were originally collected by individuals at the following institutes and were generously contributed to the ALFAM2 project: Agriculture and Agri-Food Canada, ADAS, Rothamsted Research, Aarhus University, AgroTech, Leuphana University, Bern University of Applied Sciences, Milan University, the French National Institute for Agricultural Research, the Irish Agriculture and Food Development Authority, the United States Department of Agriculture-Agricultural Research Service, and Wageningen University. In particular, the help of Martin Chantigny, Shabtai Bittman, Derek Hunt, Tavs Nyord, Tom Misselbrook, M. Bowden, Martin Hansen, Christoph Häni, Jörg Sintermann, Albretch Neftel, Thomas Kupper, Polina Voylokov, Benjamin Loubet, Marco Carozzi, Gary Lanigan, Jack Mesinger, Bert Vermeulen, and Jan Huijsmans in organizing data and contributing ideas is greatly appreciated.

References

[1] Søgaard, HT, Sommer SG, Hutchings NJ, Huijsmans JFM, Bussink DW, Nicholson F. 2002. Ammonia volatilization from field-applied animal slurry—the ALFAM model. Atmospheric Environment 36 (20): 3309–3319.

[2] R Core Team (2015). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computting, Vienna, Austria. http://www.R-project.org/

[3] Webb, J., Brian Pain, Shabtai Bittman, and John Morgan 2010 The Impacts of Manure Application Methods on Emissions of Ammonia, Nitrous Oxide and on Crop Response-A Review. Agriculture Ecosystems & Environment 137(1-2): 39–46.

[4] Ni K., Pacholski, A., Dirk Gericke, D. and Kage H. (2012): Analysis of ammonia losses after field application of biogas slurries by an empirical model, Journal of Plant Nutrition and Soil Science, 175(2): 253-264.

[5] Misselbrook, T. H., F. A. Nicholson, and B. J. Chambers 2005 Predicting Ammonia Losses Following the Application of Livestock Manure to Land. Bioresource Technology 96(2): 159–168.

[6] Bussink, D. W., Huijsmans, J.F.M., and Ketelaars, J.J.M.H. 1994. Ammonia volatilization from nitric-acid-treated cattle slurry surface applied to grassland. NJAS Wageningen Journal of Life Sciences 42(4): 293–309.

TC-O_01 Ammonia emission after slurry application to grassland

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1. Objectives

Ammonia (NH_3) losses after field application of slurry from livestock is a key loss path contributing between 30 and 50% to the agricultural ammonia emissions released by European countries [1]. The main objective of this study was to quantify NH_3 emission levels for the application of slurry to grassland and to evaluate the effects of the abatement techniques trailing hose, trailing shoe and shallow injection compared to the reference technique splash plate. Additionally, the emissions from different types of slurries and the influence of the timing of the application on the level of the losses were investigated.

2. Methodology

Slurry was applied within 17 field experiments (see [2]) on square plots of approx. 900m^2 using the reference technique splash plate and the low emission techniques trailing hose, trailing shoe and shallow injection. The slurries applied were obtained from dairy cattle and fattening pigs or breeding pigs. Up to four experimental plots were operated in parallel in order to compare the emissions from several treatments under identical conditions. The layout defined for the experiments aimed at minimizing interactions between the individual plots with respect to the emissions.

NH_3 concentrations were measured in the center of the fertilized area at approx. 1m using impinger sampling devices and upwind of the fertilized area for the inflow concentration. The corresponding emission rates were estimated with a backward Lagrangian stochastic model (WindTrax) [3], generating the model input from sonic anemometer measurements.

Before and after each application, a sample of the slurry was taken from the tank and stored at 4°C for the analysis of the total nitrogen (N_{tot}), total ammoniacal nitrogen (TAN) and dry matter content (DM) and the slurry bulk pH. The meteorological parameters were retrieved from the closest available meteorological station.

Linear regression was done using the least squares method.

3. Results and discussion

3.1 NH_3 emission level and mitigation due to abatement techniques

The average NH_3 emission of cattle slurry exhibiting a mean dry matter content of 3.3% (range between 1.0% and 6.7%) applied with the reference technique splash plate amounted to 25% of TAN applied. The emission levels obtained in individual experiments ranged from 10% to 47% of TAN. Figure 1 shows the cumulative emissions versus time of the year of application, stratified by DM content and application technique. A dependency on the slurry DM content is apparent. Slurries with a high DM content and application during the warm season tend to produce higher losses than slurries with a low DM content and application during the cool season, respectively.

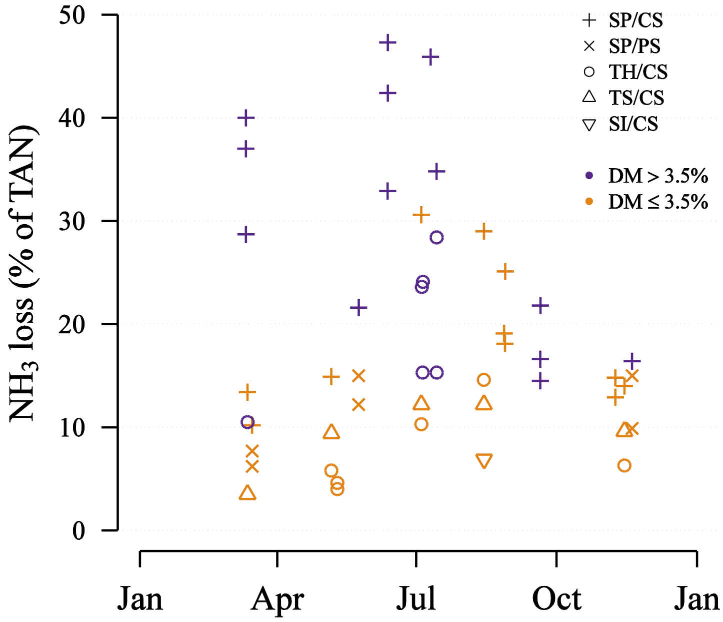

Figure 1: Measured cumulative NH_3 losses as percentage of the applied TAN arranged to the time of application over the year and stratified according to the DM content of the slurry: low ≤ 3.5% DM, high > 3.5% DM. Application techniques: SP = splash plate, TH = trailing hose, TS = trailing shoe, SI = shallow injection. Slurry types: CS: cattle slurry, PS: pig slurry.

The use of low emission techniques showed a clear mitigation effect. An average reduction of approx. 50% for the application with trailing hose and trailing shoe and approx. 75% for shallow injection was found, compared to the reference technique splash plate (Table 1). This is in agreement with the ranges given by [4].

Table 1: Abatement effect from emission reducing application techniques. n: number of observations.

Application Technique	Reference	n	Average reduction	Range
Trailing hose	Splash plate	7	51%	[22%, 68%]
Trailing shoe	Splash plate	5	53%	[36%, 71%]
Shallow injection	Splash plate	1	76%	-

Three field experiments compared cattle slurry to pig slurries. The latter revealed emission levels being systematically lower by approx. 30% as compared to cattle slurry. The reason for this difference could not be completely allocated by the analyzed slurry properties, even though the lower DM content of pig slurry may partly account for the difference in the measurements.
Application around noon produced higher emissions than in the morning and in the evening. This is in line with the findings of previous studies such as [5]. This result can be related to the much higher NH_3 emission potential at noon and the high proportion of the total loss emitted during the first few hours after application, which was found in all experiments.

3.2 Modeled NH_3 emissions

A regression analysis was carried out on the cumulative emissions after 24h, based on the subset splash plate cattle slurry. The results of the regression analysis indicate that the ambient air temperature (TAir) and the slurry DM content are the main drivers in explaining the variance in the cumulative emission. The final model (R^2 = 0.76) is given as

$$Loss24h = \exp(1.83 \pm 0.16 + (0.033 \pm 0.010) * TAir + (0.18 \pm 0.04) * DM) \qquad \text{Eq. 1}$$

where Loss24h represents the cumulative emission after 24h in % of TAN, TAir the ambient air temperature in °C and DM the slurry dry matter content in %. TAir represents an averaged value over 24 h, using a weighted average that accounted for the typical, observed decay in the emis-

sion rates. This decay was approximated based on a Michaelis-Menten type equation [6] with Km equal to 90 min, corresponding to the median Km value found for the present study. This weighted averaging guaranteed an improved representation of TAir under a temporal variation. It increases the weight for TAir in the initial period, where the major proportion of the total NH_3 loss occurs.
The effect of the air temperature can mainly be related to the correlation between surface and slurry temperatures that control the chemical and physical equilibria of the slurry TAN (e.g. [7]). Systematic measurements using wind tunnels by [8] confirm such an air temperature dependence of the NH_3 loss as well. The DM content of the slurry increases the emissions in several possible ways (see e.g. [7]). Similar to this study, wind tunnel measurements (e.g. [9]) showed a pronounced effect of increasing DM contents resulting in greater NH_3 losses. Figure 2 illustrates the dependence of the NH_3 loss after 24 h from the (weighted) air temperature and the slurry DM content as modelled by equation (1) and as observed during the individual emission measurements.

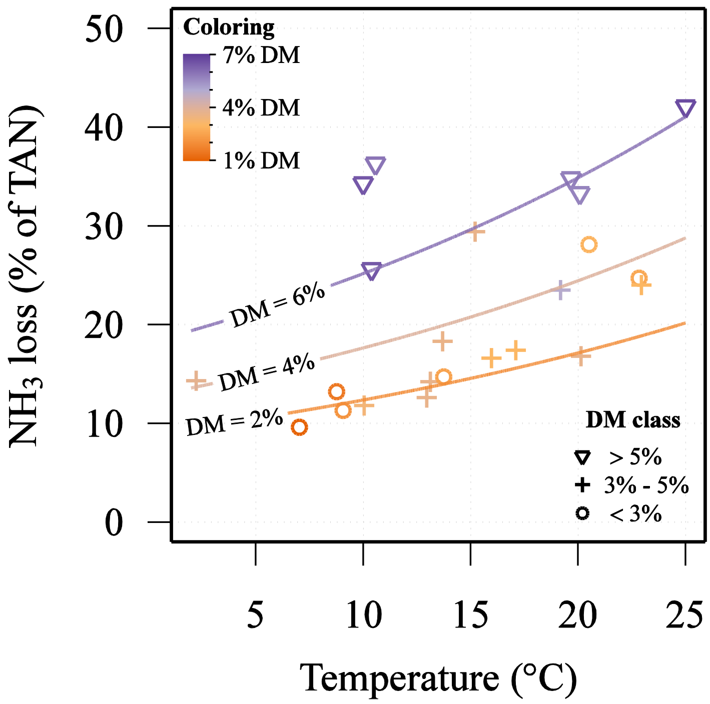

Figure 2: NH_3 loss after 24 h arranged by (weighted) air temperature and DM content. Lines: modelled loss; Symbols: observed losses.

4. Conclusion

The NH_3 emission levels for the reference technique splash plate were found in between 10% to 47% of TAN. The variation of the emission levels could be attributed to a large part to the ambient air temperature and the dry matter of the slurry. The dependence of the cumulated losses from these parameters is very similar to previously reported dependencies. The different abatement techniques significantly reduced the emission level compared to the application using a splash plate. The emission mitigation achieved was in the expected range.

Acknowledgements

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References

[1] Sintermann, J., Neftel, A., Ammann, C., Häni, C., Hensen, A., Loubet, B., Flechard, C.R., 2012. Are ammonia emissions from field-applied slurry substantially over-estimated in European emission inventories? Biogeosciences 9 (5), 1611-1632. 10.5194/bg-9-1611-2012.

[2] Häni, C., Sintermann, J., Kupper, T., Jocher M., Neftel, A. Ammonia emission after slurry application to grassland. submitted to Atmos. Environ.

[3] Flesch, T.K., Wilson, J.D., Harper, L.A., Crenna, B.P., Sharpe, R.R., 2004. Deducing ground-to-air emissions from observed trace gas concentrations: A field trial. J. Appl. Meteorol. 43 (3), 487-502. 10.1175/1520-0450(2004)043<0487:dgefot>2.0.co;2.

[4] Webb, J., Pain, B.F., Bittman, S., Morgan, J., 2010. The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response-A review. Agric. Ecosyst. Environ. 137 (1-2), 39-46. 10.1016/j.agee.2010.01.001.

[5] Sommer, S.G., Olesen, J.E., 2000. Modelling ammonia volatilization from animal slurry applied with trail hoses to cereals. Atmos. Environ. 34 (15), 2361-2372. 10.1016/s1352-2310(99)00442-2.

[6] Søgaard, H.T., Sommer, S.G., Hutchings, N.J., Huijsmans, J.F.M., Bussink, D.W., Nicholson, F., 2002. Ammonia volatilization from field-applied animal slurry - the ALFAM model. Atmos. Environ. 36 (20), 3309-3319. 10.1016/s1352-2310(02)00300-x.

[7] Sommer, S.G., Génermont, S., Cellier, P., Hutchings, N.J., Olesen, J.E., Morvan, T., 2003. Processes controlling ammonia emission from livestock slurry in the field. Eur. J. Agron. 19 (4), 465-486. 10.1016/s1161-0301(03)00037-6.

[8] Sommer, S.G., Olesen, J.E., Christensen, B.T., 1991. Effects of temperature, wind-speed and air humidity on ammonia volatilization from surface applied cattle slurry. J. Agric. Sci. 117, 91-100.

[9] Sommer, S.G., Olesen, J.E., 1991. Effects of dry-matter content and temperature on ammonia loss from surface-applied cattle slurry. J. Environ. Qual. 20 (3), 679-683.

TC-O_02 The importance of pH for ammonia emission from animal manure

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1. Objectives

This proceeding presents important pH-related reactions that affect ammonia (NH_3) emission. There is a shortage of models that give the relationship between NH_3 emissions from stored and applied slurry to both pH and total ammoniacal nitrogen (TAN) concentration at the surface of the source, although these variables must be known to give accurate estimates of NH_3 emissions. Assessment of NH_3 emission can be improved by including pH changes in slurry with a focus on the changes in the surface layer as affected by physical, chemical, and biochemical processes.

2. Methodology

pH is an important component in controlling ammonia (NH_3) release from slurry to the atmosphere, and except models based on emission factors the ammonia emission models include this variable in the calculations [1] . These models generally calculate emission rate using bulk pH of slurry in stores or in tankers spreading slurry [1]. In some, a constant is added to the measured pH, because it is known that pH at the surface may increase significantly from this bulk pH.
The intent of this article is to present results showing how slurry pH may vary as affected by transport of pH buffer components, transformation of organic components and also as affected by content of di- and monovalent ions. The compiled information is used to document the need to include these in the calculations of pH in models for predicting emission of NH_3, and as a first step concepts for developing physical-chemical-biochemical conceptual models of pH changes in slurry and in the surface of stored or applied slurry are provided. The next generation NH_3 emission models may use these relationships to accurately predict pH and more accurately predict NH_3 emission from slurry during storage and from slurry applied to fields.

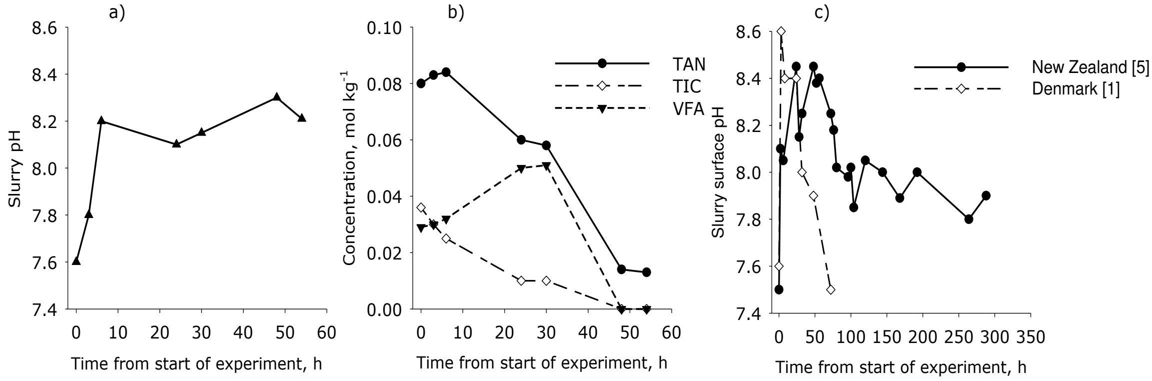
3. Results and discussion

It is the uncharged species NH_3(aq) (free ammonia) that can volatilise, and it is a base, so its concentration and emission rate are dependent on slurry pH. After TAN concentration in the source, pH is an important factor in controlling NH_3 release to the atmosphere. Emission occurs from the surface, and so it is surface pH that affects NH_3 emission. It has been shown that pH is related to the concentration of pH buffers, including total inorganic carbon (TIC=CO_2 + HCO_3^- + CO_3^{-2}), volatile fatty acids (VFA), TAN and organic particles. TIC, VFA and TAN are volatile and the emission of these to the atmosphere will affect pH [2, 3]. Calculations of pH in the surface of a source must, therefore, include convective and diffusive transport of TAN and pH buffer components as well as emission of CO_2, NH_3 and VFA. In addition the pH models must account for aerobic transformation of the components at the surface and anaerobic transformation in bulk slurry.

3.1 Transport and emission of pH buffer components

The pH at the surface of the stored slurry source is affected by transport of pH buffer components in the liquid and release of these to the atmosphere. The pH of most untreated slurry from pigs and cattle is above 6.5 [1] and at this pH the two most volatile organic components are CO_2 and NH_3. The volatility of uncharged hydrogen sulphide (H_2S) and organic acids is negligible. After mixing, release of CO_2 is about 2000 times the release of NH_3 due to the lower solubility of CO_2 than of NH_3. Thereafter CO_2 emission decreases due to depletion of TIC at the surface. The resulting elevated pH will enhance emission of NH_3, which with time will cause a reduction in surface pH, although the reduction in NH_3 emissions is also a result of depletion of TAN in the surface, as transport in the liquid is not as fast as transport in the air. This example shows that the bulk pH or pH at the start of an emission process cannot be used for assessment of the emissions from slurry during an NH_3 emission event. Instead, pH in the surface layers must be assessed.

The influence of surface processes on NH_3 emission may be illustrated by gaseous losses from a thin layer of slurry on a soiled floor or on soil, where the effect of convective transport of buffer components from layers below can be excluded. Thus pH may increase by more than 0.5 units during the first 6 hours after application of slurry due to emission of CO_2 which reduces the concentration of TIC (Fig. 1a,b). The pH increase induces a high rate of NH_3 volatilisation after application, which may cause pH to decline. The data presented in figure 1a,b is from a study where pig slurry was applied to a petri dish so slurry was not mixed with soil, while in figure 1c it is seen



that pH declines more when slurry is applied onto soil in a field due to the effect of soil pH buffering capacity.

Figure 1: a) and b) pH and b) concentration of the pH buffer components TAN, TIC and VFA in a 7 mm layer of pig slurry incubated at 16°C (data from [4]), and c) pH in the surface of slurry soil mixture.[1,5].

In stored slurry the processes mentioned above affect surface pH, and convective and diffusive transport of pH buffer components to the surface also have an effect [3]. In slurry-soil mixtures convective transport may be upwards during drying and downwards after rain. In most systems with stored slurry mass transfer will be upwards. In models known to us, it is proposed that there will be a thin still surface layer (microns to millimetres in thickness) in stored slurry below which there is convection. Diffusion will dominate in the still surface layer and below this layer the transport will be dominated by convection. In this system there will be a steep pH gradient in the still surface layer and below this layer pH will be relative constant with depth.

Conceptually the influence of these volatile components are known, but to our knowledge there is no model that precisely predict the pH changes as affected by emission of these, that can be used in decision support. So there is a challenge in developing pH models that can be used for calculating NH_3 emission from stored slurry and from slurry applied in the field.

3.2 pH as affected by organic acids

pH in slurry is affected by the concentration of organic acids that may have been produced in the intestine of the livestock or is produced afterwards in animal slurry [1]. The organic acids originate from anaerobic microbial decomposition of organic material, producing C2 to C5 acids. Generally more than 60% of the organic acid in slurries consists of acetic acid (CH_4COOH). Therefore, the pK_a of acetic acid is often used to express VFA buffering of slurry.

The production of organic acids from more complex organic components will reduce slurry pH (Figure 2). Anaerobic conditions prevail in the bulk of slurry which is crucial for the production of the acids that reduce pH (Figure 2b). Good conditions for VFA production include in addition to an anaerobic environment high concentration of acidogenic microorganisms adapted to the environment, high concentrations of digestible organic matter and high temperature [1,6]. These acids are intermediates in the anaerobic production of methane (CH_4) and CO_2, thus, if the organic matter feeding the micro-organisms is consumed then production of acids will be reduced and fermentation will gradually reduce the concentration of acids in the slurry and pH will increase with time (Figure 2b). This increase in pH can be reversed by addition of easily digestible organic matter, which will enhance acid production and reduce pH (Figure 2a). Consequently, addition of easily digestible organic matter in the form of sucrose to stored slurry have been used to increase VFA production and reduced pH to below 5, thereby significantly reduced NH_3 emissions from stored and field-applied slurry [6]. The problem with this treatment is that the organic acids are a substrate for CH_4 production, which is a very potent greenhouse gas. In biogas production the rate of transformation of organic acids to CH_4 is enhanced, therefore, most slurries that have been anaerobically digested have a pH higher than 8.

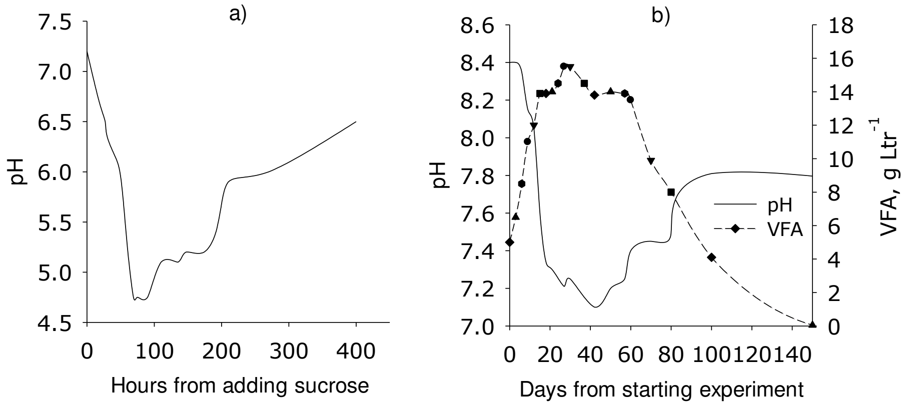


Figure 2: a) pH after adding sucrose to the slurry (Adapted from [6]) and b) pH and concentration of VFA in the bulk of untreated slurry stored at 20°C (Adapted from [7]

Oxygen will diffuse into the surface layers of stored slurry and slurry-soil mixture, and will provide an aerobic environment where aerobic micro-organisms will transform the organic acids to carbon dioxide (CO_2), which will contribute to increase in surface pH. Thus in the surface of uncovered slurry the reduction in VFA concentration may contribute to increase surface pH of stored slurry. In contrast, surface covers of stored slurry in the form of straw reduce liquid surface pH due to the production of organic acids and probably also due to a reduction in release of CO_2. The oxidation of VFA by aerobic microorganisms in the surface of slurry applied to soil will contribute to the increase in slurry soil surface pH within the first week. This effect will interact with the CO_2 produced being dissolved in the liquid and then released to the atmosphere. The net effect of oxidation of VFA and CO_2 release is an initial increase in pH in the surface of soil slurry mixtures in the first 1-3 days after slurry application and then volatilisation of NH_3 will cause a reduction in pH.
The development of models that include pH calculations as affected by the organic acids and microbial production and transformation of these will be a most challenging and important task.

3.3 pH buffering by organic particles

Weak acid functional groups dissociating at pH >7 on particulate organic matter contribute to the pH buffering capacity of slurry. These groups will contribute to alkalinity of the slurry, i.e., how much acid is required to reduce pH with one unit. This alkalinity has been shown to be linearly related to dry matter concentration (g kg^{-1}(DM)):

$$\mathrm{TAL} = \mathrm{K} \cdot \mathrm{DM} \quad \text{Eq. 1}$$

where total alkalinity (TAL) is in mol L^{-1} and DM in g kg^{-1}(DM and potassium (K) is 0.013 mol L^{-1} kg^{-1}. There is undoubtedly a need to provide a much better assessment of the acid-base properties of the organic particles of animal manure, because these components have an effect on the pH of the solution. Ammonium may also bind to the negatively charged organic matter particles (carboxylic and phenolic functional groups), which reduces the concentration of NH_3 in solution. Thus, the effect of negatively charged particles needs to be included in the pH calculation.

3.4 effect of dissolved ions

Slurry pH is also affected by the monovalent ions sodium (Na^+), potassium (K^+) and chloride (Cl^-) and dissolved divalent ions calcium (Ca^{2+}), magnesium (Mg^{2+}), sulphate (SO_4^{2-}), sulphide (S^{2-}) and hydrogen phosphate (HPO_4^{2-}). The ions originate from the feed given to the livestock and slurry pH is, therefore, affected by the feed, in particular the intake and excretion of monovalent ions. The content of these ions in feed is defined by animal physiologist as dietary electrolyte balance, dEB = $Na^+ + K^+ - Cl^-$ (meq kg^{-1}(DM)). pH decreases with decreasing dEB due to excretion of H^+, which compensates for low Na+ and K+ in relation to Cl- concentration. This response is a reflection of the electroneutrality condition. Similar models are used by soil chemists when estimating the change in soil pH in the rhizosphere as affected by plant root exudation of ions [1]. pH in slurry

may be reduced by more than 1 pH unit due to a reduction in K^+ caused by reducing soya and tapioca in feed to pigs, and supplementing the feed with essential amino acids.
Addition of soluble calcium (Ca^{2+}) salts will change the pH through precipitation of $CaCO_3$, which consumes alkalinity. An example of this process is a reduction in pH due to the addition of $CaCl_2$ to the slurry, which has been shown to reduce NH_3 volatilisation. Alternatively, replacing the Ca-CO_3 in pig diets with calcium dichloride ($CaCl_2$) may lower pH and thereby contribute to a reduction in NH_3 emissions due to the following reactions:

$$2\mathrm{HCO_3^-(aq)} + \mathrm{Ca^{2+}(aq)} \rightleftharpoons \mathrm{CaCO_3(s)} + \mathrm{HCO_3^-(aq)} + \mathrm{H^+(aq)} \qquad \text{Eq. 2}$$

$$\mathrm{HCO_3^-(aq)} + \mathrm{H^+(aq)} \rightleftharpoons \mathrm{CO_2(aq)} \uparrow + \mathrm{H_2O(l)} \qquad \text{Eq. 3}$$

The consequence of the two coupled reactions is that there is a limit for the reduction in pH, which is where the solid-liquid equilibrium between $CaCO_3$ crystal and $CaCO_3$ solute in solution is reached. Thus, these additives are not as effective in reducing pH as the inorganic acids H_2SO_4 or HNO_3.
In manure pH sub-models the effect of excretion of ions and calculation of the sum of the charge of ions as well as crystal formation or dissolution must be included.

4. Conclusion and outlook

It can be concluded that bulk slurry pH is not a suitable indicator of NH_3 emission potential of a stored slurry or slurry applied in the field, and because pH is important for the assessment of NH_3 emissions, there is a need for developing models for predicting pH. These models must take the most important buffers into account when predicting the variation in surface pH over time, and it is proposed that the model include the effect of 1) volatilisation of pH buffers, 2) microbial transformation of organic matter to acids and degradation of organic acids to CO_2 and CH_4, 3) dissolved mono- and divalent ions, 4) formation and dissolution of solid phase alkalinity and 5) acid-base property of organic particles.

References

[1] Sommer S.G. 2013. Ammonia volatilization from livestock slurries and mineral fertilizers. Dr. Techniques thesis published by University of Southern Denmark, Odense Denmark. Pp128.
[2] Petersen, V.; Markfoged, R.; Hafner, S.D.; Sommer, S.G. (2014). A new slurry pH model accounting for effects of ammonia and carbon dioxide volatilization on solution speciation, Nutrient Cycling in Agroecosystems. 100, 189-204
[3] Hafner, S.D.; Montes, F.; Rotz, C.A. (2013). The role of carbon dioxide in emission of ammonia from manure. Atmospheric Environment. 66. 63-71
[4] Sommer S.G. and Sherlock R.R. 1996. pH and buffer components dynamics in the surface layers of animal slurry. Journal of Agricultural Science, 127, 109-116.
[5] Sherlock R.R. 2002. pH and buffer components dynamics in the surface layers of animal slurry. Journal of Agricultural Science, 127, 109-116.
[6] Clemens J., Bergmann S. and Vandre R. 2002. Reduced ammonia emissions from slurry after self-acidification with organic supplements, Environ. Techn. 23, 429-435.
[7] Møller H.B., Sommer S.G., and Ahring B.K. 2004. Biological degradation and greenhouse gas emissions during pre-storage of liquid animal manure. J. Environ. Qual. 33, 27–36.

TC-O_03 Impact of field acidification and application methods on ammonia emissions, yield and nitrogen efficiency of organic liquid manures

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1. Objectives

The effectiveness of slurry application methods like band spreading, incorporation and injection combined with acidification on reducing ammonia (NH3) emissions and the relationship between related NH3 emission and crop yield are variable. Simultaneous and replicated field testing of slurry types, application and slurry treatment methods are needed for accurate evaluation of differences among application methods and effects of other factors, such as weather. A novel measurement approach was employed to obtain an extensive data set for a better understanding of emission patterns to ensure emission prevention. The objective of this study was to investigate effects of slurry type, application and treatment methods on NH3 emissions and crop production managed under the same agro-ecological conditions.

2. Methodology

Different slurry application methods and acidification of various slurry types were tested in four field experiments using a randomized block design (n = 4): two applications to spring barley, (Hordeum vulgare L.) and two applications to winter wheat (Triticum aestivum L.). Within each experiment, three to ten treatments were tested concurrently, consisting of different application methods: band spreading (BS), closed slot injection (IJ), incorporation with rotary cultivator (IN) and five slurry types: dairy cattle (CS), pig (PS) and mink slurry (MS); co-fermented anaerobic digestate (AD) and mechanically separated AD (AS). Each slurry type was evaluated with and without acidification (AC) using sulphuric acid (50 %) (Table 1).

Table 1: Slurry types, slurry characteristics and application dates, methods and conditions of four slurry trials carried out on spring barley and winter wheat at Foulum Research Centre, Denmark (2014)

Experiment	Slurry type	Applic. method	Rate	pH		H_2SO_4 (50%)	NH_4^+-N	DM -acid	DM +acid
			t ha^{-1}	-acid	+acid*	l m^{-3}	kg ha^{-1}	%	%
Application at seeding of spring barley 1/4/14 (soil moisture wet)	CS	BS	53				84.8	6.5	
	CS	IN	53				84.8	6.5	
	CS	IN:AC	53	7.2	5.6	2.7	84.8		6.4
	CS	IJ	50				80.0	6.4	
Application on standing spring barley 1/5/14 (soil moisture wet)	CS	BS	53				84.8	6.5	
	CS	BS:AC	53	7.5	5.7	3.3	84.8		6.5
First winter wheat trial at the end of tillering stage 3/4/14 (soil moisture wet)	CS	BS	53	7.3	6.5	4.4	84.8	6.2	5.7
	AD	BS	53	7.6	6.5	2.2	79.5	3.5	4.3
	MS	BS	37.5	7.2	6.0	3.7	78.8	1.4	1.5
	PS	BS	31.7	7.1	5,4	8.7	84.4	2.8	3.0
	AS	BS	53	8.0	6.6	7.7	79.5	2.4	2.7
Second winter wheat trial at the beginning of flag leaf emergence 30/4/14 (soil moisture wet)	CS	BS	53	7.4	6.2	4.4	84.8	6.2	5.7
	AD	BS	53	6.7	3.2	8.7	79.5	3.5	4.3
	MS	BS	37.5	7.0	6.1	3.7	78.8	1.4	1.5
	PS	BS	31.7	6.8	5.6	2.2	84.4	2.8	3.0
	AS	BS	53	7.9	6.5	7.7	79.5	2.4	2.7

*Target pH 6.0

Ammonia emission rate was measured using a combination of a validated calibrated dynamic chamber method and passive samplers (Gericke et al. 2011 and Quakernack et al. 2012). Crop yield and nitrogen uptake were obtained by destructive sampling.

The field experiments were carried out in two fields at the Research Centre in Foulum, Aarhus University, Denmark. The soil for both early trials (barley and wheat) had a loamy sand texture (clay 7.7 %, silt 9.8 %, sand 76.8 % and total organic C 5.8%) with the following properties: bulk density 1.32 g cm^{-3}, pH 6.1 no crop residues and no tillage < 1 week before slurry application. The second trial field also had a loamy sand texture (clay 6.8 %, silt 8.2 %, sand 79.5 % and total organic C 5.5%) with the following properties: bulk density 1.30 g cm^{-3}, pH 6.2, crop residues were present and no tillage < 1 week before manure application.

3. Results and discussion

Application techniques with CS were compared in the barley trials. In April 2014, in the first trial CS was applied by BS, BS with IN, IN plus AC, and IJ to bare soil before seeding of spring barley. All abatement treatments significantly reduced NH_3 emissions compared to BS. IN reduced NH_3 emissions by 60 % (Fig. 1 A), IN with AC by 89 %, while IJ had the highest reduction effect of 96 %. Emission from BS:IN:AC and IJ were not significantly different. Overall the order of NH_3 emissions was IJ = BS:IN:AC < BS:IN. In May 2014, in the second trial BS:AC reduced NH_3 loss from applied CS by 74 % compared to the non-acidified variant (Fig. 1 B), the effect of acidification being quite similar as in the first trial.

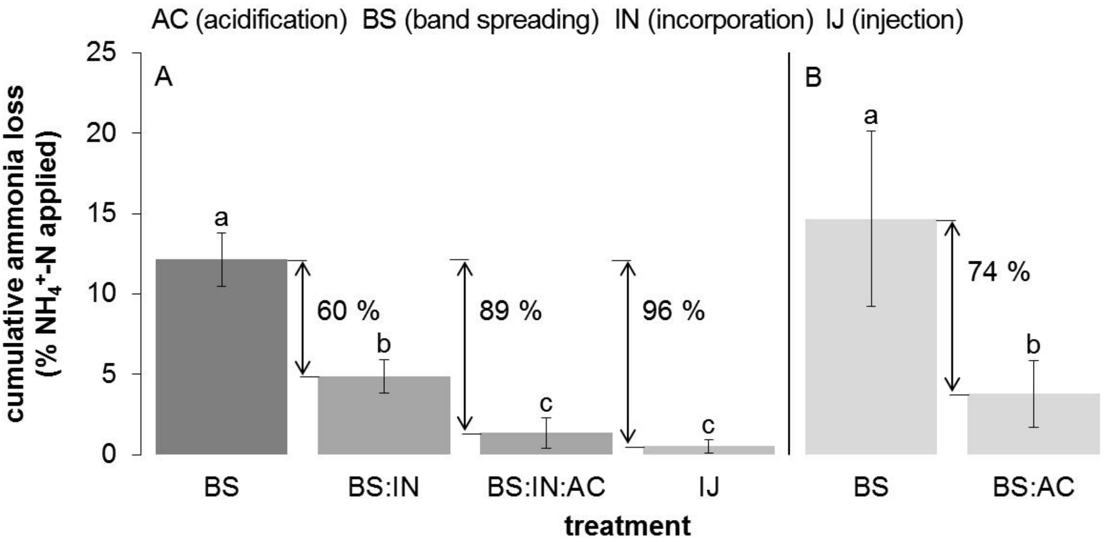

Figure 1: Ammonia losses after application of CS to spring barley by band spreading (BS), band spreading with incorporation with rotary cultivator (BS:IN), and closed slot injection (IJ), with or without acidification (AC). Application rate was 84.8 kg NH_4^+-N ha^{-1} to BS, BS:IN and BS:IN:AC and 80 kg NH_4^+-N ha^{-1} to IJ before seeding (A, early experiment). For part B (late experiment) 84.8 kg N ha^{-1} was applied at stem elongation. Lower case letters indicate significant differences between treatments at $p < 0.05$ ($n = 4$ except $n = 2$ for BS A; oneway ANOVA, LSD test).

Emissions in BS:IN were significantly higher as emissions of the other abatement techniques due to 4 hours delay of incorporation. Emissions would have probably been on the same level if CS had been incorporated immediately after application. BS:IN:AC and IJ (Fig. 1 A) both had the same NH3 emissions abatement effect. However, in farmer's practice a direct incorporation or injection of slurry is often not possible; therefore AC could play an important role in NH_3 abatement as it gives additional time windows to farmer's for low emission manure application under arable conditions. However, BS:AC has increased costs compared to BS, due to the requirement for specialised staff handling concentrated acid (Fangueiro et al. 2015), because of potential working hazards related with adding strong acids to slurries (Sommer et al. 2013). Nevertheless, due to the development of new technologies for applying acid during field application this approach can now be applied more safely. Nevertheless, in the long run, due to cost-effective sulphur fertilisation, ease of application and positive yield effects, AC could be a cost efficient alternative to IJ.
The winter wheat trials were carried out to investigate effects of acidification on NH_3 emissions and related crop effects. Slurries were applied at the beginning of April 2014 at the first wheat trial (end of tillering stage). Ammonia emissions from all slurry types were reduced using AC except for

PS (Fig. 2). Significantly, the AC treatment reduced emissions at CS by 61 %, AD by 57 %, MS by 74 % and AS by 58 % considering all slurry types individually.

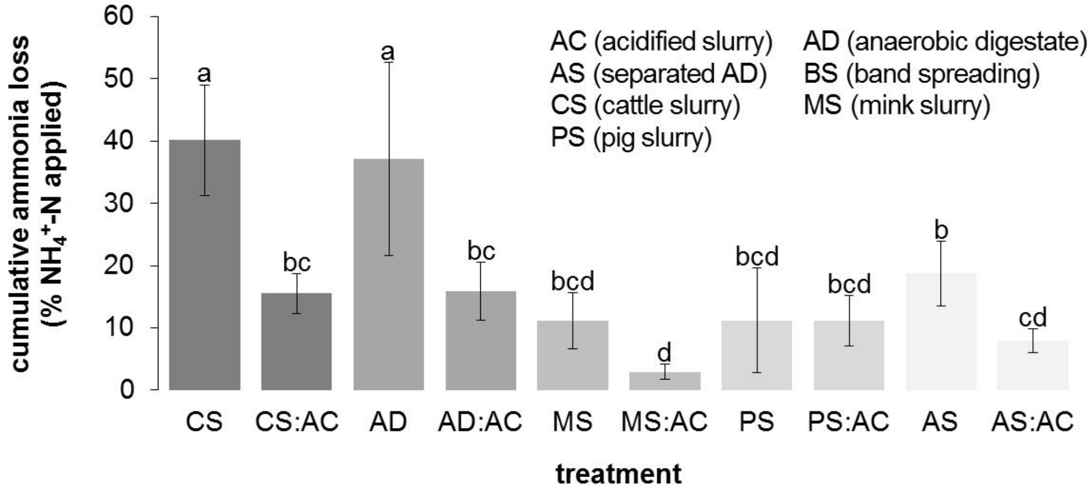

Figure 2: Ammonia losses after application of five different slurries by band spreading not incorporated and acidified variant. Applied NH_4^+-N ha^{-1}: CS = 84.8 kg; AD = 79.5 kg; MS = 78.75 kg; PS = 84.42 kg; and AS = 79.5 kg on winter wheat during vegetation on the 3/4/14. Lower case letters indicate significant differences between treatments at $p < 0.05$ ($n = 4$; oneway ANOVA, LSD test). MS and AS have a significant difference to the AC variant at $p < 0.05$ ($n = 4$; t test).

At the second trial application to winter wheat (flag leaf emergence) emissions from different slurry types were significantly reduced by AC. Compared to the first trial, acid requirements increased slightly due to changes in slurry composition after longer storage (Fig. 3). Emissions from CS were significantly reduced by acidification by 67 %, from AD by 45 % and AS by 45 %.

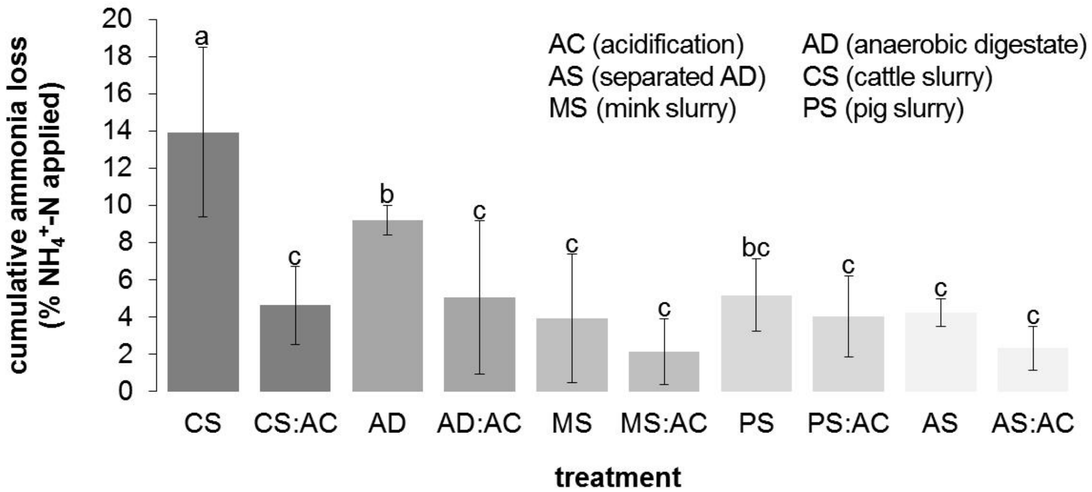

Figure 3: Ammonia losses after application of five different slurries by band spreading not incorporated and acidified variant. Applied NH_4^+-N ha^{-1}: CS = 84.8 kg; AD = 79.5 kg; MS = 78.75 kg; PS = 84.42 kg; and AS = 79.5 kg on winter wheat during vegetation on 30/4/14. Lower case letters indicate significant differences between treatments at $p < 0.05$ ($n = 4$; oneway ANOVA, LSD test). AS has a significant difference to its AC variant at $p < 0.05$ ($n = 4$; t test). Note: The y-axis limits differ from those in Fig. 2.

Acid demand for acidification varied considerably between slurry types (Table 1) indicating interactions effects between slurry type and economy and applicability of acidification. The overall NH_3 emission level differed considerably between both winter wheat trials (Figs. 2 and 3). This was probably due to canopy effects and lower wind speeds at the second application.

4. Conclusion

The experimental approach used in this work (small replicated plots with a large number of experimental treatments) provides a means for comparing multiple slurry application technologies without interference from differences between separate experiments. In general, results confirmed that in-field slurry acidification, slurry injection, and rotary incorporation all significantly reduced NH_3 emissions from field-applied slurry. Furthermore, incorporation plus acidification showed nearly the same reduction in emission as closed slot injection. Acidification may be a cost-effective approach for reducing ammonia emission and an alternative to injection depending on slurry type. Further trials may give further insight about the acid levels, for a desirable significant reduction on ammonia emissions combined with reasonable fertilization levels. Results give clear evidence that particular consideration have to be made with respect to acid requirements of different slurry types. Analysis of crop production aspects such as yields and N uptake effects related to treatments are ongoing.

References

[1] Fangueiro, D.; Hjorth, M.; Gioelli, F. (2015): Acidification of animal slurry - a review. Journal of Environmental Management, February (149), 45–56.
[2] Gericke, D.; Pacholski A.; Kage H. (2011): Measurement of ammonia emissions in multi-plot field experiments, Biosystems Engineering, 108(2), 164-173
[3] Quakernack, R.; Pacholski, A.; Techow, A.; Herrmann, A.; Taube, F.; Kage, H. (2012): Ammonia volatilization and yield response after application of biogas residues to energy crops in a coastal marsh of Northern Germany, Agriculture, Ecosystems and Environment, 160, 66-74
[4] Sommer, S.G. (eds.); Christensen, M.L.; Schmidt, T.J.; Lars, S. (2013): Animal Manure Recycling. Treatment and Management: Wiley

TC-O_04 Emission situation of anaerobic digestion of agricultural residues and bio-waste

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1. Objectives

With respect to the climate protection, a quantitative assessment of the emissions of energy generation from biomass and biological waste treatment is important. The DBFZ investigated the emission situation of biogas plants in Germany within research projects with project partners of selected biogas plants to analyze the emission situation of biogas plants. The results of the emission measurements are used to assess the ecological impact of anaerobic digestion (AD) and to describe possible mitigation measures to reduce the occurring greenhouse gas (GHG) emissions.

2. Methodology

Emissions were measured at 12 AD plants using the separately collected organic fraction of household waste (bio- waste), as well as at 10 agricultural biogas plants using energy crops and manure. The emission analysis included the determination of methane (CH4), nitrous oxide (N2O) and ammonia (NH3). Following the inspection of the biogas facilities on site, potential significant emission sources within the process chain were identified and several sampling points at AD plant were examined (see Table 1).

Table 1: Emission sample points of investigated agricultural and bio-waste biogas plants [1], [2].

Sample points	Agricultural AD plants	Bio-waste AD plants
Silage	Storage of silage Feeding system	Delivery and conditioning of bio-waste (material handling)
Fermenter	Fermenter, roof (foil /concrete), Leakage	Fermenter, roof, leakage Before and after exhaust gas treatment (acid scrubber and bio-filter)
Digestate	Digestate storage tank (open, closed, gas tight)	Storage of digestate Post composting (open, closed, aerated, un-aerated) Separation and drying of digestate
Gas utilization	Exhaust of CHP unit (combined heat and power plant), Biogas upgrading plant	Exhaust of CHP unit (combined heat and power plant)

The emission analysis included two measurement periods at each plant (one week in both winter and summer time), in which all plant components, from substrate delivery to storage of digestate and composting (in case of bio-waste) were investigated. The measured emissions of both periods were averaged. The methods used focus on the identification and quantification of single sources. According to the characteristics of the gases, the applied measurement techniques were adjusted. Leakage detection techniques were used to find the critical spots within the process; open and closed chambers were used to determine the main emission sources. Different methods of emission measurements were used for captured and diffuse emission sources. In case of captured emission sources (e.g., delivery hall with collection of exhaust, off-gas from CHP, exhaust duct to a bio-filter), the exhaust air flow was examined directly. Thereby, the volume flow and mass concentration within the investigated pipelines were determined. The volume flows were measured with vane anemometers. The quantity of the emission source was calculated from the concentration difference and the flow rate of the blower. Emissions of post-composting with active aeration (e.g., actively ventilated tunnel or container systems) were measured by using encapsulated areas with air extraction. In case of open windrows composting without active aeration, a wind tunnel as emission measurement was used. An air flow was generated by using a ventilator. After quantification of the single sources, the overall GHG emission from the biogas plant is summarized. The measurement methods, techniques and technical guidelines used are described in [3], [4]. Based on the measured emissions, GHG balances in compliance with the analysis of

GHG credits (avoided GHG emissions) were prepared. GHG credits can be generated for the substitution of fossil fuel (e.g., credits for the electricity production from biogas by substitution of fossil electricity supply, credits for the utilization of exhaust heat by substitution of fossil heat), for the use of fermentation products by substitution of fossil fertilizer and peat or including the humus effects as well as for avoided GHG emissions by using manure in fermentation processes instead of the open storage of manure. The considered GHG emissions for all processes of AD plants were converted into CO2 equivalents (CO2-eq) by using characterization factors according to the global warming potential for a 100-year time period: CO2 = 1, CH4 = 25, N2O = 298 [5].

3. Results and discussion

Various fermentation processes such as wet fermentation, dry fermentation and batch fermentation were analyzed according to the emission situation of agricultural and bio-waste digestion plants. The results show that the emissions are dominated not by the type of fermentation process or the technology used, but by the way in which the plants were managed.

3.1 Results- Agricultural Biogas Plants

The assessment of GHG emissions at the investigated agricultural biogas plants was limited to the gases CH_4 and NH_3 because (except in the CHP exhaust) no relevant N_2O emissions were detected. With respect to the overall CH_4 emissions the results show that the main emission sources are the open digestate storage tanks and the gas utilization system [1] (Figure 1).

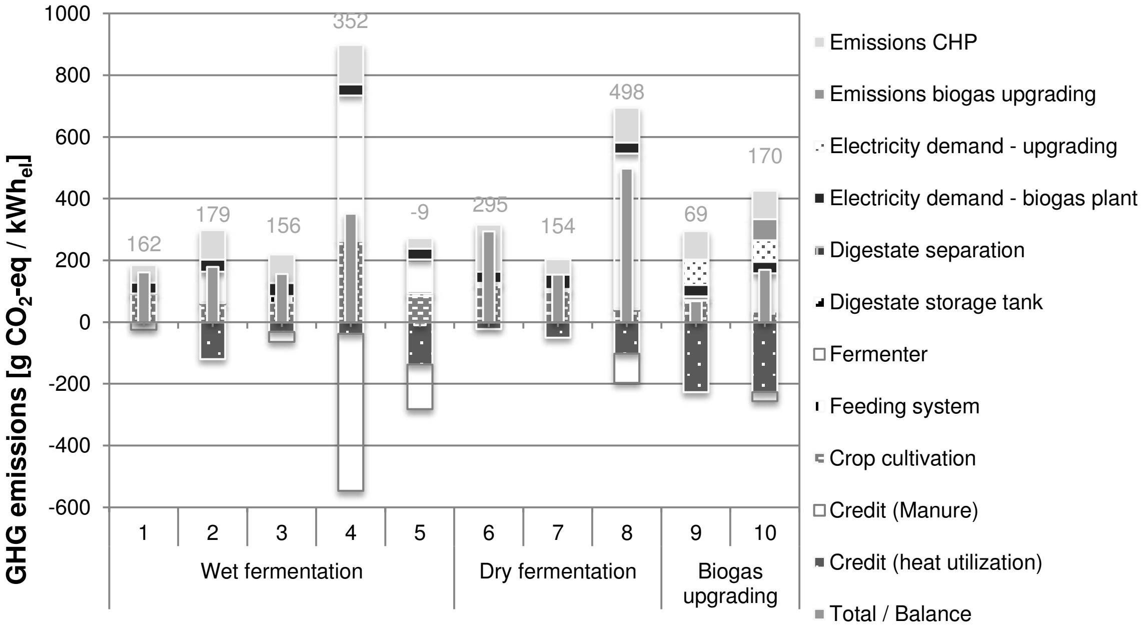

Figure 1: GHG emissions from the different plant components, and GHG credits and total GHG balance of investigated agricultural biogas plants. [1]

Biogas plant No.4 used mainly manure, therefore relatively high manure credits for "avoided emissions" in comparison to the traditional manure storage were determined. Moreover also comparatively high GHG emissions arise due to the large amounts of digestate which occur during storage of fermentation residues (open lagoons) and spreading of digestate. The total GHG balance of the biogas plant No.8 (batch system) is also dominated by high emissions of fermentation residues, because of the temporarily open storage of digestate before application. Plant nos. 9 and 10 inject the biogas to the natural gas grid after an upgrading process (biogas refining process to produce biomethane - equal to natural gas) – in these cases additional emissions occur from the upgrading process. These emissions mean methane losses – so called methane slip. For the methane slip the legal limit (in year 2011: legal limit 0.5%) were assumed, because both upgrading technologies caused higher emissions during the period of measurement [1].

3.2 Results - Bio-waste Digestion Plants

Based on the investigations [2] significant emission sources were identified. Especially the inadequate aeration directly after fermentation processes (in order to interrupt the methanogenic activity), as well as un-aerated or less aerated post-composting processes that caused extremely high GHG emissions (see plant No. 1, No. 2 or No. 12). For some of the investigated biogas plants, post-composting emissions are summarized in the amount of "emissions after bio-filter" (e.g. AD plant No.10). Post-composting processes with active ventilation (pressure ventilation) and enclosed composting systems were used at AD plant No. 7, 9 and 10. The overall emissions of AD plant No. 10 was quite low, because all parts of the fermentation and post composting process were totally encapsulated. Furthermore, AD plant No. 6 showed higher NH_3 emissions due to the drying of digestate at higher temperature and higher pH-value. In this case, the existing downstream acidic scrubber was out of operation during the measurements. All investigated biogas facilities operated with bio-filters as gas treatment. However, only four of twelve plants operated with acidic scrubbers, and, the proper operation was not always ensured. The operation of the bio-filters can also be problematic; extremely wet bio-filters for example can cause additional CH_4 production as observed at AD plant No. 8. The exhaust gas should be treated with acid scrubbers to trap NH_3 and minimize N_2O formation in the bio-filter (e.g. plant No. 5 and 9).

It should be recognized that there were also diffuse emission sources which were not collected by bio-filters (e.g. open doors of delivery hall at AD plant No. 6 and 7; post-composting at AD plant No. 8, 9, 11). Often digestate - whether separated or not separated – is stored open temporarily or for longer periods. Four of the seven examined plants which stored liquid digestate or process waters used covered storage tank (AD plant No. 4, 5, 8 and 9). Two plants (No. 5 and 8) with gas-proof covered storage tank are able to use the exhaust gas in the CHP.

Figure 2 summarises the measured GHG emissions from the different kinds of components as well as the calculated GHG credits (avoided manure emissions, heat utilization, fertilizer and humus value of digestate) and the total GHG balance (shown as column) according to [2].

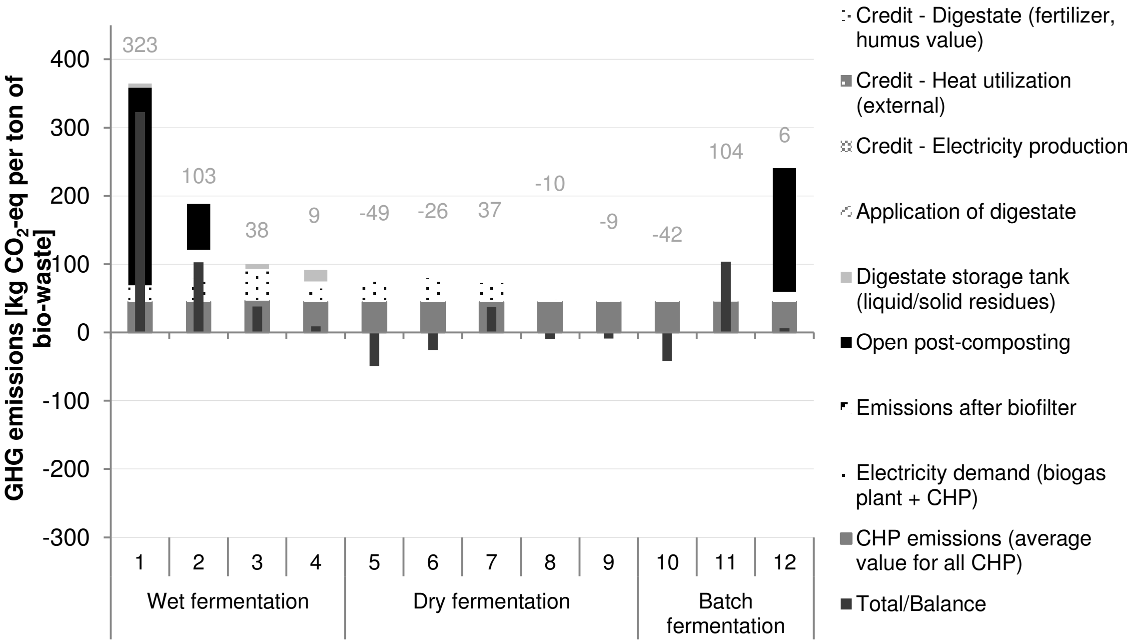

Figure 2: GHG emissions from the different plant components, and GHG credits and total GHG balance of investigated bio-waste digestion plants. [2]

The results show that the open storage of fermentation residues (with or without separation step) should be avoided. In addition to un-aerated post-composting processes and open storage of active material (e.g. solid digestate), the CHP was one of the most important sources of CH_4. According to the measured residual gas potential of digestate (bio-waste digestion) a wide range from 4% to 23% was determined, whereas ten of the twelve samples of investigated bio-waste digestion plants showed a relative residual gas potential >10% [2], [6].

4. Discussion

The problem of increased emissions is not the anaerobic process itself, but a non-optimal after-treatment of the digestate. The results show that the open storage of active material (e.g. insufficient fermented residues from batch fermentation systems), open digestate storage tanks, missing acidic scrubbers in front of bio-filters or insufficient air supply during the post-composting of digestate can cause relevant GHG emissions. Consequently avoiding open storage of insufficient fermented residues and using aerated post-composting with short turnover periods, smaller heaps and an optimized amount of structure (woody) material can reduce GHG emissions. However, the results for the open digestate tanks need to be interpreted carefully, since they cannot represent an average emission over a longer period of time. Nevertheless, for the purpose of emission reduction a gas tight cover of any open digestate storage will have a great effect on the emissions. With respect to the residual gas potential it can be stated that the determined methane potential of fermentation residues from bio-waste digestion provides basically higher values in comparison to agricultural biogas plants with multi-stage process. The consideration of GHG credits optimizes the overall GHG performance of the biogas facilities. A high share of electricity generation led to high GHG credits. As far as the utilization of exhaust heat of electricity production was possible, it had also a positive influence on the GHG performance of the AD plant. Moreover, the use of digestate showed positive effects on the GHG balances. In addition to the nutrient effect through the utilization of the fermentation residues as a fertilizer (substitution of mineral fertilizer) GHG emissions can be saved due to the humus effect of digestate. Especially composted digestate like fresh and finished compost contributed to the humus accumulation (carbon sink) and the humus reproduction of digestate. Compared to the production of fresh or finished compost digestate without post-composting process, which is used within the agriculture directly, less GHG credits were given. However, the risk of high emissions during the post-treatment of the fermentation residues was avoided.There is uncertainty about other emissions sources that are not coupled to the gas system of the plant, but still cause GHG emissions as stated in [7]. Concerning the emissions, the treatment and evaluation of temporary occurring emissions caused by certain operational conditions (e.g. emissions from pressure relief valves) are still unclear.

5. Conclusion and outlook

Methane emissions dominate GHG emissions from biogas plants. Depending on the used technology and the kind of operation, GHG emissions like CH_4, N_2O and NH_3 are occurring. Basically, the kind of operation of the plant and the handling of digestate determine the amount of GHG emissions. In general the emission situation is not uniform. It can be stated that all investigated biogas facilities showed potential for optimization. With respect to the development of methodology of emission measurements and the standardization of procedure for the determination of emissions of biogas plants, further investigations are necessary. Further scientific data about the current emission situation and the ongoing development as well as reliable measurement methods are required to determine the CH_4 emissions from the plants in operation today. In this regard the reliable measurement of stationary and diffuse emission sources is of high importance.

Acknowledgements

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References

[1] Liebetrau J.; Daniel-Gromke J.; Oehmichen K.; Weiland P.; Friehe J.; Clemens J.; Hafermann C. (2011): Emissionsanalyse und Quantifizierung von Stoffströmen durch Biogasanlagen im Hinblick auf die ökologische Bewertung der landwirtschaftlichen Biogasgewinnung und Inventarisierung der Deutschen Landwirtschaft. FNR e.V., FKZ: 22023606.

[2] Liebetrau J.; Krebs C; Daniel-Gromke J.; Denysenko V.; Stinner W.; Nebel, E.; Cuhls C.; Mähl B.; Reinhold J. (2013): Analyse von Emissionen klimarelevanter Gase durch Biogasanlagen im Hinblick auf die ökologische Bewertung der Biogasgewinnung aus Abfällen. BMU, FKZ: 03KB027.

[3] Liebetrau J.; Clemens J.; Cuhls C.; Hafermann C.; Friehe J.; Weiland P.; Daniel-Gromke J. (2010): Methane emissions from biogas producing facilities within the agricultural sector. Engineering in Life Sciences, vol 10: 1-5.

[4] Liebetrau J.; Reinelt T.; Clemens J.; Friehe J.; Weiland P. (2013): Analysis of greenhouse gas emissions from 10 biogas plants within the agricultural sector. Water Science and Technology. vol 67 (6): 1370 - 1379. doi: 10.2166/wst.2013.005.

[5] IPCC (2007): Climate Change 2007. Fourth Assessment Report (AR4) of WG 1 Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge and New York.

[6] Daniel-Gromke, J.; Liebetrau, J.; Denysenko, V.; Krebs, C. (2015): Digestion of bio-waste - GHG emissions and mitigation potential, Energy, Sustainability and Society, 5:3, doi:10.1186/s13705-014-0032-6.
[7] Dumont M, Luning L, Yildiz I, Koop K (2013) Methane emissions in biogas production. In: Wellinger A, Murphy J, Baxter D (eds.) The biogas handbook – science, production and application. Woodhead Publishing Limited, Series in Energy No. 52. Cambridge, pp 248-266

TC-O_05 Abatement of ammonia emissions from digested manure using gas-permeable membranes

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1. Objectives

A new strategy to avoid NH_3 emissions from anaerobically digested manure was tested using the gas-permeable membrane process. Evaluation of the efficiency of NH_3 recovery from digestate as well as mitigation of NH_3 emissions to the atmosphere were carried out. The maximum capacity of the sulphuric acid to recover N from digestate was also studied. The system contributed to the reduction of NH_3 emissions and recovery of nutrients from the digestate whilst converting ammonia into an ammonium salt fertilizer.

2. Methodology

Experiments were conducted in 2-L vessels consisting of PET jars for an effective digestate volume of 1.3 L. The vessels were not completely airtight. A diaphragm pump was used to continuously circulate diluted H_2SO_4 (300mL) through the tubular membranes (made of ePTFE) inside the digestate vessels and back into the acid tank using constant flow rate of 5.8 L/day. The membrane manifolds were submerged in the digestate vessels, which were kept closed but not airtight. Ports were installed on top of the vessels to obtain samples and monitor pH (Fig. 1). The digestate was continuously agitated using magnetic stirrers.

Digestate was collected directly from the digester of a 500kW biogas plant in plastic containers, transported in coolers and stored at 4ºC. The biogas plant was located in Salamanca (Spain). Three different samples were collected from the biogas plant and used for batches 1-2, 3-5 and 6-7, respectively. Seven consecutive batches of digestate were used to recover ammonia, thereby digestate was removed from the vessels and replaced by fresh digestate as NH_4^+ was depleted and recovered in the acid tank. pH adjustment of the digestate was carried out as needed to endpoint pH 9.0, (except in batch 4 where pH endpoint was 11) whenever the pH of the digestate decreased under 8.0, to increase NH_3 recovery. Digestate samples from PET jars and acid solution samples were daily withdrawn in order to monitor pH, alkalinity and NH_4^+-N. In addition, initial and final samples of digestate in each batch were analyzed for pH, alkalinity, total solids (TS), volatile solids (VS), total chemical oxygen demand (CODt), NH_4^+-N, total Kjeldahl nitrogen (TKN), nitrite (NO_2^--N), nitrate (NO_3^--N), and total phosphorous (Pt) determination. All experiments were carried out in duplicate and results are expressed as means.

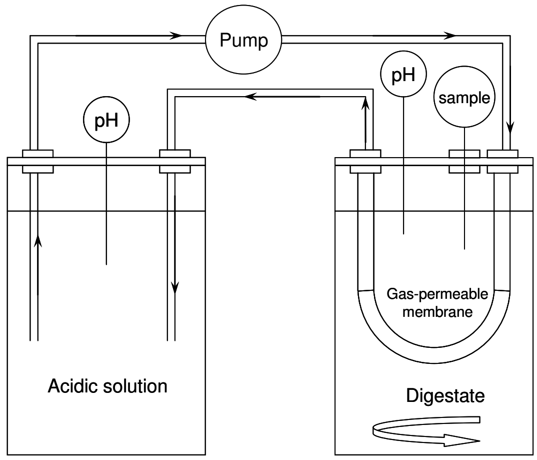

Figure 1: Experimental device for ammonia capture from digestate using gas-permeable membranes.

3. Results and discussion

3.1 Removal of NH_3 by the gas-permeable system.

Seven consecutive batches using gas-permeable membranes were used to capture NH_3 and concentrate this nutrient in the acid solution, to be used as an ammonium salt fertilizer. Digestate with an average of 5,550±840 mgNH_4^+-N L^{-1} was used. It is important to emphasize that the acidic solution was the same during the entire experiment. The experiment processed seven batches of digestate to test the maximum capacity of the sulphuric acid to recover N from digestate. The recirculation of this liquid in a closed loop between the digestate vessel and the acid tank achieved an NH_4^+concentration in the recovery solution (46,590±1,970 mgNH_4^+-N L^{-1}) about 9 times higher than the original digestate (5,550 mgNH_4^+-N L^{-1}) (Fig. 2). A total N mass of 13,980±590 mgN was recovered in the acid from the digestate during the experimental period. It was noticed that as NH_4^+ had been recovered from the digestate, the pH of the acid solution increased (maximum pH reached was 8.8), being necessary to add more acid to an endpoint pH 1. These findings are in agreement with those reported by García and Vanotti [1] who observed a high N recovery from manure with different NH_4^+ strengths using the gas-permeable membrane technology.

3.2 Abatement of ammonia from digestate in batches 1-3

During the first part of the experiment, batches 1-3, pH average maintained between 8.05 and 8.22 (Table 1), therefore NH_4^+ recovery efficiency were high (84-62%) (Fig. 3), and NH_4^+ volatilization was quite low with values of 35, 7 and 4% respectively. Therefore, as pH was maintained near 8.00 during the experimental time, the membrane device was able to recover almost all NH_4^+ from the digestate avoiding its volatilization.

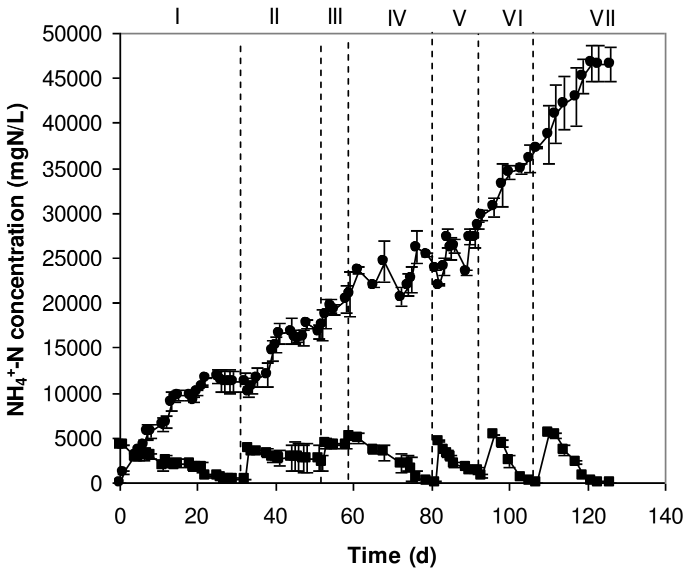

Figure 2: Removal of ammonia from digestate in seven consecutive batches (■) and corresponding N concentration increase in the acid solution (●) . Dash lines represent duration of each batch.

3.3 Abatement of ammonia from digestate in batches 4-7

In batches 4-7 pH average increased from 8.70-9.65, and NH_4^+-N recovery efficiency decreased to 13-33% (Table 1), increasing NH_4^+ volatilization between 57-86% of initial NH_4^+. Considering previous studies carried out with manure, both the maximum NH_4^+ recovery rate and the average NH_4^+ recovery rate obtained in batches 4-7 were very high: between 1003-1910 mg NH_4^+-N/L/day for the maximum rate and 483-1078 mg NH_4^+-N/L/day for the average rate. Free ammonia (FA) content was also very high in batches 4-7(Table 1). According to our previous studies [1, 2], mass recovery of NH_4^+ through the membrane increased as FA content increased in manure as a result of pH adjustment (with an aeration treatment or alkali addition) with manure containing 1000 to 2300 NH_4^+-N/L. However, although in the present study FA content maintained high during batches 4-7 (448-572 mg N L-1) (Fig. 3), allowing active permeation of NH_3 through the membrane, the absorption capacity of the membrane reactor was not enough to recover all FA in digestate and a fraction of NH_4^+ was volatilized, as the reaction vessels were not airtight. This volatilization loss was significant in the mass balance, compared with results from batches 1-3 in which volatilization were low, and from [1], who obtained volatilization values of 8% of initial NH_4^+, and [2], who obtained values of 1.5 to 6% of initial NH_4^+ using similar reactor vessels and tubular membrane length. Thus, to recover all N and to avoid significant volatilization from digestate, the reaction device should be totally airtight. In this way the membrane manifold will work at its maximum absorption capacity and high recovery efficiency although it will take more time to recover all N.

Table 1: Comparison of free ammonia (FA) and average NH4+ recovery rate by the gas-permeable membrane reactor in the seven batches with digestate.

Batch	Initial NH_4^+ in digestate	NH_4^+-N volatilized in the air[a]	Maximum recovery rate[b]	Average recovery rate	FA average	Average pH
	(mgN)		(mgNH_4^+-N/L/day)		(mgN/L)	
1	5580	1950	1244	324	150	8.12
2	5863	400	521	222	206	8.22
3	7233	268	136	91	188	8.05
4	7788	6691	1254	572	572	9.65
5	6987	3969	1003	483	448	8.72
6	8103	5833	1910	1078	572	8.87
7	8387	5588	1601	798	555	8.70

[a]NH_4^+-N volatilized in the air was equal to initial NH_4^+ in manure minus the remaining NH_4^+-N in manure minus the NH_4^+ recovered in the acidic solution.
[b]Highest NH_4^+ mass recovered in 1 day; 0.0323 m^2 of membrane surface area.

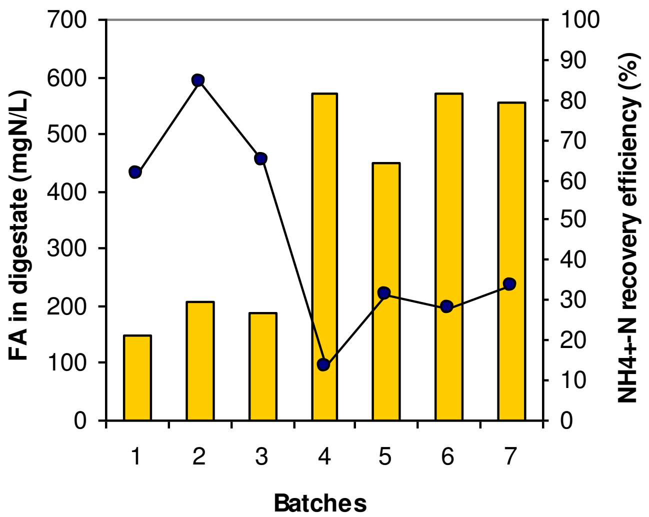

Figure 3: Free ammonia in digestate in seven consecutive batches (yellow columns) and corresponding ammonia recovery efficiency.

3.4 Expected benefits in emissions mitigation

Important mitigation strategies of NH_3 emissions for concentrated livestock operations often involve capturing or trapping the fugitive gases and subsequent treatment of the captured air with bio-filters and bio-covers to remove the NH_3; which means N loss for further agriculture use and therefore, an important economic loss.

In the case of the anaerobic digestion process the breakdown of proteins raise ammonium concentration of the medium that contribute to ammonia emissions when the digestate is used for land application, as well as ammonia-mediated inhibition of the anaerobic digestion process. Therefore, new strategies for reducing or minimizing NH_3 losses from livestock production (manures, digestate) are needed. In this sense, the gas-permeable technology can recover and concentrate NH_3 from digestate (or any other highly NH_3 charged wastewater) without using high pressure or temperature, pre-treatment of the digestate or addition of additives.

Regarding the perspectives of utilization for the N enriched acid solution, it is well known that the Haber-Bosh process is used by the chemical industry to produce NH_3, which is a plant fertilizer, and can be combined with acids to produce ammonium salt like NH_4NO_3 and $(NH_4)_2SO_4$, which are the most commonly used fertilizers in Europe [3]. The price of N fertilizers is directly related to the price of natural gas, since its production is linked to the Haber-Bosch process as mentioned; besides this process emits carbon dioxide to the atmosphere. Therefore, environmental friendly and cost-effective alternatives to produce N fertilizers are welcome by society. In this work the gas-permeable membranes successfully removed NH_3 from digestate, recovering N in the form of $(NH_4)_2SO_4$, which is desirable to export N off the nutrient surplus areas, like Nitrate Vulnerable Zones, to other regions were N is needed, and thus avoiding environmental pollution in the form of ammonia emissions to the atmosphere. The remaining digestate, that contained less N, also can be used for land application in higher dosages due to the reduction of its N content.

4. Conclusion and outlook

As long as digestate pH was maintained around 8.00, ammonia was successfully recovered from digested manure avoiding its volatilization to the atmosphere and preventing air pollution. In this case FA content in digestate allows an excellent permeation through the membrane avoiding volatilization. When digestate pH reach 8.70, FA content in digestate increases promoting volatilization under non airtight tank conditions. In this case it is possible to increase NH_4^+ recovery efficiency if the area of the membrane system is adjusted to maximize N recovery efficiency and to shortening N recovery time, also considering an airtight system. Therefore, gas-permeable membrane technology could be combined with anaerobic digestion to avoid ammonia emissions and at the same time recovering the N as a valuable nutrient in the form of a concentrated ammonium salt fertilizer with 46,000 mg N/L NH_4-N.

Another application of gas-permeable membranes is related to NH_4^+ inhibition in the anaerobic process. It is well known that unionized or free ammonia (NH_3) has been reported to inhibit methanogenesis at initial concentration of 100–1100 mg N L^{-1}, thus this technology is a suitable strategy to avoid this inhibition as it removes NH_3 from digestate. Moreover, if removal of NH_4^+ does not damage the carbonaceous compounds as [1] pointed out the process will be improved as more methane will be produced, thus future trends will be focus on this point.

Acknowledgements

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References

[1] García-González M.C., Vanotti M.B. 2015. Recovery of ammonia from swine manure using gas-permeable membranes: Effect of waste strength and pH. Waste Management, 38:455-461.
[2] García-González M.C., Vanotti M.B., Szogi A.A. 2015. Recovery of ammonia from swine manure using gas-permeable membranes: Effect of aeration. Journal of Environmental Management, 152: 19-26.
[3] IFA (International Fertilizer Industry Association), 2015. http://www.fertilizer.org/Statistics (Accessed March 2015).

TC-O_06 Reduction of ammonia emissions and related greenhouse gas fluxes from separated anaerobic digestates

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1. Objectives

Separation of organic manures can cause increasing ammonia (NH_3) and concomitant greenhouse gas (GHG) emissions from the solid phase [1]. In addition to the environmental damage, ammonia emissions are a loss of valuable nutrients, in particular in organic agriculture. Therefore, a new approach including incubation of manures with microbes and a nutrient source was tested to reduce NH_3 emissions without increasing greenhouse gas (GHG) fluxes.

2. Methodology

Anaerobic digestates (AD) have a strong propensity to emit NH_3 from raw and separated substrates [2] calling for reduction measures. Addition of a mixture of molasse (30% water content) with indigenous microbes (IM) to AD was hypothesized to reduce NH_3 emissions due to pH reduction and NH_4^+-uptake by microbes. It was shown in earlier studies that addition of fermentation substrates to slurries can reduce NH_3 emissions [3] but this addition may also increase GHG emissions in form of CH_4 and N_2O emissions [4]. Therefore, testing the novel approach consisted of two steps: A. incubation trial to test the effects of the addition of solutions containing IM and a nutrient source (molasse) on NH_3 and GHG emissions and N turnover. The second aim of the lab trial was the identification of the smallest dose effective for the reduction of NH_3 emissions, as amount of molasse probably also has an effect on the GHG. B. on-farm essay to test the approach on the scale of practical farm conditions. **A.** The effect of varying doses of molasse and incubation enhancers (dolomite clay, biochar) in combination with the addition of IM to the solid phase (500 g, n = 3) of AD on NH_3 and GHG emissions were tested in an aerobic incubation laboratory trial. Addition of IM also included one treatment with 1 day pre-incubation of IM in molasses solution at a temperature of 20° (Tab. 1). Samples of solid phase AD were taken immediately after separation and instantaneously mixed with the treatment solutions. NH_3, N_2O, CH_4, CO_2 emissions were determined with a Lumasense® photoacoustic field gas monitor (PA) for 4 weeks [5]. Similar to farm practice solid phase AD were packed in the incubation containers to bulk densities determined in on-farm heaps prior to experimentation. To account for high NH_3 emissions immediately after separation, first PA measurements were done on the farm; after two days samples were moved to the lab and measurements were continued. **B.** Two heaps (0.5 t Fresh Matter (FM)) of AD solid phase were mixed immediately after separation with the most effective incubation solution (*D pre-inc* small amount of molasse, small GHG emissions, strong NH_3 reduction) and incubated for 6 weeks on farm compared to two untreated AD heaps. Substrates were produced within 1 day. Only NH_3 lossess were qualitatively quantified in acid traps [6] and N compounds were determined. Heaps were stored in a barn to avoid rain.

Table 1: Substrate added to freshly separated AD (0.5 kg) for the reduction of ammonia emissions; biochar was produced from riparian reeds and grass from Elbe river banks.

Treatment	Molasse	IM	Water Added	Total mass added	Dolomit clay	Biochar	Pre-incubation
			Unit				
	g/pot	ml/pot	ml/pot	g/pot	g/pot	g/pot	
Ck	0	0	35.75	0	0	0	no
A	27.5	10	17.50	55	10	0	no
B	13.75	10	21.63	45.38	10	0	no
C	6.875	10	23.69	40.56	10	0	no
D	3.4375	10	24.72	38.16	10	0	no
D no clay	3.4375	10	24.72	38.16	0	0	no
D biochar	3.4375	10	24.72	38.16	0	10	no
D pre-inc	3.4375	10	24.72	38.16	10	0	yes, 1 day

3. Results and discussion

3.1 Incubation trial

Addition of IM with molasse yielded in all cases a significant reduction of NH_3 emissions by up to 35 % in the incubation trial (Fig. 1). High rates of molasse addition (treatments A, B) resulted in the strongest NH_3 loss reductions but pre-incubation of IM with small amounts of molasse reduced emissions to the same level. Reduction of NH_3 emissions did not occur immediately but after about 2-7 days after separation due to shift from dominance of anaerobic to aerobic microbial community (shift from CH_4 emission to CH_4 uptake, data not shown). After that, emission dynamics showed a close agreement between all treatments. Treatment *D pre-inc.* showed a slightly smaller slope in this second phase of the incubation.

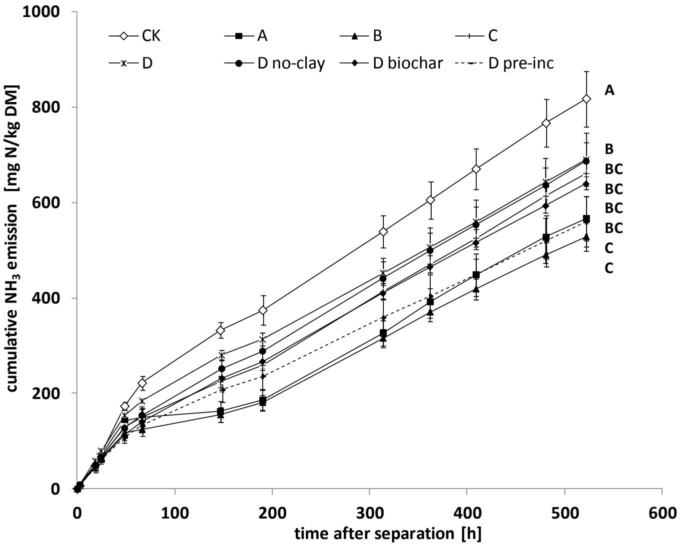

Figure 1: Cumulative NH_3 emissions after separation of AD with and without addition of fermentation solution with microbes (IM) under aerobic incubation, different levels of molasse addition and incubation enhancers, for treatments see Tab. 1, one-way-ANOVA (n=3), letters = levels of significance, LSD-test ($p<0.05$), error bars = std. error.

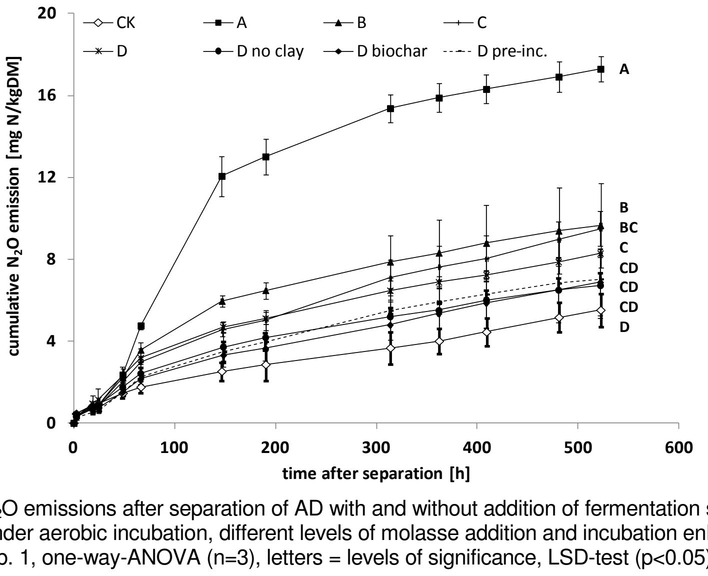

Figure 2: Cumulative N_2O emissions after separation of AD with and without addition of fermentation solution with microbes (IM) under aerobic incubation, different levels of molasse addition and incubation enhancers, for treatments see Tab. 1, one-way-ANOVA (n=3), letters = levels of significance, LSD-test ($p<0.05$), error bars = std. error.

Concomitantly, N_2O emissions increased with increasing amounts of molasse addition (Fig.2). As a result, the two highest doses showed significant differences to almost all other treatments. Emissions from treatment 'D pre-inc.', which was also characterized by a strong reduction of ammonia emissions, were not significantly different from the untreated control. Overall the stimulation of N_2O emissions seems to depend linearly on molasse addition. However, addition of biochar, omission of dolomite clay and pre-incubation slightly decreased the cumulative emissions.

With respect to the final cumulative CH_4 emissions no significant differences between treatments were observed. However, the emission pattern differed considerably (data not shown). First, separated ADs were a methane source depending on the amount of molasse added while they changed to a methane sink about 3 days after application. Initial increases in CH_4 fluxes due to addition of molasse were counterbalanced by a stronger CH_4-uptake in the second aerobic phase of the incubation experiment.

All trace gas emissions measured were integrated as total GHG emissions expressed in CO_2-equivalents, NH_3 being considered as an indirect GHG source (Fig. 3). Carbon dioxide emissions; which were strongly dependent on molasse addition, were not included, as molasse-C as a renewable C-source would be transformed to CO_2 independent of its use as fermentation substrate. Addition of high amounts of molasse resulted in significantly higher, up to 100% increased GHG emissions compared to the untreated control. Solid phase AD mixed with pre-incubated fermentation solution was characterized by the lowest GHG-emissions even slightly lower than the untreated control.

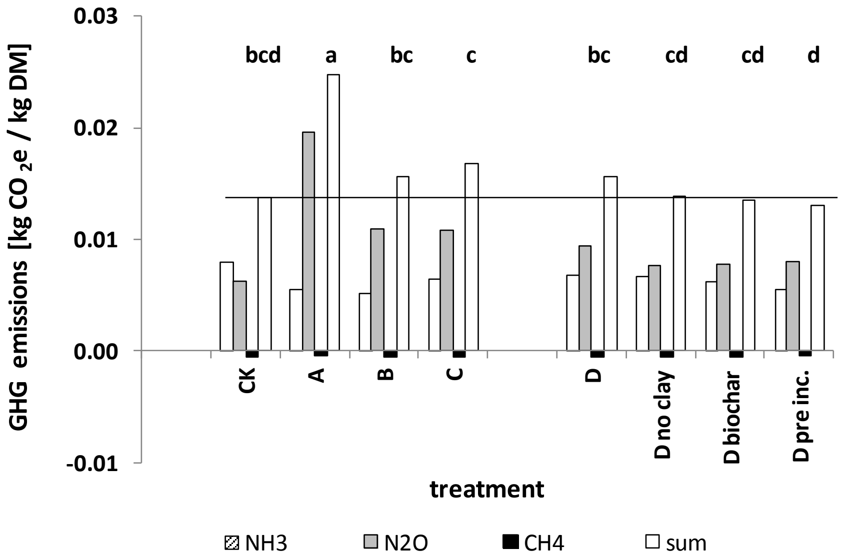

Figure 3: GHG emissions expressed as CO_2 equivalents of separated AD with and without addition of fermentation solution with microbes (IM), different levels of molasse addition and incubation enhancers, for treatment discription see Tab. 1, one-way-ANOVA (n=3), letters = levels of significance, LSD-test ($p<0.05$), error bars = std. error.

Overall, the addition of high amounts of molasse strongly reduced ammonia emissions. But this was connected with strong adverse effects on GHG emissions making this no viable option for the prevention of ammonia emissions from separated AD. However, when pre-incubated, the addition of small amounts of molasse yielded the same emissions reduction without increased, even slightly reduced GHG emissions. Low costs due to small amounts of molasse applied make this approach potentially economically feasible for practical applications.

3.2 Farm trial

The incubation trial set-up was repeated with the best performing treatment ‘D pre-inc.’ on the farm scale in August – September 2014. Temperatures were considerably higher (~20°C) than during the first days of the incubation trial (~10 °C) when first trace gas measurements were done on the farm. The reduction of ammonia emissions by adding pre-incubated fermentation substrate also yielded a considerable reduction in ammonia emissions (Fig. 4) but the emission patterns differed between both trials. Strong reduction of emissions already occurred on the first day after AD separation (about 35 %). This difference in emission pattern is attributed to differing temperature conditions which sped up the microbial processes in the farm trial. Similar to the incubation trial reduction of ammonia emissions mainly occurred during a limited time window up to 4 days after separation. Later on emissions from treated heaps and the control behaved in the same way reducing the reduction effect in the course of time. However, even after 3 weeks emissions were reduced by about 10 % compared to the untreated control. Nevertheless, under practical management conditions, a heap will be built up within a time period of several weeks, with fresh material packed on top of old material, so that the initial effect of the treatment on ammonia emissions is more relevant than the effects occurring at later stages of the experimental AD storage period.

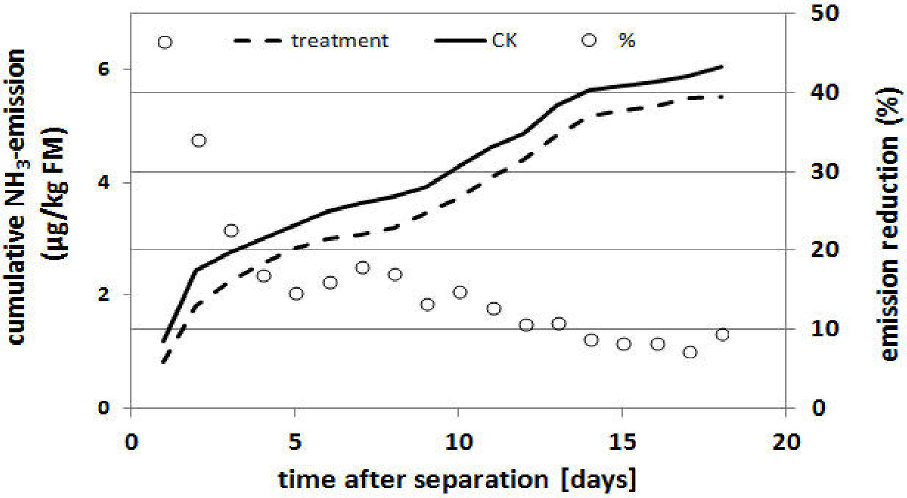

Figure 4: Cumulative ammonia emissions and emission reduction after addition of pre-incubated fermentation solution (treatment D pre-inc.) to 0.5 t FM of solid phase AD in on-form trial (FM = fresh matter).

The treatment showed also a significant effect on AD mineral N (ammonium and nitrate) concentrations with an average increase over the whole heap profile of 30% compared to the control, the effect being most pronounced in the lower layer of the heap. At the actual size of AD heaps on practical farms the upper layer of a heap is much smaller compared to heap interior volume as in the small heaps investigated in this trial. So the positive effects of the treatment on AD mineral N concentration would probably be more pronounced under practical conditions due to its stronger effect on lower layers in the solid phase AD heap.

4. Conclusion and outlook

Ammonia emissions from separated AD were significantly reduced by addition of pre-incubated IM and small amounts of molasse as incubation starter without increasing GHG emissions. In this trial only one type of AD from organic farming with comparably low ammonium N concentrations was tested. To confirm this approach GHG emissions from large AD heaps and other solid phase substrates with higher ammonium N concentration and larger AD heaps should be investigated.

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References

[1] Perazzoloa F.,Mattachinia G., Tambonea F., Misselbrookb t., Provolo G. 2015. Effect of mechanical separation on emissions during storage of two anaerobically codigested animal slurries. Agr. Ecosyst. Environ. 207: 1-9

[2] Ni K., Pacholski, A., Dirk Gericke, D. and Kage H. 2012. Analysis of ammonia losses after field application of biogas slurries by an empirical model, Journal of Plant Nutrition and Soil Science, 175(2): 253-264

[3] Vandré R, Clemens J 1997. Studies in the relationship between slurry pH, volatilization processes, and the influence of acidifying additives. Nutrient Cycling in Agroecosystems. 47, 157-165

[4] Morley N., Baggs E.M., 2010. Carbon and oxygen controls on N_2O and N_2 production during nitrate reduction. Soil Biol Biochem. 42, 1864–1871.

[5] Monaco S., Sacco D., Pelissetti S., Dinuccio E., Balsari P.,Rostami M, Grigniani C. 2012. Laboratory assessment of ammonia emission after soil application of treated and untreated manures. J. Agr. Sci. (150) 65-73

[6] Vandré, R., Kaupenjohann, M., 1998. In Situ Measurements of Ammonia Emissions from Organic Fertilizers in Plot Experiments. Soil Sci. Soc. Am. J. 62, 467-473

TC-O_07 Model-based quantification of indirect nitrous oxid emissions in a crop rotation with biogas digestate-fertilized OSR

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1. Objectives

Emissions of nitrous oxide (N_2O) from arable land are a major contribution to the global GHG balance. Direct N_2O emissions can be measured directly at the place of origin. Indirect N_2O emissions result from reactive nitrogen compounds like ammonia (NH_3) or nitrate (NO_3^-). Fertilizing oilseed rape (OSR) with biogas digestates may lead to high amounts of NH_3. The aim of the present work was to quantify indirect N_2O emissions through a model-based N balance approach.

2. Methodology

Starting in autumn 2012, a field experiment including a crop rotation (OSR, wheat, barley) was carried out for two years in northern Germany. In spring OSR was fertilized with biogas digestate (180 kg NH_4-N ha^{-1}, without and with nitrification inhibitor (NI); split application via trail hoses). As NI Piadin was used (SKW Piesteritz; Germany; application rate: 6 l*ha^{-1}). Wheat and barley were fertilized using mineral fertilizer (CAN; 220 kg N ha^{-1} and 200 kg ha^{-1}respectively). Ammonia emissions were measured following digestate fertilizing events by use of the Draeger tube measurement technique [1]. Direct N_2O emissions were measured weekly using manual chambers. Soil mineral N (SMN) samples were taken weekly (depth: 0-30cm) and three times per growing season (depth: 0-90cm), respectively.

To quantify indirect N_2O emissions we used measured data and a dynamic simulation model to calculate N leaching. SMN data at the beginning of the experiment were used to initialize the model-based N balance approach. As N input fertilization values from the experiment and modelled mineralization were used. System N output via plant N uptake was modelled. Gaseous N losses from the system via ammonia volatilization and N_2O emission were measured; emissions of NO_x and N_2 were modelled. Finally we calculated indirect N_2O emissions resulting from ammonia volatilization and N leaching using corresponding IPCC TIER 1 emission factors [3].

Two experimental periods were analyzed (2013 = summer 2012 to spring 2014 and 2014 = summer 2013 to spring 2015).

3. Results and discussion

3.1 Sources of indirect N_2O emissions

3.1.1 Ammonia volatilization

Ammonia volatilization of OSR fertilized with biogas digestate did not differ in both experimental periods, whether NI was used or not. Measured total NH_3 emissions for both treatments were 45 and 42 kg NH_3-N ha^{-1} after fertilizer application in spring 2013 and spring 2014, respectively. They account for 25% (2013) and 21% (2014) of total NH_4-N applied. Quakernack et al [4] found comparably high NH_3-N losses related to applied NH_4-N in wheat and maize after digestate application.

Compared to total N applied with digestates, relative NH_3 losses of 16.2% and 14.6% occurred in the first and the second experimental period, respectively. This is somewhat lower than the IPCC default value of 20% [3]. Wolf et al [5] also measured no differences the in relative NH_3 losses between digestate with or without NI addition (13.1% (+NI) and 13.7% (-NI) of total digestate-N applied via drag-hoses to maize). Our results agree with Wolf et al [5] that there is no effect of NI on NH_3 emission.

3.1.2 N leaching

To quantify N losses of both treatments via NO_3^- leaching, a dynamic simulation model was used for both experimental periods. Each simulation run started at harvest of the preceding crop (barley) before OSR and ended before stem elongation of the following wheat. The simulation model consists of dynamic crop growth models for OSR and wheat describing the N uptake by the respective crop. The crop growth models were able to reproduce measured N uptake (shown for OSR, figure 1). Mineralization of N was calculated by a 5 pool coupled C and N model. Input of organic matter (crop residues at harvest, senescent crop components of OSR and organic component of biogas digestates) was distributed to the decomposable plant material (DPM) and the resistant plant material (RPM) pools. The turnover rate for soil organic matter (SOM) was estimated to fit measured SMN. The other turnover rates were taken from Hansen et al. [2]. Nitrification rate was assumed to be lower for the NI treatment. Nitrate movement in the soil profile was calculated by a dispersion convection model combined with a layered soil water model using the water content based solution of the Richards equation. Transport below the rooting zone of OSR (measured and simulated to be up to 1.8 m in the loamy soil at the experimental site) was considered as NO_3^- leaching.

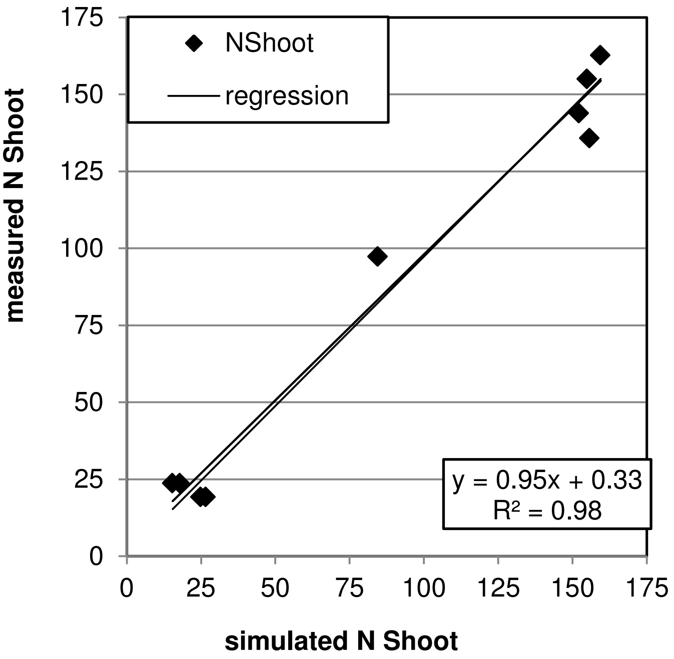

Figure 1: Regression of simulated and measured nitrogen taken up by above-ground OSR components (N Shoot in kg N ha^{-1}).

The N uptake of OSR was about 190–200 kg N ha^{-1} in both periods and both treatments. Under our model assumptions the NI treatment caused a delay of mineralization resulting in higher SMN amounts at OSR harvest (about 60 kg N ha^{-1} compared to about 30 kg N ha^{-1} without NI in both periods). This led to higher N uptake of the following wheat in NI treatments. Simulated total mineralization amounted to 90-110 kg N ha^{-1} with slightly lower values in NI treatments. During the leaching period over winter SMN in the soil profile was generally low due to the N uptake of the crops in autumn (Except for NI treatment in 2013 where leaching was prevented by unusually low precipitation). As well the mineralized N from the preceding crop residues as from the crop residues of OSR was thus taken up. Fertilization events increased SMN temporarily which was taken up quickly by the growing OSR.

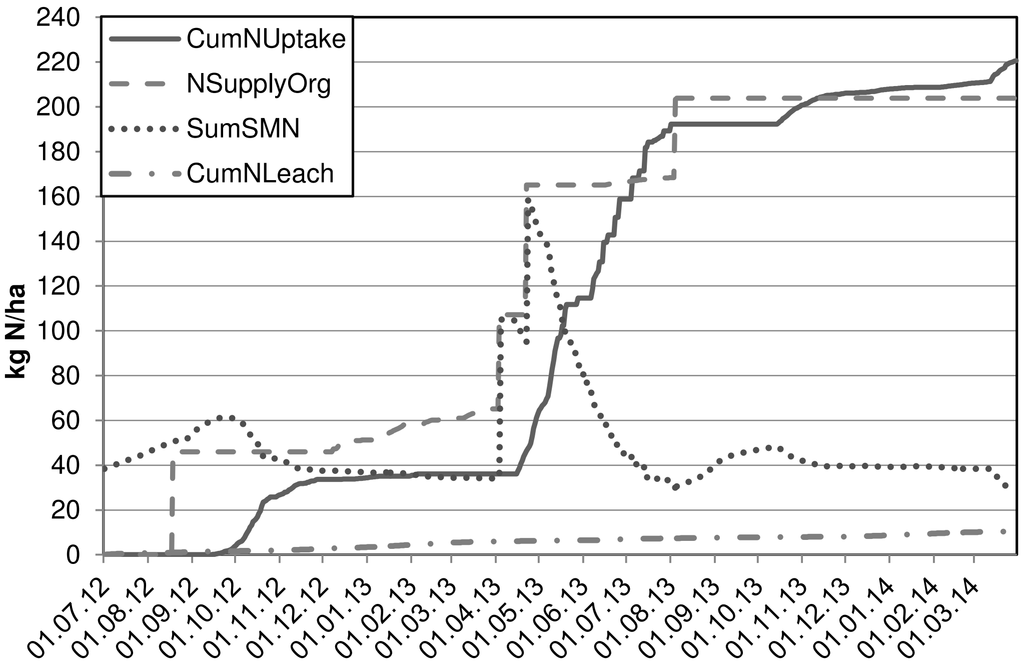

Figure 2: Model-Output of N balance approach for biogas digestate- fertilized OSR without NI in 2013. (CumNUptake = cumulated N uptake by OSR and wheat; NSupplyOrg = cumulated supply of N via organic matter; Sum SMN = time course of mineral N in soil profile (0-1.8 m); CumNLeach = Sum of leached NO_3^{2-}-N)

In the first period, 10.5 kg NO_3^--N ha^{-1} were leached in the treatment without NI (Fig.2) and 11.6 kg NO_3^--N ha^{-1} were lost in the NI treatment. Approximately half of these losses were calculated for the second period (5.5 kg NO_3^--N ha^{-1} and 5.5 kg NO_3^--N ha^{-1}, respectively). This equates to around 3% of total N input in 2013 and 1.5% in 2014, respectively, which is considerably lower than the IPCC default value of 30% [3].]. It could be an effect of year, because there were higher amounts of leached N measured in the past (not shown) There was no observable effect of NI usage to N leaching in both experimental periods.

3.2. Indirect N_2O emissions

Indirect N_2O emissions from NH_3 volatilization were calculated to be 0.45 kg N ha^{-1} and 0.42 kg N ha^{-1} in 2013 and 2014, respectively. Indirect N_2O emissions from NO_3^- leaching were calculated to be 0.08 kg N ha^{-1} and 0.04 kg N ha^{-1} in 2013 and 2014, respectively (Table 1).
In both periods indirect N_2O originating from NH_3 volatilization was the main contribution to total indirect N_2O emissions due to the low N loss via nitrate leaching resulting in low indirect N_2O emissions from NO_3^-.

Table 1: Calculation of indirect N_2O emissions for 2013 and 2014 with corresponding IPCC TIER 1 emission factors for indirect emissions from ammonia volatilisation (EF_NH_3) and from nitrate leaching (EF_NO_3^{2-}).

Experimental period	**2013**		**2014**	
Treatment	BD-NI	BD+NI	BD-NI	BD+NI
NH_3 loss [kg N ha^{-1}]	45.4	45.4	42.2	42.2
NO_3^- leaching [kg N ha^{-1}]	10.5	11.6	5.5	5.5
EF_NH_3 (%)	1.0	1.0	1.0	1.0
Indirect N_2O from NH_3 [kg N ha^{-1}]	0.45	0.45	0.42	0.42
EF_NO_3^- (%)	0.75	0.75	0.75	0.75
Indirect N_2O from NO_3^- [kg N ha^{-1}]	0.08	0.09	0.04	0.04
∑ indirect N_2O [kg N ha^{-1}]	**0.53**	**0.54**	**0.46**	**0.46**

For the periods when direct N_2O emissions were measured (December 2012 to March 2014 and September 2013 to December 2014, respectively) indirect N_2O emissions accounted for 26% to 44% of total N_2O emissions (Table 2). Direct N_2O emissions were on a generally low level during the investigated periods. Also Wolf et al [5] found low direct N_2O emissions after biogas digestate application with and without NI resulting in high proportions of indirect N_2O emissions originating from NH_3 volatilization.
In 2014 NI prevented high N_2O emissions after biogas digestate application which led to considerable differences in total direct emissions.

Table 2: Relation of indirect emission of N_2O to direct emission.

Experimental period	12.2012-3.2014		9.2013-12.2014	
Treatment	BD-NI	BD+NI	BD-NI	BD+NI
∑ direct N_2O [kg N ha^{-1}]	0.64	0.69	1.27	0.76
∑ indirect N_2O [kg N ha^{-1}]	0.51	0.52	0.44	0.44
Indirect N_2O: % of total N_2O	44	43	26	37

4. Conclusion and outlook

Fertilizing OSR with biogas digestate leads to relatively high amounts of indirect N_2O emissions originating mainly from ammonia volatilization compared with measured direct N_2O emissions regardless of the usage of NI. Nevertheless, the measured NH_3 emissions were lower than the IPCC default value.

Simulated nitrate leaching and indirect N_2O emissions originating from this pathway were generally low. This might be caused by high N uptake in autumn, deep rooting of OSR and low precipitation over winter (2013). Due to low direct N_2O emissions the proportion of indirect emissions (mainly from NH_3 volatilization) of total N_2O emissions was considerably high.

To evaluate the found results further data from other sites and scenario simulations for different years should be analyzed.

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References

[1] Gericke, D.; Pacholski, A.; Kage, H. (2014): Measurement of NH3 emissions in multiplot field experiments, Biosystems Engineering 108,164-173
[2] Hansen, S.; Jensen, H. E.; Nielsen, N. E.; Svendsen.H. (1990): DAISY-Soil Plant Atmosphere System Model. *NPo-forskning fra Miljøstyreisen,* **A10:** 272 pages
[3] IPCC (1997): Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change/Organization for Economic Cooperation and Development, OECD/OCDE, Paris
[4] Quakernack, R.; Pacholski, A., Techow, A.; Herrmann, A.; Taube, F.; Kage, H. (2012): Ammonia volatilization and yield response of energy crops after fertilization with biogas residues in a coastal marsh of Northern Germany, Agriculture Ecosystems & Environment, 160, 66-74
[5] Wolf, U.; Fuß, R.; Höppner, F.; Flessa, H. (2014): Contribution of N2O and NH3 to total greenhouse gas emission from fertilization: results from sandy soil fertilized with nitrate and biogas digestate with and without nitrification inhibitor, Nutrient Cycling in Agroecosystems 100, 121-134

TC-O_08 Controlled drum composting with limited climate impact – Emissions and heat recovery

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1. Objectives

EU Directive 1069/2009 requires manure to be sanitised before being sold on the market. Approved sanitisation plants use a drum composter to reach compost temperatures of 52°C for at least 13 hours. The objectives of this study were to:

- Determine ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) release from drum composting, including pre- and post-composting stages
- Identify and evaluate measures to reduce gaseous emissions
- Identify cost-effective methods for energy recovery.

2. Methodology

Measurements were made in two different composting plants, both using horse manure complemented either with vegetable residues (10% of total mass) (Plant 1) or other animal manure (40% of total mass; solid phase of separated pig slurry and deep straw litter from beef cattle and sows) (Plant 2). The drum composter (QuantorXL®, ECSAB, Enköping, Sweden) consisted of a rotating drum, with manure transported in the axle direction with added aeration. In both plants, the manure was pre- and post-composted in batch heaps without turnings, duration given in Table 2. It was weighed, sampled and analysed at the start and end of the different composting steps and the compost temperature was measured continuously. The gases NH_3, N_2O and CH_4 were measured from pre-compost, drum compost and post-compost.

- Method 1: When the exhaust air was confined in pipes from the drum composter and from the pre-composting hall as in Plant 1, air flow and gas concentrations were measured on-line by a pressure-based flow meter (Micatrone MFS-SS) and two gas analysers: An ABB Bomem MB100 Fourier Transform Infrared spectrometer (FTIR) and a J.U.M. 3-900 heated FID (Flame Ionization Detector) equipped with a hydrocarbon cutter. The duration of each of these measurements varied between 4 and 8 hours and the sampling period was one minute. Number of sampling occasions is given in Table 2.
- Method 2: For batch composts (post-composts and pre-compost in Plant 2), a micro metrological mass balance method [1], based on exposure of passive flux samplers placed on masts surrounding the heaps, was used for NH_3 measurements. For CH_4 and N_2O, closed chamber techniques were applied repeatedly during composting [2] with six and five samplings for the post-composts on Plant 1 and 2, respectively and on Plant 2 the pre-compost were sampled twice.

Measures for reducing gas emissions studied were: use of a prototype of a simple exchange heater/ammonia trap (type of air scrubber) and for, post-compost heaps, use of a plastic cover. The potential for heat recovery in the exhaust air was also determined, by calculations based on exhaust air temperature, air flow and relative humidity. Climate data for a normal year in the region were used for calculations.

3. Results and discussion

3.1 Composition of compost

Compost properties during the three composting steps in Plants 1 and 2 are shown in Table 1. In both plants, the compost had about 32% dry matter (DM) content. The compost in Plant 2 had a higher N content than that in Plant 1, but about the same Total-C content, giving a lower C/N ratio in Plant 2. The greatest changes in concentration occurred during the post-composting period, which lasted much longer than the preceding pre-composting and drum composting steps.

Table 1: Properties of the manure in each stage of the composting process in Plants 1 and 2.

Stage in compost plant	DM content (%)	Total-N (kg tonne^{-1})	NH_4-N (kg tonne^{-1})	Total-C (kg tonne^{-1})	C/N
Plant 1					
IN pre-composter (17/18 March)	31.1	4.1	0.8	142	35
OUT pre-compost/ IN drum composter (19-21 March)	32.3	4.3	0.9	141	33
OUT drum composter/ IN post-composter (24 March)	33.1	3.7	0.9	135	37
OUT post-composter (4 July)	38.8	5.8	0.3	119	21
Plant 2					
IN pre-composter (10 September)	32.6	6.3	1.3	141	22
IN drum composter (17 September)	31.2	7.0	1.7	133	19
OUT drum composter = IN post-composter (18 September)	33.1	6.5	1.2	122	19
OUT post-composter (2 December)	26.0	6.9	0.1	105	15

3.2 Gas emissions

In general, emissions of CH_4-C, N_2O-N and NH_3-N were limited and rather low compared with literature [3] or default values. The main greenhouse gas (GHG) was CH_4 originating from the compost piles (Table 2). In Plant 1, pre-compost was the dominant source, while in Plant 2 the main CH_4-C release originated from post-compost.

In Plant 1, the pre-compost heaps were piled indoors, while in Plant 2 they were stored outdoors. This may have limited the oxygen present in Plant 1, stimulating more anaerobic conditions and CH_4 emissions. Otherwise, higher temperature and longer retention time for the pre-compost in Plant 2 compared with Plant 1 could have be the reason for the opposite result.

In both plants, most N_2O was produced during post-composting, which was the longest period (Table 2). A plastic cover reduced N_2O emissions efficiently and also CH_4 emissions, as reported previously [3]. In total, a plastic cover reduced the global warming potential (GWP_{100}) by about 33% for the post-composting step and overall by 27% for all three composting steps.

GHG release from the drum compost contributed a very small part of the total, amounting to about 1.3-4.5% of the total impact in CO_2e. As a consequence, implementation of measures to reduce GHG emissions from the drum compost was not considered meaningful.

Total NH_3 emissions in Plant 1 from the three composting steps comprised ~3% of Total-N in substrate, which is low compared with 11% of total-N from frequently turned compost of horse manure with straw (Swedish data, not shown). Post-composting gave the highest NH_3 emissions in Plant 1 (1.4-2.2% of total N), while in Plant 2, with a higher content N in compost, losses from pre-compost were highest. In total for the three steps, NH_3-N losses in Plant 2 were about 5% of total-N. Losses from the drum composter in Plant 1 were only 0.3-0.6% of total-N, corresponding to ~290 kg N per year.

A simple combined NH_3 trap and heat exchanger reduced NH_3 release by about 17 and 35% of the NH_3 discharged from the drum composters in Plants 1 and 2, respectively.

Table 2: Emissions of methane (CH_4-C), nitrous oxide (N_2O-N) and ammonia (NH_3-N), expressed in g tonne^{-1} compost and as global warming potential (GWP_{100}) in CO_2e per tonne manure from the three individual compost steps. Step duration, mean temperature in manure and volume weight of compost heaps (start-end of period; per compost/measurement) are also shown.

		Emissions (g tonne^{-1})			CO_2e	Mean temperature	Volume weight
Compost step[a]	Duration	CH_4-C	N_2O-N[b]	NH_3-N	(kg tonne^{-1})	(°C)	(tonnes m^{-3})
Plant 1							
Pre-compost (n=2)	0.5 week	1087.1	0.28	20.9	36.4	37.3/43.5	0.47
Drum compost (n=3)	About 26 h	18.77	0.02	23.52	0.6	58.4/58.3/60.0	-[c]
Post-compost, no cover (n=2)	3 months	104.9	9.6	66.1	8.3	49/53	0.72-0.75/ 0.68-0.61
Sum[c]		1210.8	9.9	110.5	45.3		
Plant 2							
Pre-compost (n=1)	1 week	32.6	1.16	155.6	2.4	50.5	0.43-0.46
Drum compost (n=4)	About 50 h	0.12	0	13.1	0.8	61.0/59.7/60.2/61.4	-[c]
Post-compost, no cover (n=1)	2 months	536.7	6.36	67.2	21.2	50.3	0.55-0.78
Post-compost, plastic cover (n=1)	2 months	425.0	0.16	-[d]	14.6	46.9	0.55-0.79
Sum[c], no cover		569.4	7.5	235.9	24.4		
Sum[c], plastic cover		536.7	6.4	67.2	21.2		

[a] n= number of measurement occasions (pre-compost at Plant 1 and drum composts, both plants) or number of compost heaps

[b] Indirect N_2O from NH_3 not included

[c] Weight losses in each step not taken into account

[d] Not measured, presumably low

3.3 Economic analysis of investment for heat recovery

Potential energy in exhaust air for use by the heat exchanger was ~54 MWh per year, corresponding to the energy requirement for heating two detached houses in the region.

Depending on present fuel in use, investment in a heat exchanger in the exhaust air flow gave very different payback times (Table 3). When wood chips were used there was little incentive to invest in heat recovery. For oil with no subsidisation, the payback time was 3 years, which is reasonable with a technical life time of 15-20 years.

Table 3: Pay-back time for investment in a heat exchanger of the type studied here, based on 42 MWh being substituted by investing in this equipment. The investment cost is based on information from manufacturers of relevant products.

Present fuel	**Cost for fuel (Euro MWh^{-1})**	**Investment cost (Euro)**	**Payback time (years)**
Oil	147[a]	16,000	3
Oil, subsidised	63[b]	16,000	6
Wood chips	20	16,000	19

[a] Corresponds to a price of 1.5 Euro L^{-1} (prices from the Swedish market)

[b] Corresponds to a price of 0.7 Euro L^{-1} (prices from the Swedish market)

4. Conclusion and outlook

In total, GHG and NH_3 emissions from the three composting steps were moderate, with the drum composter releasing a very small fraction of the total loss. The largest impact on global warming was from CH_4 emitted from the batch composts, which was highest from the pre-compost (Plant 1) or post-compost (Plant 2), and direct N_2O-emissions from post-compost. Covering the post-compost with plastic reduced most of the N_2O emissions and also reduced CH_4 emissions by one third.

It was possible to trap up to 35% of the NH_3 in exhaust air from the drum composter in a simple prototype combining an NH_3-trap and heat exchanger. The climate impact could be reduced mainly by replacing heat from fossil fuel with heat exchanged from exhaust air from the drum composter. The payback time on investment in heat recovery was 19 years when replacing wood chips as fuel, but only three years when replacing fossil oil (no tax subsidy included). Low temperature in the exhaust air from the drum composter made it more suitable for heating buildings close to the plant, which in most cases excludes dwelling-houses.

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References

[1] Schørring, J.K.; Sommer, S.G.; Ferm, M. (1992): A simple passive sampler for measuring ammonia emission in the field, Water, Air, and Soil Pollution, 62, 13-24

[2] Rodhe L.; Ascue J.; Nordberg Å. (2009): Emissions of greenhouse gases (methane and nitrous oxide) from cattle slurry storage in Northern Europe, IOP Conf. Ser.: Earth Environ. Sci. 8, 012019, http://doi.org/10.1088/1755-1315/8/1/012019

[3] Webb, J.; Sommer, S.G.; Kupper, T.; Groenestein, K.; Hutchings, N.J.; Eurich-Menden, B.; Rodhe, L.; Misselbrook, T.H.; Amon, B. (2012): Emissions of ammonia, nitrous oxide and methane during the management of solid manures, Sustainable Agriculture Reviews 8, 67 – 107

TC-O_09 Emission patterns of separated digestate obtained from field pilot-scale stores

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1. Objective

This paper reports results of research to determine carbon and nitrogen losses from storage of a digestate and its separated fractions, as well as to identify the effect of disturbance actions on gaseous emissions from the materials. This information is needed to improve the precision of emission inventories, which usually are based on standard methodology with default emission factors, and are not country-specific. Furthermore emission inventories consider only the origin of organic wastes, and not the treatment effects applied to them.

2. Methodology

Digested slurry (digestate) was collected from a biogas plant (250 kW) located on a commercial dairy farm (Lodi Vecchio, Italy). The anaerobically co-digested (**UN**) slurry (90% cattle slurry, 10% corn silage) was subjected to mechanical separation with roller presses that generated a liquid (**LF**) and solid fraction (**SF**).

Eight cylindrical tanks were filled with liquid slurries (UN and LF) (operative volume: 0.8 m^3), while four square containers were filled with 250 kg SF each. Each type of material was investigated under disturbed (**D**) and undisturbed treatments (**ND**) in duplicate (total of 12 containers). For the D treatment, UN and LF were thoroughly stirred with a mixer for about 6 min, once a week to simulate the filling and emptying processes typical in full-scale storage. SF samples were mixed manually. The experiment was conducted from 8th of May to 31st of July 2014 (90 days). ND samples were untouched once containers were filled.

Gaseous emissions were measured once every 2 weeks using a dynamic chamber method consisting of funnel systems [1] placed on the surface of samples. This system was comprised of a PVC funnel covering 0.071 m^2 of surface area, fixed on four PVC spherical floats. Airflow through the funnel system was approximately 9 L min^{-1}. NH_3 was captured using acid traps and, greenhouse gas (GHG) including carbon dioxide (CO_2) concentrations were measured using a photoacoustic gas analyser as in [2].

To evaluate the emissions of GHG in terms of CO_2 equivalents, conversion factors of 298 and 25 for a 100-year time horizon were used for nitrous oxide (N_2O) and methane (CH_4), respectively [3]. Emissions from separated fractions (LF and SF) were mathematically combined (TR) to compare the emissions from UN and TR digestate. The combined fluxes (F_{TR}) were calculated using equation 1:

$$F_{TR} = X_1 * F_{LF} + X_2 * F_{SF} \qquad \text{Eq.1}$$

in which F_{LF} and F_{SF} are the fluxes for the liquid and solid fractions, respectively, and X_1 and X_2 are the mass separation efficiencies for liquid (88 %) and solids(12 %), respectively.

The emission factors were compared with those indicated by the IPCC [4]. In particular, for CH_4 and N_2O, emission factors were calculated using Tier 2 and Tier 1 methods, respectively. Statistical analysis was performed using SAS Software for Windows (SAS version 9.4, SAS Institute Inc., Cary, NC). Fluxes (NH_3, CO_2, N_2O and CH_4) were compared between treatments. The data were not normally distributed within equal variance (Kolmogorov–Smirnov and Shapiro–Wilk tests, α = 0.05). Accordingly, Wilcoxon Signed Ranks non-parametric analysis was carried out to determine significant differences for treatment (UN vs. TR). P values less than 0.05 were considered to be statistically significant in all tests (**P < 0.05; *** P < 0.01).

3. Results and discussion

The period considered was on average characterized by the following temperature, wind speed and relative humidity: 22 ± 5.3 °C, 0.37 ± 0.42 m s^{-1} and 72.6 ± 20%, respectively. The total amount of rain during the experiment was 285 mm, approximately 50 mm above the long term average.

3.1 Ammonia and nitrous oxide emissions

Figure 1 shows the NH_3 and N_2O emissions. As in other studies [5, 6] the SF samples showed remarkable NH_3 fluxes only at the beginning of the storage period, with peaks in excess of 200 mg m^{-2} h^{-1}. This can be related to the physical condition of the heaps and to the climatic conditions during sampling. Both very abundant precipitation and very dry conditions in the heap are unfavourable for NH_3 emissions. On average, NH_3 fluxes from UN and LF samples were 60 and 51 mg m^{-2} h^{-1}, respectively, and the differences were more marked after mixing. Disturbance enhanced NH_3 emission. Among samples that were mixed, UN emitted more NH_3 than TR. When a sample is open to atmospheric exchange, the sample's N content is one of the most important factors driving NH_3 emission [6].

Remarkable fluxes of N_2O were observed from both ND (SF_ND) and D (SF_D) SF samples, with peaks during the 14th day of storage (SF_ND: 16.91 ± 9.73 mg m^{-2} h^{-1}; SF_D: 11.1 ± 0.14 mg m^{-2} h^{-1}). Fluxes from the SF were expected to be relatively high because favourable (aerobic) conditions for N_2O production existed inside the heaps [5]. In contrast LF and UN remained mainly anaerobic; thus nitrification processes were practically absent [2]. However some N_2O fluxes were observed from the ND digestate samples (UN_ND); these emissions may be explained by the presence of a superficial crust that formed during storage. Recent studies have shown an increased dry matter content of organic slurries may promote N_2O emissions; in particular, stored slurry with a natural crust may be a source of N_2O emissions [7]. UN digestates developed a thicker and drier crust than did the LF, providing a better aerobic-anaerobic interface that enhanced N_2O generation [7].

3.2 Methane and carbon dioxide emissions

Average CH_4 emissions were 70.7, 38.6, 4.5 mg m^{-2} h^{-1} for UN, LF and SF samples, respectively (Figure 2). Solid-fraction samples always showed negligible CH_4 fluxes, presumably because of restricted oxygen, temperature and heap compaction. In other studies stores of solid manures have been shown to be a source of CH_4 emissions [2] but, in our study, it was not very evident. CH_4 emissions from the liquid samples varied with volatile solids (VS) content, thus digestate samples (highest VS) showed the highest fluxes. Furthermore from Figure 2 it can be seen that CH_4 emissions followed trends in temperature (except for day 56). The CH_4 peak emission (224 mg m^{-2} h^{-1}) occurred on the day of the highest temperature.

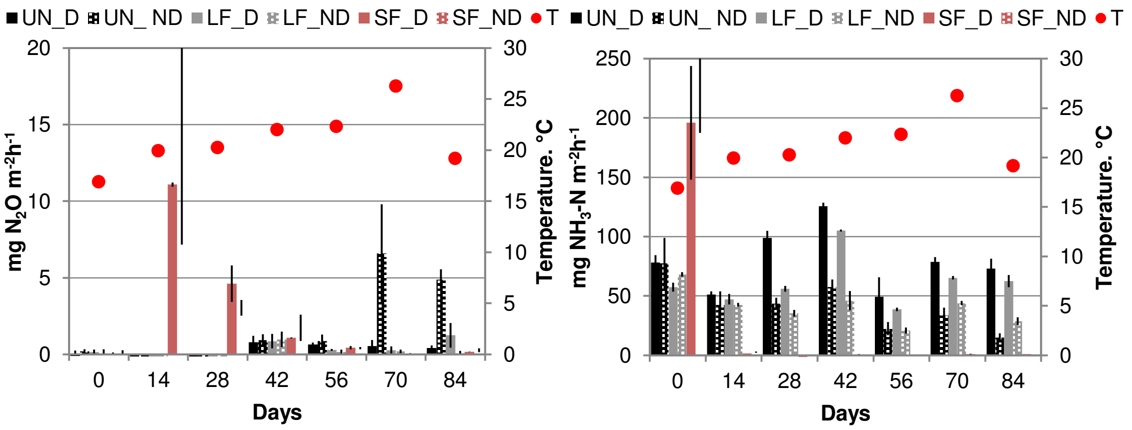

Figure 1: Net fluxes of NH_3 and N_2O (mg m^{-2} h^{-1}) from digestate (UN) and its liquid (LF) and solid (SF) fractions in undisturbed (ND) and disturbed (D) storage conditions. Errors bars represent standard deviation. The red points described the mean temperature recorded during the sampling day.

Figure 2 also shows that CO2 emissions during storage were characterized by a high variability without clear trends. Average CO2 fluxes were 635.6, 541.2, 787.25 mg m-2 h-1 for UN, LF, and SF samples, respectively. CO2 was the predominant carbon emission (Table 1), accounting for about 80% of the total. CO2 emissions show a probable relation with carbon content as samples characterized by higher total solids (TS) and VS contents had the higher CO2 emissions. Mixing increased CO2 fluxes.

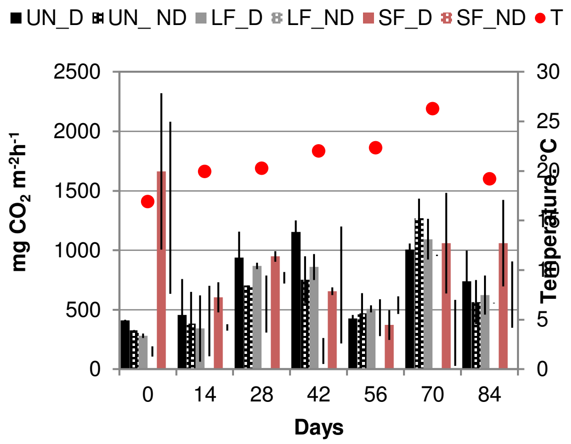

Figure 2: Net emission fluxes of CH_4 and CO_2 (mg $m^{-2}h^{-1}$) measured from digestate (UN) and its liquid (LF) and solid (SF) fractions in undisturbed (ND) and disturbed (D) storage conditions. Errors bars represent standard deviation. The red points represent the mean temperature recorded during the sampling day.

3.3 Effect of mechanical separation, disturbance and season

While C losses mainly occurred as CO_2 release, and to a lesser extent as CH_4, N losses occurred almost entirely (more than 99.5%) as NH_3 (Table 1). Mechanical separation had no significant effect on NH_3 emission, probably because a crust developed on both digestate (UN) and its LF. Thus, differences in NH_3 emission that were expected due to different chemical characteristics of samples were not expressed. In contrast, mechanical separation resulted in significantly lower CH_4 (40%) and CO_{2eq} emissions. LF was the greater contributor (75%) to CO_{2eq} emission. Warm temperatures during storage stimulated methanogenic bacteria which, within UN digestate, found a better environment because of the major amount of easily degradable organic matter. Concerning CO_2 emissions mechanical separation had a significant effect. Emissions from D and ND separated fractions were 50% and 30% higher than that from the respective unseparated digestates, due to the solid fraction rich in carbon and characterized by high CO_2 emissions.

Table 1: Flux increments and reductions due to mechanical separation and disturbance.

	Untreated (UN)		**Separated (TR)**		**Coefficient of variation due to mech. sep. (%)**		**Coefficient of variation due to disturbance (%)**	
	D	**ND**	**D**	**ND**	**D**	**ND**	**UN**	**TR**
N-NH_3, (g t^{-1} d^{-1})	2.14 (0.11)	1.08 (0.17)	1.86 (0.12)	1.4 (0.5)	-13.1	29.3	97.7***	32.8*
N_2O, (g t^{-1} d^{-1})	0.02 (0.01)	0.05 (0.02)	0.05 (0.37)	0.05 (0.01)	457***	6.6*	-82.1**	-7.4
CH_4, (g t^{-1} d^{-1})	2.45 (1.16)	3.27 (0.71)	1.47 (0.06)	1.85 (0.1)	-39.9***	-43.4*	-25*	-20.4**
CO_2 (g t^{-1} d^{-1})	21.0 (3.4)	18.4 (1.2)	31.7 (1.2)	23.9 (0.1)	50.5***	30.1***	14.6	32.6***
CO_2 eq (g t^{-1} d^{-1})	64 (31)	97 (23)	52 (2)	63 (10)	-19.0***	-33.9	-35.4***	-17.1**

* Mean difference with P < 0.10; ** Mean significant difference with P < 0.05; *** Mean significant difference with P < 0.01. D (disturbed); ND (undisturbed); UN (unseparated digestate); TR (mathematically combined separated fractions). Standard deviations are given in brackets.

Disturbance affected all gaseous emissions that were examined, but in particular NH_3 emissions. The development of a crust on both UN digestate and its LF effectively reduced the NH_3 emissions from ND samples. The average NH_3 emission from D digestate (UN) was 2.14 g t^{-1} d^{-1} while the combined emission (TR) was 1.86 g t^{-1} d^{-1}. NH_3 emissions from undisturbed samples (UN_ND and TR_ND) were 50% and 25% lower, respectively. Disturbance had a significant effect on N_2O emissions only in the case of digestate; N_2O emissions from UN_D were 82 % lower than those from UN_ND. This result highlights the importance of the condition of the superficial layer on stores of organic materials in either favouring or retarding N_2O production. CH_4 emissions from ND samples were always higher than from D samples (Table 1); the crust that formed is believed to have enhanced methanogenic activity and acted as sink of CH_4 [6]. On a CO_2eq basis disturbance provided a significant reduction in emissions of GHG from both UN ($P < 0.05$) and TR samples ($P < 0.01$).
The IPCC [4] gives default N_2O emission factors, expressed in kg N_2O- N per kg N excreted, for liquid/slurry systems of: 0 for storages without crust formation; 0.005 for storages with crust formation; 0 for anaerobic digesters and 0.005 for storage of solids.. Except for those from solid fractions, our results are in general accordance with the default IPCC emission factors. SF samples had emission factors between 0.02 - 0.03 kg N_2O- N per kg N and were more similar to the IPCC default emission factors for composting-passive windrow (0.01 kg N_2O- N per kg N excreted) [4]. Emissions from the SFs suggested a probable establishment of composting processes within the heaps. The CH_4 default emission factor calculated using the Tier 2 method of IPCC (based on the VS content of the digestate used in the present study) is 4.66 g t^{-1} d^{-1}, a value 43% higher than the emission factor observed from undisturbed digestate. This is most likely because anaerobic digestion has already used the more easily degradable part of organic matter.

4. Conclusions and outlook

The methodology used in this study allowed a comparative analysis of gaseous emissions from co-digested cattle slurry under field conditions. On average 99.5% of nitrogen emissions were in the form of NH_3 and 80% of carbon emissions were in form of CO_2.
Mechanical separation of digestate had an overall positive effect in reducing GHG emissions. In summer conditions, the increase in NH_3 emission after mixing of material probably is limited in time because the crust tends to reform very quickly after disturbance. Furthermore, periodic disturbance of digestate (or its separated fraction) was shown to be an effective way to reduce GHG emissions, which are very influenced by the state of the superficial layer and in particular by the presence of a crust. However to confirm these findings, studies that continuously monitor emissions during the storage period are necessary. Nevertheless, our results underline the importance of adopting mitigation techniques to control gaseous emissions from organic materials, especially during the warm season when the major part of emissions are expected. In particular considering the separated fractions of digestate, the liquid fraction (stored without disturbance) was responsible for 95% and 75% of NH_3 and CO_{2eq} emissions (i.e., 70 - 80% CH_4), respectively. This means that if mitigation techniques have to be adopted these should be directed especially to the storage of the liquid fraction.

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References

[1] Balsari, P., Airoldi, G., Dinuccio, E., Gioelli, F., 2007. Ammonia emissions from farmyard manure heaps and slurry stores-Effect of environmental conditions and measuring methods. Biosyst. Eng. 97 (4), 456 - 463.
[2] Perazzolo, F., Mattachini, G., Tambone, F., Misselbrook, T., Provolo, G., 2015. Effect of mechanical separation on emission during storage of two anaerobically codigested animal slurry. Agric. Ecosyst. and Environ. 207, 1-9.
[3] IPCC, et al., 2007. Climate change. The physical science basis. In: Salomon, S. (Ed.), Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.
[4] IPPC, 2006. Guidelines for national greenhouse gas inventories. Agriculture forestry and other land use. Intergovernmental Panel on Climate Change, 4. IGES, Hayama, Kanagawa, Japan.
[5] Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B., Misselbrook, T., 2011. Manure management: implication for greenhouse gas emissions. Anim. Feed. Scien. and Techn. 166 – 167, 514 – 531.

[6] Hansen, M.N., Henriksen, K., Sommer, S.G., 2006. Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: effects of covering. Atmos. Environ. 40, 4172–4181.
[7] Wood J.D., Gordon, R.J., Wagner-Riddle, C., Dunfield, K.E., Madani, A., 2012. Relationships between dairy slurry total solids, gas emissions, and surface crusts. J. Environ. Qual. 41, 694-704.

TC-O_10 Reduction of gaseous emission from slurry storage tanks by different covering materials

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1. Objectives

Slurry stores are potential sources of odor and atmospheric pollutants, namely ammonia (NH_3) and greenhouse gases (GHG). Covering slurry storage tanks is one of the possible solutions for the reduction of such emissions. The objective of the research was to investigate the potential of four different floating covering systems (100 mm Leca® balls layer, Hexa Cover Tiles®, Floating Joint Elements and a Membrane for biogas recovery) to reduce NH_3 and GHG (nitrous oxide, N_2O; methane, CH_4) emissions during storage of pig slurry.

2. Methodology

Three trials (Table 1) were carried out at a pig-fattening farm in Cuneo, Piemonte (Italy).

Table 1: Experimental layout.

Trial (n)	Tested covering systems	Environmental condi-	Trial duration (days)
1	100 mm Leca® balls layer Hexa Cover Tile® Floating Joint Elements (Ecomembrane®)	spring–summer	180
2	Membrane for biogas recovery (Ecomembrane®)	spring–summer	90
3	Membrane for biogas recovery (Ecomembrane®)	winter	90

Each trial was performed using a 250m^3 pilot scale storage tank (9.0m in diameter) filled with 220m^3 of fresh pig slurry. In trial 1 the slurry surface was divided into four identical sectors (ca. 16 m^2 surface each) with the help of a plastic sheet. Three sectors of slurry surface were covered by the tested covering systems, whereas the remaining (fourth) sector of the storage tank was left uncovered (control). In trial 2 and trial 3 the slurry surface was divided into two halves; one half was covered with the system for biogas recovery, while the other half (control) was left uncovered. During the trials, NH_3 and GHG emissions were measured simultaneously, from each sector of the storage tank, three times a week, with three replicates, by means of a set of Funnel systems [1]. The biogas collected by the membrane system for biogas recovery was continuously recorded by a gas meter (SamGas, mod. G2.5). Methane and CO_2 biogas concentrations were measured and recorded weekly by means of a portable gas analyzer (Draeger X-AM 7000). Recorded data were normalized to normal litres (LN) (dry gas, T= 0 °C, P= 1013 hPa) according to [2]. At the beginning of each trial, samples of manure were collected for total solids (TS), volatile solids (VS), total nitrogen (TN), total ammonium nitrogen (TAN), and pH analysis, according to standard procedures [3]. Along the trials, the environmental and slurry temperatures were continuously recorded by means of data loggers (Hobo Onset). At the end of the experiment, operative and economic evaluations were carried out, based on the behavior of the covering materials along the trial period. Specifically, a cost-benefit analysis of the four tested covering systems has been carried out according to the parameters reported in Table 2 and to a case study finishing pig farm (2000 heads, producing approximately 6650m^3 slurry per year and a required storage time of 180 days).

Table 2: Capital cost and estimated life of the tested floating covering systems (data provided by supplying company).

	100 mm Leca® balls layer	Hexa Cover Tiles®	Floating Joint Elements	Membrane for biogas recovery
Capital Cost (€/m^2)	9.50	37.5	32.0	60.0
Estimated life (years)	10	25	10	10

3. Results and discussion

The main chemical characteristics of the slurry samples collected at the beginning of each of the three trials are listed in Table 3.

Table 3: Main chemical characteristics of the slurry at the beginning of the Experiments.

Trial (n)	pH	TS (%)	VS (%)	TN (%)	TAN (%)
1	7.4	4.29	2.95	0.30	0.24
2	7.4	4.0	2.60	0.31	0.27
3	7.2	3.8	2.40	0.35	0.28

TS-total solids; VS-volatile solids; TN-total nitrogen; TAN-total ammonium nitrogen

In agreement with other studies [1, 4], emissions of all investigated gases were positively correlated ($p<0.05$) with environmental and slurry temperature (Table 4). Specifically, average NH_3, N_2O and CH_4 fluxes from uncovered slurry surface resulted, respectively, up to 5, 4 and 3 times higher in summer than winter conditions.

Table 4: Average ammonia and GHG emissions from uncovered slurry and total biogas yields recorded during the trials.

Trial conditions	Average slurry temperature (°C)	Average emissions ($g\ m^{-2}\ day^{-1}$)			Biogas yield ($l_N\ kgVS^{-1}$)	Methane yield ($l_N\ kgVS^{-1}$)
		NH_3	N_2O	CH_4		
Spring-summer	22.8 (12.4-27.1)	0.95 (0.69-2.61)	0.03 (0.00-0.06)	32.2 (1.32-156)	181	125
Winter	5.73 (3.37-9.76)	0.19 (0.07-0.36)	<0.01 (0.00-<0.01)	12.3 (0.63-54.6)	69.0	47.0

TS-total solids; VS-volatile solids; TN-total nitrogen; TAN-total ammonium nitrogen

Biogas yield recorded from the slurry surface covered by the membrane for biogas recovery also varied seasonally, with the least yields occurring in winter (Table 4). On average, during the trials, approximately 4.7 m^3_N of biogas was collected daily from the covered slurry surface (ca. 32 m^2), which corresponds to a daily production of 0.147 m^3_N biogas m^{-2} and approximately 0.04 m^3_N biogas m^{-3} of slurry stored into the tank. The average methane concentration of the biogas was 63.5% (range 61.5%-66.1%). Total methane yield ranged from 125 (spring-summer conditions, Trial 2) and 47 (winter conditions, Trial 3) $l_N\ kgVS^{-1}$ (Table 4), values that agreed well with those reported by [5] from different farm scale pig slurry storage facility in Denmark over a 1 year timeframe. However, the average methane yield recorded during the trials was much lower than the potential yield value of 200 $l_N\ kgVS^{-1}$ of stored pig slurry reported by [6]. All the covering systems showed to be effective in reducing ammonia emission (Figure 1).

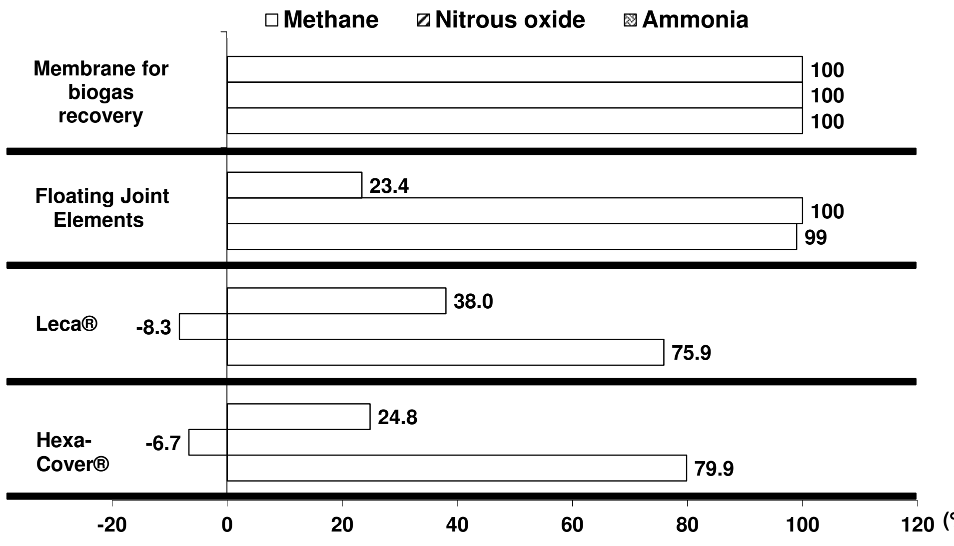

Figure 1: Percentage reduction in total ammonia and GHG emissions compared with uncovered slurry (control).

Compared to the control (uncovered slurry), the Hexa Covers Tiles® and Leca® balls showed approximately a 80% reduction, whereas the 99% of ammonia losses were abated by using the Floating Joint Elements. No ammonia nor greenhouse emission were recorded from the slurry surface covered by the membrane for biogas recovery. Nevertheless, the other floating covering materials were less effective on GHG abatement. In particular, Hexa Covers Tiles®, Leca® balls and Joint Elements significantly ($p<0.05$) reduced CH_4 emissions by 25%, 38% and 23%, respectively. By the floating joint elements N_2O emissions were virtually zero. Hexa Covers Tiles® and Leca® balls did not significantly influence NO_2 emission. However, from the slurry surface covered by the latter materials +7% and +8% NO_2 emission compared to the control (uncovered slurry) were recorded respectively. Berg et al. [7], concluded that permeable cover materials (such as Leca® balls) over the slurry surface may create favorable conditions (e.g., mosaic of anaerobic and aerobic micro-sites over the slurry surface) for N_2O production.
In Table 6 the cost-benefit analysis of the four tested covering systems is presented. Calculation are based on data presented in Table 2, Table 4 (e.g., average methane yields) and Figure 1 (e.g., percentage reduction in total ammonia emissions) and the livestock farm scenario described above. Findings from cost-benefit analysis showed that all the investigated covering systems were economically sustainable but the Floating Joint Elements. Indeed, for the latter covering system a benefit-cost ratio (B/C) lower than 1 and a negative net present value (NPV) was calculated (Table 6), mainly due to the high specific capital costs (Table 1). The Membrane for biogas recovery resulted the best cost effective option, resulting in a B/C ratio of 3.4 and approximately 78000 € in net benefits.

Table 6: Cost-benefit analysis of the four tested covering systems.

Covering systems	NPV (€)	B/C
100 mm Leca® balls layer	3414	1.7
Hexa Cover Tile®	-8927	0,6
Floating Joint Elements (Ecomembrane®)	4167	1.3
Membrane for biogas recovery (Ecomembrane®)	77574	3.4

NPV-net present value; B-benefits (economic value [0.72€ kg^{-1}] of avoided nitrogen ammonia emissions; economic value [0.70€ m^{-3}] of methane potentially recoverable by the Membrane for biogas recovery); C-costs (annual budget allocation for replacement of the covering systems)

4. Conclusion and outlook

Experimental results confirm that covers can be an effective method for mitigating ammonia and GHG emissions from slurry storage. Considering the gaseous losses abatement efficiency, investment costs and handling easiness, the slurry tank covered with the membrane for biogas recovery resulted to be the most suitable system. Besides the environmental benefits, it also enables the energetic valorization of the recovered methane, thus increasing the economic sustainability of this solution (e.g., payback time shorter than 4 years according to specific costs and benefit items considered in the study). All the covering systems showed to be persistent over time, thus suggesting the possibility to re-use them several times. However, the permeable cover materials (Leca® balls layer, Hexa Cover Tiles®) showed to lose their effectiveness in some circumstances (e.g., when a natural surface crust formed on the slurry). Neither of the tested options allows for vigorous agitation.

References

[1] Balsari, P.; Airoldi, G.; Dinuccio, E.; Gioelli, F. (2007): Ammonia emissions from farmyard manure heaps and slurry stores - effect of environmental conditions and measuring methods, Biosystems Engineering 97, 456-463
[2] VDI 4630, (2006): Fermentation of organic materials, Characterisation of Substrate, Sampling, Collection of Material Data, Fermentation Tests, VDI Gesellschaft Energietechnik
[3] AOAC, (2000): Official Methods of Analysis, 15th ed. Association of Official Analytical Chemists, Arlington, VA, USA
[4] Dinuccio, E.; Berg, W.; Balsari, P. (2008): Gaseous emissions from the storage of untreated slurries and the fractions obtained after mechanical separation, Atmospheric Environment 42, 2448–2459
[5] Husted, S. (1994): Seasonal Variation in Methane Emission from Stored Slurry and Solid Manures, Journal of Environmental Quality 23, 585-592
[6] Sommer, S.G.; Petersen, S.O.; Møller, H.B. (2004): Algorithms for calculating methane and nitrous oxide emissions from manure management, Nutrient Cycling in Agroecosystems 69, 143–54
[7] Berg, W.; Brunsch, R.; Pazsiczki, I. (2006): Greenhouse gas emissions from covered slurry compared with uncovered during storage, Agriculture, Ecosystems & Environment 112, 129-134

TC-O_11 Influence of surface processes on gaseous emissions from manure slurry – Surface oxidation and pH gradient

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1. Objectives

Emission of various gases from livestock production facilities and manure application has adverse impacts on the environment on several levels [1]: Greenhouse gases released from agriculture play a key role in global warming. Ammonia (NH_3) emissions contribute to eutrophication of the aquatic environment, to the greenhouse effect through formation of nitrous oxide and to formation of harmful particulate matter through gas-particle conversion in the atmosphere. Volatile organic compounds (VOC) contribute to tropospheric ozone formation and certain VOC contribute to odor nuisance associated with intensive livestock production and manure application. Furthermore, inorganic and organic sulfur-containing compounds contribute to odor nuisance and to formation of particulate matter following atmospheric oxidation to sulfate.

Abatement of gaseous emissions from manure slurry requires knowledge about factors influencing the release of ammonia (NH_3), greenhouse gases, VOC and sulfur compounds on a detailed level. Surface pH plays a crucial role for the release of many compounds that are either basic (e.g. NH_3) or acidic (e.g. H_2S or acetic acid) and has been suggested to be different from bulk pH [1], although measurements with high spatial resolution in real manure slurry has not been reported. In addition, chemical or microbial surface oxidation by oxygen penetrating few µm to mm into the slurry surface [2] may reduce emissions of compounds that are easily oxidized.

In this work, a unique combination of pH micro-sensors and online mass spectrometry has been used for the first time to investigate these processes. Furthermore, the effects of slurry stirring and headspace oxygen content on surface pH and on emissions are reported.

2. Methodology

The experiments were performed in a 2 liter reactor containing 1 liter of finisher pig manure slurry. Headspace was continuously flushed with air at a rate of 2 L/min. Slurry mixing was induced by mechanical mixing using a top mounted motor driving a two blade axial impeller. A schematic of the setup is presented in Figure 1.

A pH micro-sensor positioned in a micro-manipulator was used to carry out measurements of pH gradient in increments of 250 µm in the top 0-100 mm of slurry. A 0.5 mm wide pH microelectrode (pH-500, sensitive pH glass 400 – 800 µm; Unisense, Aarhus, Denmark) equipped to external reference electrode (Ag/AgCl, Radiometer analytical, France) and connected to a 4-channel microsensor amplifier (microsensor multimeter, Unisence, Aarhus, Denmark) were used to measured slurry surface pH profiles. Because of placement in a micromanipulator (MM33, Unisence, Aarhus, Denmark) the microelectrode could be positioned in three dimensions with a precision of 100 µm in the x and y-plane. The micromanipulator motorized in the z-plane improved the manipulator precision to 0.50 µm and made it possible to make automatic profiling. The microelectrode could move freely (z-plane) through a channel in the cylinder lid, while the pH reference electrode was fixed in the cylinder lid. Profiles measuring were programmed to measure the top 0 – 100 mm in increments of 0.25 mm. A total of 59 – 86 pH profiles were measured during the experimental trials.

The bulk slurry pH was measured with a pH electrode (VWR pH 100, combination pH/reference electrode, VWR International, USA) inserted through a port in the cylinder wall, though which bulk measurements were done at a fixed depth of 140 mm.

Emissions were measured by proton-transfer-reaction mass spectrometry (PTR-MS) supplemented by photo-acoustic detection. The concentration of volatile compounds in the outlet from the cylinder headspace was measured with both PTR-MS and photo icoustic Infrared spectrometer (PAS). PAS was used as a supplement to the PTR-MS and made it possible to detect CO_2, N_2O and CH_4, which cannot be detected by PTR-MS due to their low proton affinities.

The PTR-MS (Ionicon Analytik, Innsbruck, Austria) was operated under standard conditions for drift tubes with a total voltage of 600 V, a pressure of 2.1 to 2.2 mbar and a temperature of 60 °C, giving an energy-to-density ratio (E/N) of 135 Td. Calibration was carried out as described by Feilberg et al. [4]. Measurements were carried out on cylinder headspace air diluted (1:4) with a zero air gas standard. The PAS system includes a 1312 photo-acoustic multi-gas analyzer (Innova Air Tech Instruments A/S, Denmark). The PAS shifted between measurements of cylinder headspace and background air, it measured in each location for 5 minutes making the total cycle time 10 minutes long.

Measurements were done under stagnant liquid conditions as well as under slurry stirring. The effect of surface oxidation was investigated by exchanging air with N_2 in the headspace.

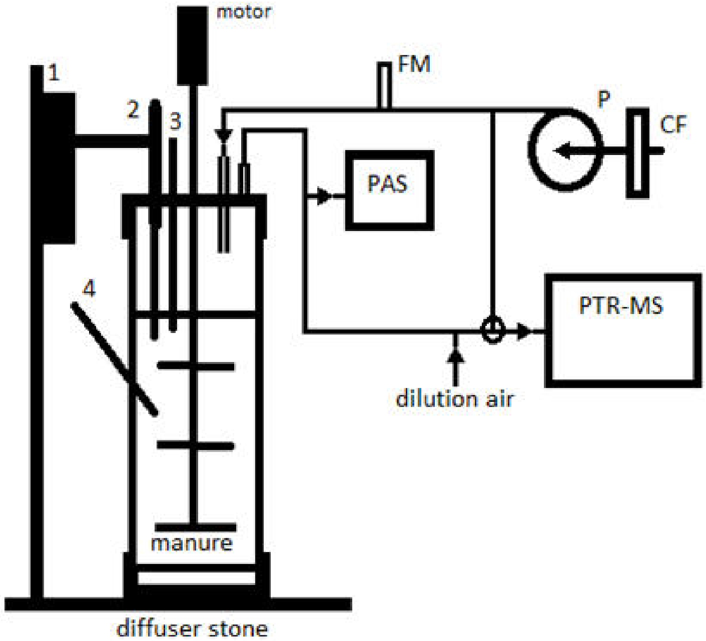

Figure 1: Schematics of the test setup (not to scale). 1 = motor driven manipulator, 2 = 0.5 mm pH sensor, 3 = reference pH sensor, 4 = bulk pH electrode, FM = flow meter, P = pump, CF = charcoal filter, PAS = photo acoustic infrared spectrometer, PTR-MS = proton transfer-reaction-mass spectrometer. PAS and PTR-MS have internal pumps. The pH reference sensor and bulk pH electrode was fixed to the reactor, whereas the pH sensor was free to be moved by the manipulator.

3. Results and discussion

3.1 Slurry stirring and storage.

A strong pH gradient was observed to develop within few hours of slurry storage. The gradient reached a level of ~1 pH unit (~6.8 to ~7.8) over a depth of less than 10 mm and as expected [1] pH was higher near the surface, which is ascribed to differential release rate of CO_2 and NH_3. During stirring of the manure, surface pH approaches bulk pH and becomes more acidic. These results are in excellent agreement with recent data based on an aqueous solution of ammonium bicarbonate in which similar processes take place. The reduction in surface pH is accompanied with reduced emission of NH_3 and increased emissions of acidic components (H_2S, CO_2 and carboxylic acids). Selected results are presented in Figure 2. A more complete breakdown of the surface pH gradient was observed in an additional experiment in which the slurry was flushed by N_2 (data not shown). This flushing results in a more complete mixing of the slurry. However, emissions of H_2S and CO_2 were increased far beyond what can be explained by decreased surface pH, which is ascribed to reduced liquid-side resistance to the flux during stirring in the case of relatively poorly soluble compounds. The same was observed for emission of methane and for methanethiol (both poorly water soluble), which should not be affected by pH at the range observed here (pKa of methanethiol is 10.3).

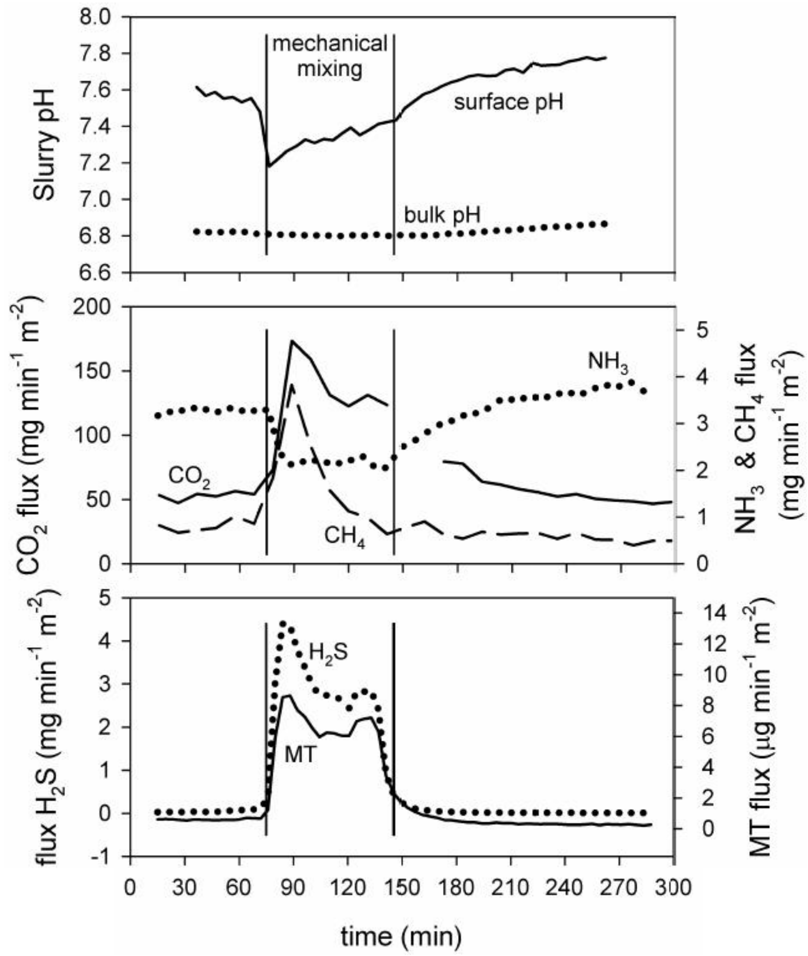

Figure 2: Changes in headspace concentrations of NH_3 and H_2S during slurry disturbance, which takes place between the vertical lines.

3.2. Surface oxidation.

The presence of oxygen in the headspace strongly reduced emissions of H_2S and methanethiol compared to anoxic conditions as seen in Figure 3 and 4. This is a clear demonstration of surface oxidation of these compounds even if O_2 only penetrates little into the slurry surface [2]. NO other compounds were affected by the headspace oxygen content.

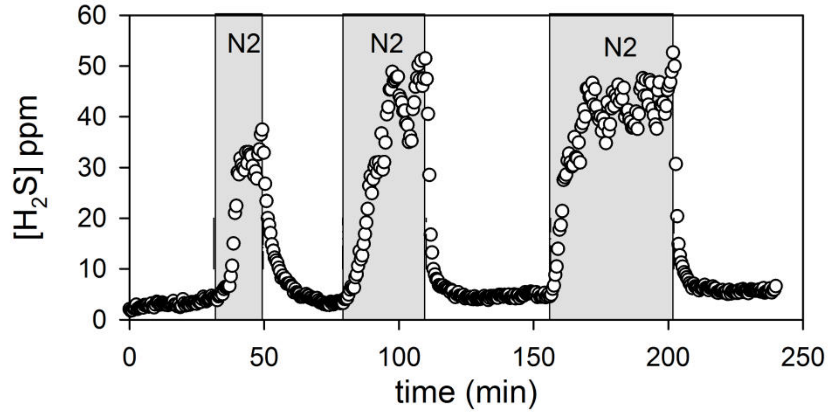

Figure 3: The headspace concentration of hydrogen sulfide (H_2S) (ppm) during the experiment. White background represents concentration in normal ambient air, the grey areas represent times where the headspace was flushed with N_2 and hence the system was completely anaerobic.

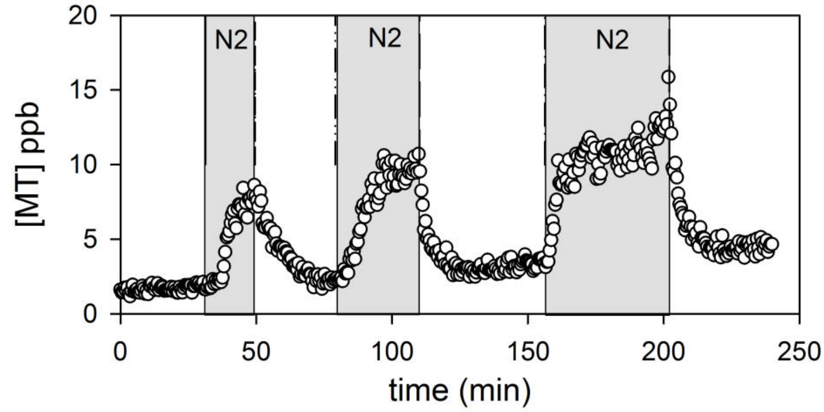

Figure 4: The headspace concentration of methanethiol (MT) (ppb) during the experiment. White background represents concentration in normal ambient air, the grey areas represent times where the headspace was flushed with N_2 and hence the system was completely anaerobic.

Surface pH was also not affected by exchanging the headspace composition between air and nitrogen (Figure 5). Hence, any effects of pH on changes in emissions observed during head-space exchange can be ruled out.

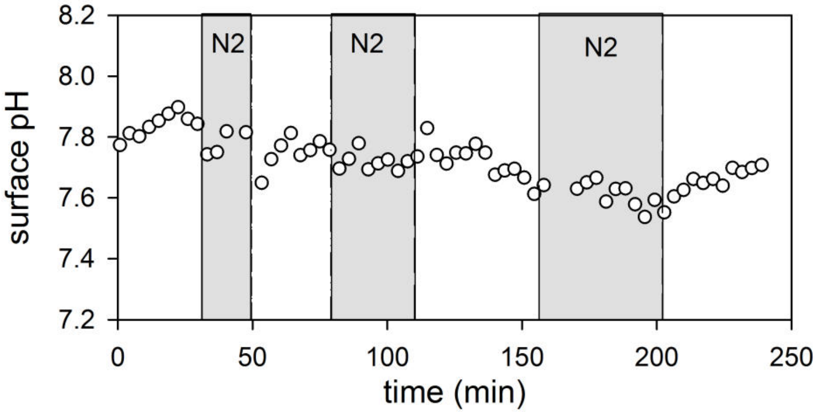

Figure 5: Changes in slurry surface pH measured at a depth of 250 mm during the experiment. White background represents concentration in normal ambient air, the grey areas represent times where the headspace was flushed with N_2 and hence the system was completely anaerobic.

4. Conclusion

The compound flux from the surface of manure slurry is affected by the pH gradient in the upper mm, including the effect of stirring on this gradient. Stirring leads to reduced surface pH and lower emissions of NH_3. Surface oxidation of H_2S and methanethiol is evidenced by strongly increased flux in the absence of oxygen in the headspace.

References

[1] Aneja, V. P.; Schlesinger, W. H.; Erisman, J. W. (2009): Effects of Agriculture upon the Air Quality and Climate: Research, Policy, and Regulations, Environmental Science and Technology 43, 4234-4240

[2] Petersen, V.; Markfoged, R.; Hafner, S. D.; Sommer S. G. (2014): A new slurry pH model accounting for effects of ammonia and carbon dioxide volatilization on solution speciation, Nutrient Cycling in Agroecosystems 100, 189-204

[3] Nielsen, D. A.; Nielsen, L. P.; Schramm, A.; Revsbech, N. P. (2010): Oxygen Distribution and Potential Ammonia Oxidation in Floating, Liquid Manure Crusts, Journal of Environmental Quality 39, 1813-1820

[4] Feilberg, A.; Liu, D.; Adamsen, A. P. S.; Hansen, M. J.; Jonassen, K. (2010) Odorant Emissions from Intensive Pig Production Measured by Proton-Transfer-Reaction Mass Spectrometry, Environmental Science and Technology 44, 5894-5900

TC-O_12 Bio-acidification of manure – By supplying manure with 2-3% sugar or cellulose

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1. Objectives

Ammonia emissions from manure can be decreased by 70% through lowering the pH to 5.5 with sulfuric acid in-house [1]. However, it is prohibited for organic farmers, may cause inhibition in biogas plants and an excessive soil-S, and handling is hazardous. An alternative is in-situ production of lactic acid [2] and acetic acid through lactic acid fermentation or acetogenesis. However, the quantities of bacterial and substrate supplement must be assessed, which requires clarification of the treatment metabolism. Hence, the focus of this study was to answer three questions:

- Does bio-acidification work?
- How does it work?
- How should it be performed?

2. Methodology

To bio-acidify cattle manure, the cattle manure must contain carbohydrate, acid producing bacteria and hydrolytic enzymes. In this experiment, cattle manure was supplied with 10 to 100 g of glucose, cellulose or starch; with 10^8-10^{11} colony forming units (CFU) per kg manure of lactic acid producing bacteria (LAB) *Pediococcus acidilactici, Pediococcus pentosaceu* and *Bacillus subtilitis*; and/or with the cellulytic enzymes β-gluconase and xylanase.

Once or twice a week 12.5% or 25%, respectively, of the sample was replaced by fresh manure, carbohydrate, microorganisms and/or enzymes, which is equal to a one month turnover. Treatments were run for 2-6 weeks in 0.2 L cattle manure batches with vertical shaking at 20-25 °C with continual or initial additions. Subsequently, treatments were stored without mixing for 4-8 weeks at 20-25 °C.

The stoichiometric requirement of lactic acid and substrate was estimated by performing titration of the input slurry with hydrochloric acid to a pH of 5.5. pH was monitored continuously during treatment. Quantities of glucose, lactic acid, acetic acid, propionic acid, and lactic acid producing bacteria were determined. Further, treatment bottles were sealed and flushed with nitrogen, and volume of produced gas and concentration of methane and CO_2 was determined.

3. Results and discussion

3.1 Does bio-acidification work?

The intent was to convert carbohydrates into lactic acid by lactic acid producing bacteria. The substrate concentrations tested were equal to three and five times the carbohydrate amount required to obtain pH 5.5 (estimated by a titration two days before), if all carbohydrates were converted into acid. This was equal to 30-50 g/kg for the applied cattle manure. Removal and addition of 12.5% of the manure and carbohydrate was performed twice a week.

The pH was reduced to 4.2 – 6.2 by supplying with glucose and cellulose (Figure 1). This was equal to a pH reduction of 0.5 – 2.7 pH units. When treatments were performed weekly, the pH was observed low from one week after experiment initiation, and for the remaining four weeks. Hence, indeed successful pH reductions of cattle manure were observed, when carbohydrates were added to the manure.

Treatment with amounts of lactic acid stociometrically equal to the glucose and cellulose treatments caused successful pH reductions, and indicated that success could be expected upon full conversion of the carbohydrate into acid. Treatment with the low doses of glucose and cellulose lead to a pH being -0.1 – 1.4 pH higher than the lactic acid after seven days. Treatment with the high carbohydrate dose led to a pH 0.6 – 1.3 higher than the lactic acid. Hence, not all added carbohydrate was observed present as lactic acid, in particular not at the largest cellulose supplements.

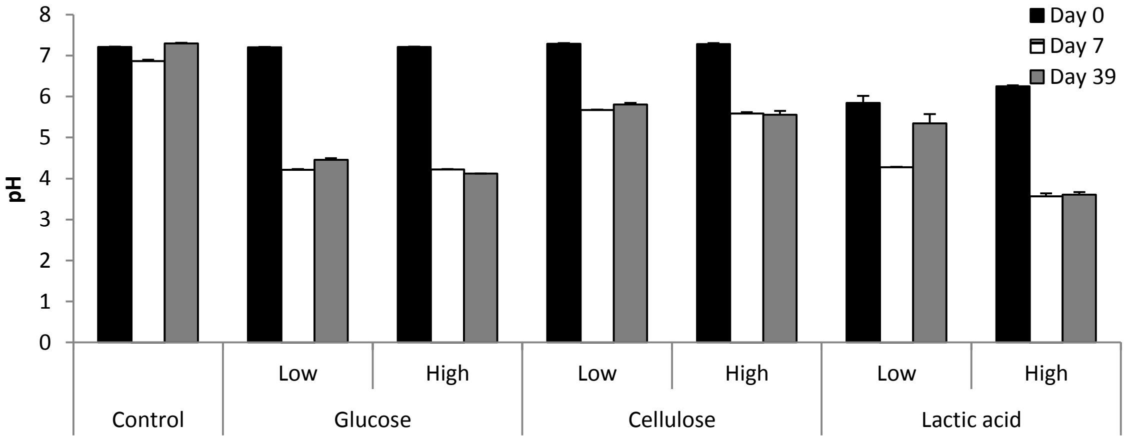

Figure 1: pH after 0-39 days of carbohydrate or acid addition to cattle manure. Low: three times the calculated dose to reach pH 5.5 = 20 g lactic acid per kg manure. High: five times the calculated dose to reach pH 5.5 = 50 g lactic acid per kg manure.

When the manure was added lactic acid producing bacteria and cellulolytic enzymes but no carbohydrates, the bio-acidification was not successful (data not shown). Comparing (i) addition of the bacteria and enzymes to the manure together with carbohydrate to the manure, with (ii) addition of the bacteria and enzymes to the manure together without carbohydrate to the manure, the pH and number of lactic acid bacteria was equal (Table 1). Thus, it appears that the inherent lactic acid producing were sufficient to cause the required conversion. And that the tested supplementary addition of bacteria and enzymes were unable to adjust the microbial consortium.

3.2 How does it work?

The intend of the treatment with glucose was to convert it into lactic acid. Supplements of 30-100 g pure glucose per kg manure all decreased the pH to 4.2-4.5 (Figure 1, Table 1). Lactic acid was observed present. The numbers of lactic acid producing bacteria increased three fold over the period. And residual glucose was observed in the end of the incubations in the treatments. Thus the lactic acid producing bacteria produced lactic acid from glucose; indeed the presence of lactic acid with an acid constant pKa of 3.8 matches the observed pH of 4.4. The observed terminal presence of glucose indicates that the glucose to lactic acid conversion was only occurring until the pH had dropped to pH 4.4; indeed typically the pH level providing optimal conditions for the lactic acid bacteria.

Cellulose and starch were intended hydrolysed into glucose and subsequently converted into lactic acid. Supplement with 30-100 g cellulose powder per kg manure all decreased the pH to 5.6-5.8 (Figure 1, Table 1). High acetic acid and propionic acid levels were present. The amount of lactic acid producing bacteria was not elevated. No effect of adding β-gluconase or xylase was observed. Thus the manure's indigenous content of hydrolytic exoenzymes appeared to cause a sufficient hydrolysis rate. But, the absences of glucose terminally indicates the hydrolysis to be the rate limiting step. It is indicated that acetic and propionic acid was formed; the pKas are higher than lactic acid explaining the obtained higher pH (pH 5.6 compared to 4.4). Hence, primarily acido-/acetogenic bacteria likely produced the acetic acid. Indeed these bacteria can produce hydrolytic exoenzymes in contrast to lactic acid bacteria, which could explain the occurrence of acido-/acetogenesis rather than lactic acid fermentation.

Subsequent degradation of the produced acid would cause an increase in pH. Upon addition of 100 g/kg glucose, 54% remained as glucose, 27% was observed converted into lactic acid, and 3% was observed converted into other acids after three weeks. Hence, 16% of the input glucose was indicated degraded into other components as CO_2, CH_4 and ethanol. Upon storage of the treated samples, the pH level has been observed to increase again to the pH of the input manure; some treatments within two weeks while some after more than two months. Production of methane was only observed above pH 5.5, while production of CO_2 was observed at all pH levels although with largest production at highest pH levels. Hence, the produced acid was likely degraded to products as CO_2 at the lower pH, and to CH_4 and CO_2, when higher pH levels were observed. This could be explained by a primary conversion at pH 4-5.5 of lactic acid into products as acetic acid and CO_2, and degradation above pH 5.5 of the produced acetic acid via methanogenesis to CH_4 and CO_2. This would terminally result in absence of both lactic and acetic acid, and thus a pH equal to the input manure.

Table 1: Characteristics of products from bio-acidification treatments. Results from two experiments are presented, therefore are both control treatments presented. na: data not available. +LAB: addition of Lactic Acid Bacteria (experiment a: 10^5 CFU/kg, experiment b: 10^8 CFU/kg). +enz: addition of enzyme.

Experi-ment	Treatment	Added carbohydrate (g/kg)	pH	Glucose (g/kg)	Acetic acid (g/kg)	Propionic acid (g/kg)	Lactic acid (g/kg)	Lactic acid bacteria (10^x CFU/kg)
a	Control	none	7.1	< 1	6	2	< 1	na
a	Glucose	10, +LAB	6.5	< 1	8	3	< 1	na
a	Glucose	100, + LAB	4.4	54	8	3	27	na
a	Cellulose	9, +LAB, +enz	6.2	< 1	8	5	< 1	na
a	Cellulose	90, +LAB, +enz	5.5	< 1	8	5	< 1	na
b	Control	none	7.5	na	1	1	na	5
b	Glucose	30	4.3	na	4	1	na	5
b	Glucose	50	4.3	na	4	1	na	5
b	Glucose	50, +LAB	4.3	na	4	1	na	5
b	Cellulose	30	5.6	na	6	5	na	9
b	Cellulose	50	5.6	na	5	6	na	9
b	Cellulose	50, +LAB, +enz	5.6	na	5	6	na	9

3.3 How should it be performed?

Cattle manure could be acidified by merely supplementing with a carbohydrate source. No addition of microorganisms or enzymes was observed useful under the conditions tested in this study.
To reach a pH below 5.5, glucose can be added. Above 30 g/kg was observed successful for reaching a pH below 5.5. At 10 g/kg, only pH 6.5 was obtained rather than the expected 5.5, and the amount of acid observed was only equal to 50% of what could be produced from the carbohydrate. Therefore, may addition of double the amount likely result in the intended pH 5.5. Thus 2-3% glucose should be added to manure. By increasing the amount, the duration of the period with low pH will be extended.
Using cellulose to reach pH 5.5, the data indicate the required amount to be equal to glucose, i.e. 2-3% cellulose. Addition of cellulose requires an initial hydrolysis to glucose. This hydrolysis, the exoenzymes in the manure appears sufficient.
Agricultural by-products such as molasses, maize silage, deep litter or straw could potentially replace the pure glucose or cellulose powders. These products would contain acid, free sugar as glucose, or xylose, it would contain cellulose and hemicellulose, or starch. The requirement for this includes an estimation of the carbohydrate concentration, the carbohydrate availability, identification of troublesome components, and considerations into the practical manure handling.

4. Conclusion and outlook

A successful treatment would thus be addition of a sugar, cellulose or hemicellulose rich agricultural by-product in an amount equal to 20-30 g carbohydrate per kg manure. No bacterial and likely no enzyme supplement should be added. Thereby would pH of 4.4-5.5 be obtainable. The carbohydrates could perhaps be added directly in the slurry channels. Compared to sulfuric acid-acidification with has been observed to reduce NH_3 emission by 70% in pig production, bio-acidification may reduce NH_3 emission equivalently. It would be a requirement to the treatment that the NH_3 emission is not merely swapped by methane emission; methane emission was from the treatment bottles in this study not indicated increased by the carbohydrate addition, as long as the pH was kept below the targeted pH of 5.5.

Acknowledgements

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References

[1] Fanguerio, D; Hjorth, M; Gioelli, F. (2015) Acidification of Animal Slurry – a Review, Journal of Environmental Management 149, 46-56.

[2] Clemens, J., Bergmann, S., Vandré, R., 2002. Reduced ammonia emissions from slurry after self-acidification with organic supplements. Environmental Technology 23: 429-435.

TC-O_13 Proposal of a composting model to predict and manage gaseous emissions and quality of compost

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1. Objectives

Composting is commonly used to treat solid organic residues and to recycle them as a soil improver or as an organic fertilizer for agricultural soils. An environmental limit of the composting process is linked to the gaseous emissions (GHG and NH_3) released by this biological treatment. The objective of this study was to propose a numerical model predicting gaseous emissions and the final compost quality in order to optimize composting management practices.

2. Methodology

Based on previous work by Tremier et al. [1], Denes et al. [2] and Oudart [3], a numerical model formed of three coupled modules (a biodegradation module describing organic matter fractions and microbial kinetics, a mass and heat transfers module and a nitrogen module describing the dynamics of nitrogenous forms) was designed as described in the results section.
In order to strengthen modelling assumptions and calibrate the parameters of the model, composting experiments were performed at a pilot scale:

- Studied substrates: solid manure from a finishing steer feedlot (MAN) and the solid phase of the digestate from the anaerobic treatment of this manure (ADMAN). Three composting mixtures were tested: MAN with woody bulking agent (R1); ADMAN with woody bulking agent (R2); MAN without bulking agent (R3).
- Composting trials were performed in a composting pilot reactor: airtight 300-litre stainless steel cylindrical insulated chamber (Reacting volume: diameter = 0.7 m; height L = 0.8 m). Aeration was supplied via an air blower, from the bottom through the material, and gases were collected at the top in order to analyze them.The following parameters were monitored: gas flow rate via a volumetric gas meter, temperature of the matter and gas in the reactor with Pt 100 temperature probes, entering gas temperature and humidity, inlet and outlet gas oxygen concentration thanks to a paramagnetic analyzer, carbon dioxide, methane and nitrous oxide concentrations thanks to an infra-red analyzer. Ammonia emissions were monitored thanks to bubbling of exhaust air in acid trap.
- Samples analysis: composting mixture was sampled before composting, after about 20 days of composting and at the end of the composting trial (about 45 days). Samples were analysed for the C and N content of the biochemical fractions that are described below.

3. Results and discussion

3.1 Composting modelling development

The developed model was based on three modules of coupled processes:

- A module describing the microbiological degradation of carbon: C module;
- A module describing the microbiological degradation of nitrogen: N module;
- A module describing mass and heat transfers along the composting process: transfers module.

The C module [2] describes the organic matter as 7 fractions (Figure 1.). 6 of them (SOLnd-R, SOLnd-B, HEM, CEL, LIC) have to be hydrolysed to give water soluble low molecular compounds ($SOLH2O_1$) that are used by two heterotroph biomasses (X_1 and X_2). Hydrolysis and microbial kinetics (growth and death) are limited by oxygen content, temperature and moisture conditions which are simulated in the transfer module.

The general principle of the N module (Figure 2) is the same as that of the C module: solid N organic fractions (NSOLnd-R, NSOLnd-B, NHEM, NCEL, NLIC) first undergo hydrolysis to form a slowly biodegradable water soluble fraction (NSOL $H2O_{org}$) that undergoes ammonification to form nitrogen directly assimilated by microorganisms. To simplify the model, the slowly biodegradable water soluble fraction is considered as soluble organic nitrogen, formed by the hydrolysis of particulate fractions of OM, while directly biodegradable water soluble fraction is considered as NH_4^+. In order to represent the evolution of mineral forms of nitrogen, an autotrophic biomass was added (NXa). Its growth determines the rate of nitrification. Based on the hypothesis that nitrates do not accumulate before the maturation phase, we calculate a potential of denitrification. Depending on the moisture, denitrification is complete and results in the reduced form (N_2), or is incomplete and results in the formation of N_2O. In addition to biological transformation of nitrogen, ammonia volatilization is also simulated (Henry's law). All kinetics are limited by oxygen content, temperature and moisture conditions which are simulated in the transfer module.

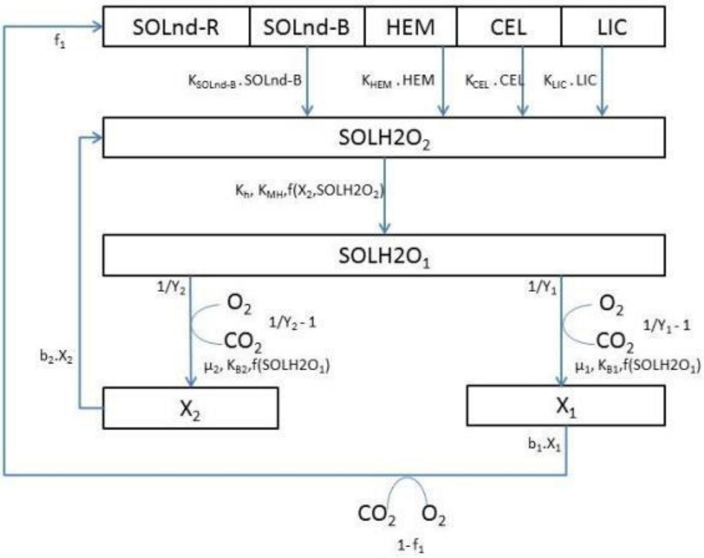

Figure 1: Scheme of the model for carbon biodegradation

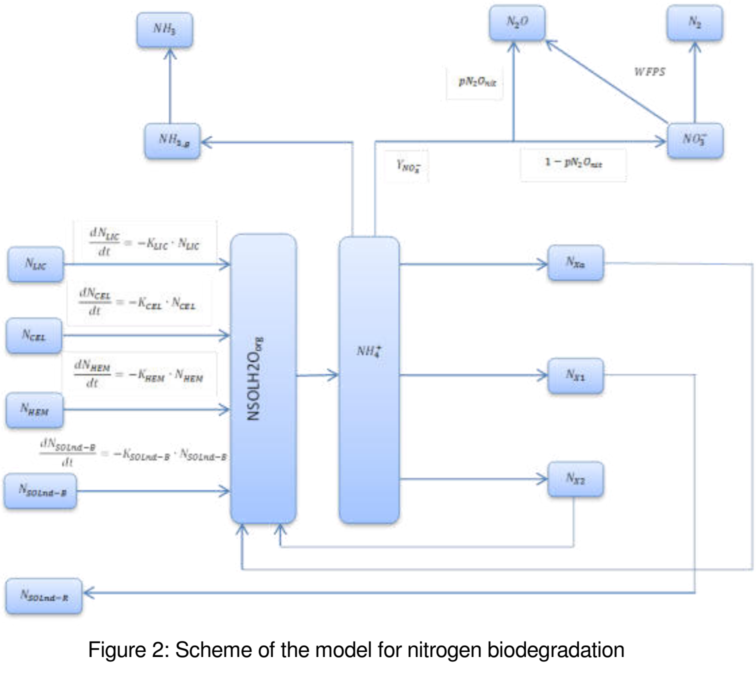

Figure 2: Scheme of the model for nitrogen biodegradation

The originality of this model relies on the description of the biodegradable organic matter that can be initialized thanks to analytical measurement on the substrate through a modified Van Soest method. Moreover, such a model will predict O_2 consumption, CO_2, N_2O and NH_3 emissions along the treatment. Thus using this model it will be possible to study how to manage substrate mixture, or aeration conditions to minimize nitrogenous emissions. Only CH_4 emissions are not yet integrated in the model equations. The model will also predict the quality of the compost organic matter at the end of the treatment both in terms of C and N. This last point is particularly innovative because it will permit to couple the composting model with a model of organic matter decomposition and mineralisation in soil (CANTIS). This will enable the simulation of the impact of composting treatment and of compost when applied to agricultural soils.

3.2 Mass balances and gaseous emissions along composting experiments

Substrates initial characteristics: Substrates (MAN and ADMAN – Table 1) mainly differed in terms of water soluble carbon content which was twice as high for the non-digested substrate (MAN) than for the pre-digested one (ADMAN). In terms of nitrogen content, even if the total content and its distribution within the fractions was comparable for the both substrates, the N composition within the water soluble fraction differed. Indeed, SOLH2O nitrogen of MAN was composed of 84% of organic N, whereas SOLH2O of ADMAN was composed of 45% of organic N and 55% of ammonia N, due to N mineralization along AD.

Table 1: Substrates characteristics before composting.

Sample	WC	VM	TC	TN	SOLH$_2$O		SOLnd		HEM		CEL		LIC	
	(%FM)	(%DM)	(%VM)		%TC	%TN	%TC	%TN	%TC	%TN	%TC	%TN	%TC	%TN
MAN	64.1	84.2	55.1	3.3	28.2	55.4	8.5	17.0	22.5	7.6	32.8	16.9	8.0	3.0
ADMAN	63.2	73.1	52.1	3.0	11.9	53.1	13.3	16.3	21.9	7.5	44.1	19.2	8.8	3.9

FM-fresh matter; VM- volatile matter; DM- dry matter; TC –total carbon; TN-total nitrogen; SOLH$_2$O-water soluble C or N; SOLnd-C or N soluble in neutral detergent; HEM-C or N of the hemicellulose like fraction; CEL-C or N of the cellulose like fraction; LIC-C or N of the lignin like fraction

Carbon losses and emissions: 55 to 45 % (R1 versus R2 and R3) of the initial carbon content of the waste were degraded along the composting process. As awaited, less degradation was observed for pre-digested substrate (R2). Concerning MAN, biodegradation was more efficient with bulking agent highlighting better conditions for aeration and substrate colonization by microorganisms. Losses of carbon were mainly explained by CO2 emissions. Nevertheless, 3 to 6 % of the initial carbon content was emitted as CH4. CH4 emissions occurred concomitantly to oxygen consumption peaks and seemed to be correlated to the extent of the biodegradation. It could be explained by local lack of oxygen when the respiration rate is maximum. Such an observation could lead in the future to integer in the model a stoichiometric equation linking methane emissions to the maximum of the oxygen consumption rate in order to predict methane emissions.

Nitrogen losses and emissions: Nitrogen losses along the composting treatment largely differed from one trial to the others. More than 46% of the initial nitrogen of MAN was lost when composted with bulking agent (R1), whereas only 19% was lost when composted without bulking agent (R3). During composting of ADMAN (R2), 24% of the initial nitrogen was lost.
When considering the nitrogenous emissions in gaseous flows and in leachates it appeared that:

- Rather no leachates were obtained,
- Measured emissions of nitrogen (NH_3 + N_2O) plus nitrogen adsorbed on the bulking agent covered 20 to 85% of the nitrogen losses. The rest of the nitrogen losses in the substrate could be due to N_2 emissions
- Ammonia emissions occurred mainly during the first week of composting concomitantly to the temperature peak, and represented a larger part of nitrogenous emissions when composting was performed without bulking agent
- N_2O emissions occurred mainly after the oxygen consumption rate decrease. N_2O emissions were higher for the pre-digested substrate.

These experimental results showed that nitrogenous emissions during composting were both influenced by the biodegradation behaviour and by the physical structure of the composting mixture. Indeed, the use of a bulking agent influenced the yield of the biodegradation (higher in R1 than in R2) but also the availability of ammonia-nitrogen for volatilization or nitrification. This physical structure is a key issue for modelling of nitrogenous emissions. In the developed model, the physical structure is considered through free air space and air permeability constants in the transfers' module. Nevertheless, these constants are not considered in the limitations of kinetics for N_2O production. Moreover adsorption of ammonia nitrogen on bulking agent is not yet considered in the developed model.

Variation of organic matter quality: Variations of C and N distribution in the substrate along composting is presented on Figure 3 (exemple of trial R1).

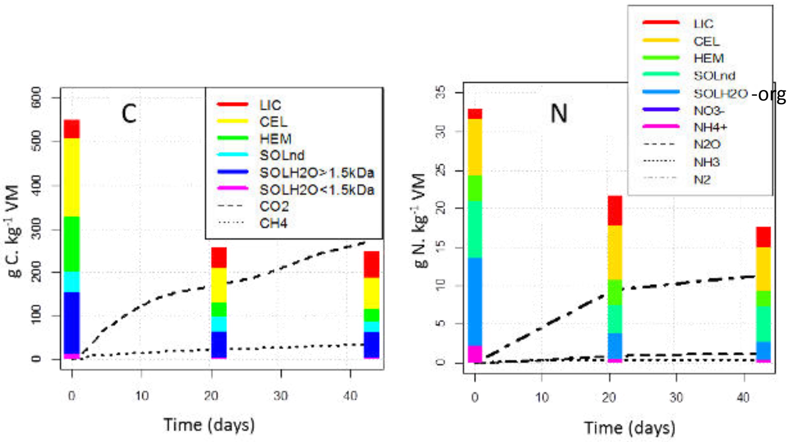

Figure 3: Carbon and Nitrogen distribution along composting treatment (Example of trial R1: MAN + bulking agent).

Along composting, carbon disappeared mainly from hemicellulose and cellulose like fractions. Carbon of cellulose-like fraction is the most degraded when the initial amount of water soluble carbon is law. In terms of nitrogen, losses along composting are mainly explained by losses of N-ammonia nitrogen and of water-.soluble organic nitrogen. Nitrogen from the SOLnd fraction participated in nitrogen losses at the beginning of the treatment, but re-increased in the second stage of the process. The nitrogen content of other fractions was rather constant. As a consequence, C/N ratio of each fraction may vary along composting.

These results confirmed our modelling hypothesis concerning carbon transformation. They also confirmed the hydrolysis pathway from water soluble organic nitrogen to ammonia nitrogen, before ammonia volatilization or nitrification but contradicted modelling assumptions for nitrogen contents of lignin-, hemicellulose- and cellulose-like fractions since their hydrolysis pathways were not identical for carbon and nitrogen.

3.3 Advances in composting modelling

The composting model was tested with constant parameters values issued from literature (for nitrogen and transfers modules) and from respirometric tests (for carbon module). Variable parameters were initiated from the composting trial of MAN without bulking agent.

CO_2 production kinetics was qualitatively well simulated (Figure 4). Errors between experiments and numerical simulation can be due to simplifications on particulate carbon fractions hydrolysis that do not take into account delay for cellulose hydrolysis. The total loss of carbon in the substrate was well predicted (45%). Errors on carbon distribution between the simulated fractions were observed: storage in SOLnd fractions and biomasses fraction was overestimated. On the contrary, nitrogen variations were badly simulated: simulated nitrogen losses were too low (9%) whereas nitrogen was stored in ammonia-nitrogen and water-soluble organic nitrogen fractions. Ammonia emissions and N_2O emissions were thus underestimated. It probably indicates that fixing the same hydrolysis kinetics for carbon and nitrogen is not a good way to simulate nitrogen transformation processes.

Concerning heat transfers, temperature was well simulated at the beginning of the treatment but then didn't decrease enough rapidly. This error in heat transfer kinetics was probably linked to errors in carbon biodegradation kinetics that was partly overestimated.

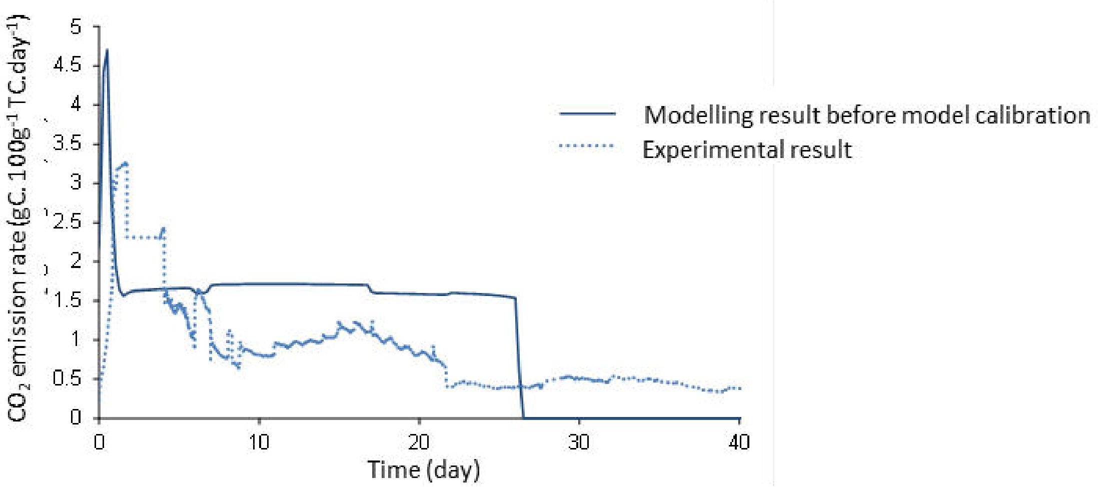

Figure 4: Test of the composting model before strict calibration: comparison of simulation and measure for CO_2 emission rate.

4. Conclusion and outlook

The developed composting model proved to have a big potential in simulating gaseous emissions and compost quality. Nevertheless, experimental results compared to model simulation showed that some assumptions of the model have to be corrected. Moreover the model has to be precisely calibrated. When calibration will be improved, such a composting model will be very useful to test composting management scenarios in order to limit nitrogenous emissions and to obtain a valuable compost, which properties will then be tested through modelling of soil mineralisation. Such a test will be possible because of the good compatibility between the composting and the CANTIS model.

Acknowledgements

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References

[1] Trémier, A., et al.: Management strategies for a forced-aerated composting process: use of a numerical model to predict and improve the treatment. In: 5th International conference biological waste management from local global, Weimar, DEU, 13-15 september 2006 pp. 203-12 (2006)

[2] Denes, J., et al.: Numerical simulation of organic waste aerobic biodegradation: A new way to correlate respiration kinetics and organic matter fractionation. Waste Management. (In Press)

[3] Oudart, D.: Modélisation de stabilisation de la matière organique et des émissions gazeuses au cours du compostage d'effluents d'élevage [Thèse de doctorat], Institut des sciences appliquées de Toulouse (2013)

TC-O_14 Gaseous emissions from slurry storage – Influence of temperature and potential mitigation methods

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1. Objectives

The objectives of this study were to assess the influence of storage temperature on methane (CH_4) and ammonia (NH_3) emissions from pig and cattle slurry storage and to assess the potential of two mitigation techniques – namely a clay granule floating cover and acidification, to effectively reduce both NH_3 and CH_4 emissions.

2. Methodology

The experiment was conducted using six pilot-scale slurry tanks (c. 1 m^3, 1m depth, 1.2m diameter) with specially adapted lids for gaseous emission measurement situated in a polytunnel. A total of six experiments were conducted (Table 1), each of two months duration, covering two slurry types (pig, cattle), three temperature regimes (winter, summer, spring/autumn) and two potential mitigation practices (floating clay granule cover for the pig slurry, acidification for the cattle slurry), with 3 replicates of each treatment.

Table 1: Details of the slurry storage experiments

Experiment	Slurry type	Storage regime	Mean air temperature (°C)	Slurry DM (g kg^{-1})	Slurry pH	Slurry TN (g kg^{-1})	Slurry TAN (g kg^{-1})
1	Pig	Spring	11.1	81.1	8.1	6.32	2.83
2	Pig	Summer	17.1	61.7	7.9	5.74	2.88
3	Cattle	Autumn	11.0	66.2	7.1	2.76	0.84
4	Cattle	Winter	7.3	54.2	7.3	2.49	0.78
5	Pig	Winter	9.2	61.5	7.1	5.62	3.69
6	Cattle	Summer	17.2	51.1	7.3	2.76	0.95

DM-dry matter; TN-total nitrogen; TAN-total ammoniacal nitrogen

The slurry storage tanks were fitted with specially adapted lids, which had a central circular hole of c. 10 cm diameter to which a fan was fitted to draw air from the tank headspace. Air was drawn into the tank headspace via ten holes (each of 3 cm diameter) around the outer edge of the lid. The air was vented via the fan through a duct to an area outside the polytunnel. The lids were left in-situ throughout the storage period with fans running continuously at c. 0.04 $m^3 s^{-1}$. The pig and cattle slurries were obtained from local commercial farms. Slurry samples taken at the start of storage were analysed for total and volatile solids content, total nitrogen and total ammoniacal nitrogen (TAN) content and pH. In addition, the CH_4 producing potential (B_o) of the slurry was determined using a purpose-designed laboratory incubation system (Bioprocess Control, Lund, Sweden).

For the floating cover treatment on the pig slurry, a layer of 2 cm diameter expanded clay granules was applied to the slurry surface on each of the three treatment tanks to a depth of 7 cm. For the acidification treatment with the cattle slurry, 5 L of concentrated sulphuric acid was added to each of the three treatment tanks during the filling process for the first cattle slurry experiment (Experiment 3). This proved to be too much, lowering the slurry pH dramatically to approximately 5 and causing excessive foaming during addition. For subsequent experiments (Experiments 4 and 6), 2.5 and 3.5 L, respectively, were added to each of the three treatment tanks.

Ammonia concentrations in the outlet air from each tank and the ambient inlet air were measured twice per week by passing a subsample of the air flow through acid absorption flasks. Methane concentrations were monitored semi-continuously using a Los Gatos Ultra-Portable Greenhouse Gas Analyser (Los Gatos Research, California). Estimates of flux for each gas (F, µg s^{-1}) could therefore be made according to:

$$F = V(C_o - C_i) \qquad \text{Eq. 1}$$

where V ($m^3\ s^{-1}$) is the air volume flow rate and C_o and C_i the outlet and inlet gas concentrations ($\mu g\ m^{-3}$), respectively.

3. Results and discussion

3.1 Ammonia emissions

Ammonia emission rates were much greater from the pig than the cattle slurry, with average respective emission rates from the control tanks across the experiments of 13.8 and 2.8 g NH_3-N m^{-3} slurry d^{-1}. This 5-fold difference in average emission rate reflects the greater TAN content and higher average pH of the pig slurry (Table 1) and also the greater propensity for cattle slurry to form a crust during storage, which is associated with emission reduction [1].

Slurry temperature during storage of the covered pig slurry was slightly greater than for the control pig slurry which was not significantly different from ambient air temperature as was the case with the control and acidified cattle slurry (data not shown). Storage temperature clearly had an influence on the NH_3 emission rate (Fig. 1), with an average emission rate for the control treatments increasing in the order winter < spring/autumn < summer for both the pig and cattle slurries. Variation in emission rate during each storage period also correlated well with temperature (data not shown).

The floating clay granule crust on the pig slurry was effective in reducing the NH_3 emission rate throughout each of the three 2-month storage periods (Fig. 1), with the pattern in emission rate following that of the respective control treatment (i.e. correlated with temperature). Acidification of cattle slurry was also effective in reducing emission rate, but the effectiveness varied between experiments, being most effective for the autumn experiment where the greatest quantity of acid was added to the slurry (Fig. 1). For the winter experiment, the effectiveness of the initial acid addition decreased with time. This decrease in effect after 15-30 days was also noted for the summer experiment, but emissions then declined again from the acidified treatment which was associated with the development of a dry, intact crust for that treatment compared with a wetter, less complete crust on the control.

a) b)

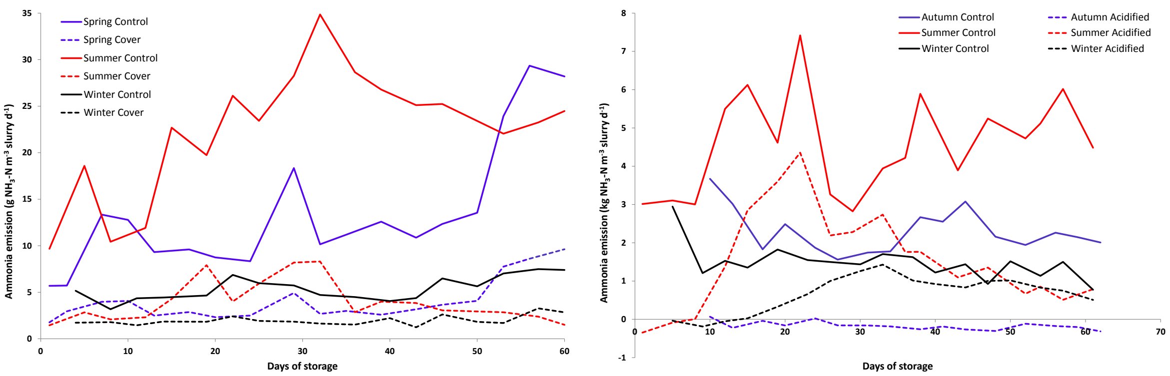

Figure 1: Ammonia emission rates (g NH_3-N m^{-3} slurry d^{-1}) from a) pig slurry without (solid lines) or with (dashed lines) a clay granule cover and b) cattle slurry without (solid lines) or with (dashed lines) acidification during storage at three different times of year.

The average cumulative NH_3 emission over the 2-month storage period across both slurry types and all experimental timings was 29% of the initial slurry TAN for the control treatments (35 and 23% for the pig and cattle slurries, respectively). This is greater than the current UK emission factor for slurry stored in above-ground tanks of c. 10% of slurry TAN [2], which is most likely because of the greater surface area to volume ratio of the slurry in the pilot-scale tanks compared with on-farm slurry tanks, making them more similar to the conditions in a slurry lagoon which are associated with a higher emission factor.

There was a significant effect of storage temperature on cumulative emissions from the control treatments and this was more pronounced for the pig slurry than the cattle slurry (Table 2), presumably because of the greater crust development on the cattle slurry under warm conditions compared with pig slurry. For the pig slurry covered with clay granules, there was some evidence of a temperature effect on cumulative emission, with that in winter being significantly less than that in spring or summer. For the acidified cattle slurry, there were significant differences in cumulative emission for the different times of year, but these were not related solely to temperature differences but also to the quantity of acid added and incidence of crust development.

Table 2: Cumulative ammonia emissions (as % of the initial slurry TAN content).

Slurry type	Treatment	Cumulative ammonia emission (as % of initial TAN content)		
		Winter	Spring/Autumn	Summer
Pig	Control	10.5[a]	42.1[b]	52.5[c]
Pig	Clay granule cover	4.2[a]	10.7[b]	9.1[b]
Cattle	Control	12.6[a]	22.6[b]	33.2[c]
Cattle	Acidified	5.4[b]	0.3[a]	10.9[c]

Within rows, values with different superscripts differ significantly (P<0.05)

Comparing the average cumulative NH_3 emission of the control with the treated slurries across the three storage timings showed an effective reduction efficiency of 77% for covering pig slurry with a layer of floating clay granules and 76% for acidification of cattle slurry. The effectiveness of the covering treatment tended to increase as the emissions from the control treatment increased (i.e. with temperature), with emission reductions of 60, 75 and 83% for winter, spring and summer, respectively. These observations fit within the upper end of the reported range in reduction effectiveness achieved with floating materials on slurry stores [3]. The effectiveness of the acidification treatment varied across the timings with emission reductions of 58, 99 and 67% for winter, autumn and summer, respectively, with differences in the duration of effect as shown in Figure 1 and discussed above. Petersen et al. [4] reported very effective reduction (>95%) in NH_3 emissions from cattle slurry following acidification to pH<6, but discussed the need to maintain a sufficiently low pH throughout the entire storage period.

3.2 Methane emissions

There was a distinct diurnal pattern of CH_4 flux, correlating well with diurnal temperature variation. Mean daily flux from the control pig slurry tanks was 2.9, 19.2 and 20.9 g CH_4 m^{-3} slurry d^{-1} for winter, spring and summer, respectively, and for the control cattle slurry was 1.2, 0.6 and 64.5 g CH_4 m^{-3} slurry d^{-1} for winter, spring and summer, respectively.
The mean cumulative CH_4 emission over all storage timings from the control treatments, expressed in terms of the initial VS content of the slurries, was significantly greater from the cattle than the pig slurry with respective values of 29.7 and 17.6 g CH_4 kg^{-1} VS. However, this was strongly influenced by the much greater emission from cattle slurry during summer storage. Storage temperature significantly influenced the cumulative emission (Table 3) with emissions from pig slurry being significantly different for each storage timing, increasing with temperature in the order winter < spring < summer. For the cattle slurry, however, there was no significant difference in emissions between the winter and autumn storage times for the control treatment, with very low emissions for both and much greater emission from summer storage.

Table 3: Cumulative methane emissions (g CH_4 kg^{-1} slurry VS).

Slurry type	Treatment	Cumulative methane emission (g CH_4 kg^{-1} slurry VS)		
		Winter	Spring/Autumn	Summer
Pig	Control	4.1[a]	21.5[b]	27.1[c]
Pig	Clay granule cover	4.4[a]	20.2[b]	27.7[c]
Cattle	Control	1.7[a]	0.8[a]	86.7[c]
Cattle	Acidified	5.4[b]	0.3[a]	10.9[c]

Within rows, values with different superscripts differ significantly (P<0.05)

The average slurry VS content was 55 and 48 g kg^{-1} for the pig and cattle slurries, respectively. Average B_o values were determined as 0.37 and 0.20 m^3 CH_4 kg^{-1} VS for pig and cattle slurry, respectively; that for pig slurry is lower than the IPCC default value of 0.45 m^3 CH_4 kg^{-1} VS, while that for cattle compares well with the IPCC default values of 0.24 and 0.18 m^3 CH_4 kg^{-1} VS for dairy and other cattle, respectively [5]. Maximum potential CH_4 emission was determined for each of the stored slurries based on these VS and B_o measurements and the methane conversion factor (MCF) for slurry storage was then derived as the measured emission expressed as a percentage of the maximum potential emission. The MCF values for the pig slurries were 1.6, 8.7 and 11.5% for winter, spring and summer, respectively, and for the cattle slurries were 1.4, 0.6 and 63.5%, respectively, for the 2-month storage period. Slurries are typically stored for longer than two months in the UK, but based on these results we can estimate an average 6-month storage MCF for pig slurry of 22%, assuming storage may be at any time of year, which compares favourably with the IPCC default value of 17% appropriate for UK temperatures. For cattle slurries, storage is generally through the autumn, winter and spring months, giving an MCF based on this study of c. 2%, much lower than the IPCC default value and in agreement with the observations of Rodhe et al. [6] for pig slurry storage in Sweden. However, any storage over summer months would greatly increase this value. Further measurements are required for a range of slurries across the range of typical storage temperature to develop robust MCF values for the UK, but results from this study would suggest that the current value of 17% for cattle slurry used in the UK greenhouse gas inventory is too high.

The clay granule covering on the pig slurry had no significant effect on cumulative CH_4 emission (Table 3). This agrees with Rodhe et al. [6] who observed no effect of a floating straw cover, and suggests that either methanotrophic activity did not develop in the cover or that CH_4 passed through the cover at too high a rate for oxidation to occur [7]. These results therefore do not support the lower MCF value given by IPCC for crusted slurry stores [5]. Mean emission from the acidified cattle slurries across the three storage timings was 61% lower than for the control slurries. Reduction efficiencies at each storage timing were 82, 88 and 60% for winter, autumn and summer, respectively, but control emissions for the winter and autumn timings were already very low as discussed above. These results agree well with those of Petersen et al. [4] who reported emission reductions of between 67 and 87% when acidifying cattle slurry to pH 5.5.

4. Conclusion and outlook

Temperature has a significant influence on NH_3 and CH_4 emissions from stored cattle and pig slurries, and the development of national emission factors for use in emission inventories should take account of the temperature, timing and duration of storage. Slurry acidification is an effective method to reduce both NH_3 and CH_4 emissions, with respective emission reductions of 76 and 61% observed for cattle slurry. However, a low pH needs to be maintained throughout the storage period to ensure effective emission reduction. A floating clay granule cover reduced NH_3 emission from pig slurry storage by an average of 77%, but gave no effective reduction in CH_4 emissions. Further work is required to assess the effectiveness of such covers over longer storage duration and to investigate the possibility of enhancing methanotrophic activity in the floating layer.

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References

[1] Misselbrook, T.H.; Brookman, S.K.E.; Smith, K.A.; Cumby, T.R.; Williams, A.G.; McCrory, D.F. (2005): Crusting of stored dairy slurry to abate ammonia emissions: pilot-scale studies, Journal of Environmental Quality 34, 411-419

[2] Misselbrook, T.H.; Gilhespy, S.L.; Cardenas, L.M.; Williams, J.; Dragosits, U. (2014): Inventory of ammonia emissions from UK agriculture - 2013, available at http://naei.defra.gov.uk/reports/

[3] Van der Zaag, A.C.; Gordon, R.J.; Glass, V.M.; Jamieson, R.C. (2008): Floating covers to reduce gas emissions from liquid manure storages: a review. Applied Engineering in Agriculture 24, 657-671

[4] Petersen, S.O.; Andersen, A.J.; Eriksen, J. (2012): Effects of cattle slurry acidification on ammonia and methane evolution during storage, Journal of Environmental Quality 41, 88-94

[5] Dong, H.; Mangino, J.; McAllister, T.A.; Hatfield, J.L.; Johnson, D.E.; Lassey, K.R.; de Lima, M.A.; Romanovskaya, A. (2006): Emissions from Livestock and manure management; Chapter 10 of 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 4, Agriculture, Forestry and Other Land Use, available at http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

[6] Rodhe, L.; Abubaker, J.; Ascue, J.; Pell, M.; Nordberg, A. (2012): Greenhouse gas emissions from pig slurry during storage and after field application in northern European conditions, Biosystems Engineering 113, 379-394

[7] Petersen, S.O.; Dorno, N.; Lindholst, S.; Feilberg, A.; Eriksen, J. (2013): Emissions of CH_4, N_2O, NH_3 and odorants from pig slurry during winter and summer storage. Nutrient Cycling in Agroecosystems 95, 103-113

TC-O_15 Ammonia emissions factor modelling of naturally ventilated dairy housings using on-farm measurements, climate data and nitrogen levels

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1. Objectives

There is a pressing need in the agricultural and environmental policy sectors for up-to-date ammonia (NH3) emissions data from dairy farming. The aim of this study was to quantify NH_3 emissions for the projected most common future dairying situation in Switzerland, and to derive a year-round emission factor (housing) as a contribution to emission inventories. This paper focuses on nitrogen input in feed and nitrogen output in milk, urine and slurry.

2. Materials and methods

Emission measurements in six dairy loose housing systems with natural ventilation, solid floors and an outdoor exercise area were performed using a dual tracer ratio method [1]. Measurements in two out of three seasons (summer, transitional period, winter) on each farm covered the variation in climate over the course of the year. Each measurement period lasted for three days. A large set of accompanying parameters such as descriptive farm data, climate data, aisle/exercise-area soiling, and nitrogen input, output and utilisation were recorded to characterise each measuring situation and to deduce relevant influencing variables. Whilst, for example, it was only possible to gather feeding data and tank-milk urea content for the entire herd, urine was sampled at the individual animal level differentiated in to three lactation stages. For each measuring exercise, urine samples were taken from three cows per lactation stage during the act of urination. For this, cows were locked into the feed fence during the morning milking. Slurry samples from the floor surfaces in the housing and from the outdoor exercise area were taken at three times per day and combined into bulk samples for each measurement. Sampling took place in the aisles during dung removal via an automatic scraper system immediately before disposal, as well as at four to six locations in the outdoor exercise area.

A linear mixed-effects model was used to describe NH_3 emissions by means of fixed effects and taking into consideration of the hierarchical data structure of measuring day b_{ijk}, measuring period b_{ij} and farm b_i in the form of nested random effects. Mean values were calculated for the respective measurement cycles for parameters with a higher temporal resolution, e.g. air temperature, wind speed, relative atmospheric humidity, air pressure, global radiation and ground temperature. Time-aggregated values over the three-day measuring periods were available for the other fixed effects, i.e. tank-milk urea content, NH_4-N content in urine, total N content in urine, total N content in feed, traffic-area soiling, total N content, and NH_4-N content in soiling. The co-linearity of the fixed effects was first investigated using scatter-plot matrices to select sets of uncorrelated explanatory variables. The model was selected from several models constructed on the basis of sets of weakly correlated variables:

$$E_{ijkl} = \mu + b_i + b_{ij} + b_{ijk} + \beta_1 OT + \beta_2 UMC + \beta_3 WS + \beta_4 OT \cdot UMC + \varepsilon_{ijkl} \qquad \text{Eq. 1}$$

where E_{ijkl} is the response variable (NH_3 emission), μ the intercept, and the fixed effects are outside temperature OT (°C), wind speed in the housing WS (m s^{-1}) and tank-milk urea content UMC (mg dl^{-1}).

Using the presented statistical model (Equation 1) as a starting point, NH_3 emission factors were calculated by bootstrap point estimates. The underlying data for this model-based calculation are milk-urea data from the Brown Swiss, Fleckvieh and Holstein breeders‘ association milk-inspection data, differentiated over a five-year period according to plain and mountain region. The model-based calculation is also based on air temperatures gathered at 43 weather stations at two altitudes (plain region, mountain region) by the Swiss Federal Office for Meteorology and Climatology, MeteoSchweiz. In order to depict the high temporal resolution of the emission data of the statistical model, the modelling was based on hourly averages of diurnal cycles per calendar week over the years 2004 to 2008. These detailed supporting data allowed the depiction of typical progressions over a five-year period. In order to show the effect of wind speed in the housing, two wind speeds of 0.3 m s^{-1} and 0.5 m s^{-1} were derived from our own measurements and from the literature. The modelling is described in detail in [1].

3. Results and discussion

The six farms differed in terms of herd size (20–74 cows; 28–90 livestock units (LUs)), live weight of the animals (average herd: 669–871 kg), milk yield (average herd: 18.6–30.6 kg cow^{-1} d^{-1}), and feed ration. On three farms (Farms 1, 2 and 3), the feed ration basically consisted of silages, hay and concentrate (Tab. 1).

Table 1: Description of farms and measuring periods: herd, milk, urine, traffic-aisle soiling, and feed (Su =summer; Tr = transition period; Wi = winter; No. = Number; Av. = Average; LU = livestock unit, DM = dry matter; TMR = total mixed ration).

Parameter	**Farm 1**		**Farm 2**		**Farm 3**		**Farm 4**		**Farm 5**		**Farm 6**	
	Su	**Tr**	**Su**	**Wi**	**Tr**	**Wi**	**Tr**	**Wi**	**Su**	**Wi**	**Su**	**Tr**
Herd	Dairy cows		Dairy cows, breeding bull		Dairy cows, female off-spring		Dairy cows, breeding bull		Dairy cows		Dairy cows	
Breed	Holstein Frisian		Brown Swiss		Brown Swiss		Brown Swiss, Fleckvieh		Holstein Frisian, Fleckvieh		Holstein Frisian, Fleckvieh	
No. of LUs [n]	28	28	58	70	94	97	39/40	41	77	78	85/90	83
Av. live weight of cows [kg]	693	690	724	871	669	709	713	730	824	849	825	849
Av. milk yield [kg cow^{-1} d^{-1}]	25.9	25.6	22.3	18.6	19.8	19.3	28.8	26.1	30.6	28.1	30.4	28.2
Av. milk urea content [mg dl^{-1}]	17.0	18.0	26.5	17.5	17.5	13.0	17.0	26.0	25.0	27.5	25.5	21.0
Urine												
Total N [g LU^{-1} d^{-1}]	93	64	131	77	88	55	102	120	130	146	112	102
NH_4-N [g LU^{-1} d^{-1}]	9[a]	37	48	32	36	24	27	35	56	57	51	41
Traffic-aisle soiling (housing) Total N [g kg^{-1} DM]	35	38	37	34	39	31	39	35	45	42	41	37
NH_4-N [g kg^{-1} DM]	12	16	13	9	12	13	12	22	20	13	14	14
Feed												
Feed intake [kg DM LU^{-1} d^{-1}]	11.0	12.2	9.6	12.9	12.2	12.3	11.4	12.0	9.6	7.4	14.4	12.7
N input feeding [g LU^{-1} d^{-1}]	240	280	170	300	210	280	250	280	220	210	350	300
Feed components									TMR		TMR	
Grass silage	x		x		x				x (Wi: alfalfa)		x	
Hay	x		x		x		x		x (Wi: alfalfa)		x (alfalfa)	
Maize silage			x		x				x (Wi)		x	
Maize-grain silage									x		x	
Concentrate	x		x		x		x					
Extracted soybean meal									x		x	
Further components			Su: green forage						rapeseed cake; Wi: potato, sugar beet-pulp silage Su: urea		corn-cob mix, maize gluten, extracted rape-seed meal	

[a] Very low reading caused by methodological errors in sample preparation

Farms 5 and 6 provided a total mixed ration (TMR). Farm 4 did not use silage. In summer, the Farm 2 ration included green forage. At 7.4 to 14.4 kg DM LU^{-1} d^{-1}, the feed-intake values are lower than those of the literature (13.8 to 19.6 DM LU^{-1} d^{-1} [2]). This underestimation is possibly attributable to inaccuracies and shifts in the recording of the quantities of dispensed feed and trough leftovers on a sliding basis over a period of several days. The nitrogen input in Farm 2's winter measurement was the lowest at 170 g LU^{-1} d^{-1}, and the highest in Farm 6's summer measurement at 350 g LU^{-1} d^{-1} (Tab. 1). At 146 and 130 g LU^{-1} d^{-1} respectively, total nitrogen content in urine was the highest in both measurements on Farm 5, accounting for approximately two-thirds of the nitrogen input in both cases. The winter measurement on Farm 3 had the lowest total nitrogen content in urine, 55 g LU^{-1} d^{-1}. The percentage of ammonium nitrogen content in urine ranged between 10 and 27% of the nitrogen input.

The tank-milk urea level for the farms and measuring periods varied from 13.0 to 27.5 mg dl^{-1} (Tab. 1). In both seasons, Farms 1 and 2 exhibited fairly low urea content. Urea content tended to be higher for Farms 5 and 6 (Fig. 1). For Farms 2 and 4, differences in tank-milk urea content between the two seasons were detectable. The assignment of protein and urea content to the new fields shows that there was no crude-protein surplus.

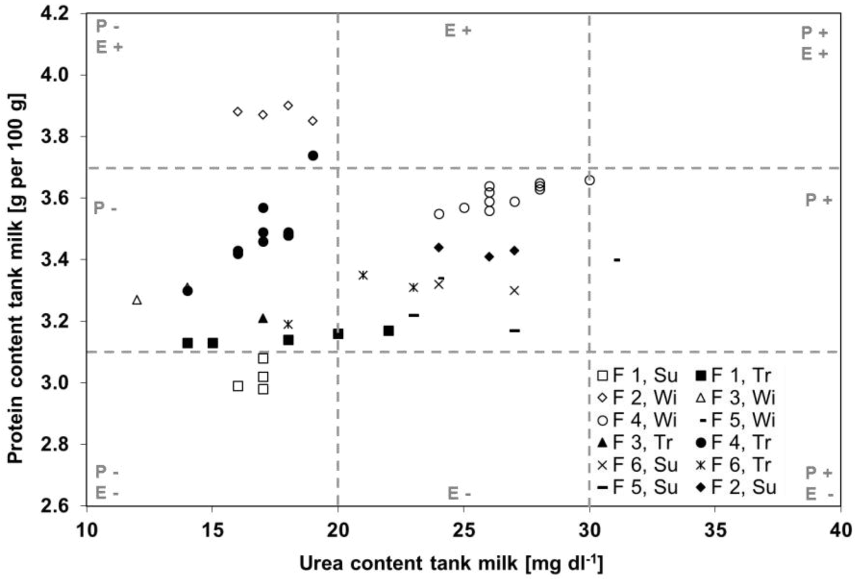

Figure 1: Urea content in tank milk [mg dl^{-1}] over the entire twelve measuring periods according to farm and season (F = farm, Su = summer, Tr = transitional period, Wi = winter, E energy, P protein, + surplus, - deficit)

Tank-milk urea content was strongly correlated with total urine nitrogen content (Fig. 2). Tank-milk urea content was taken into account during the modelling, and provides information on the entire herd, whilst the total nitrogen excreted in the urine is derived from a sample of only nine cows per farm in each case.

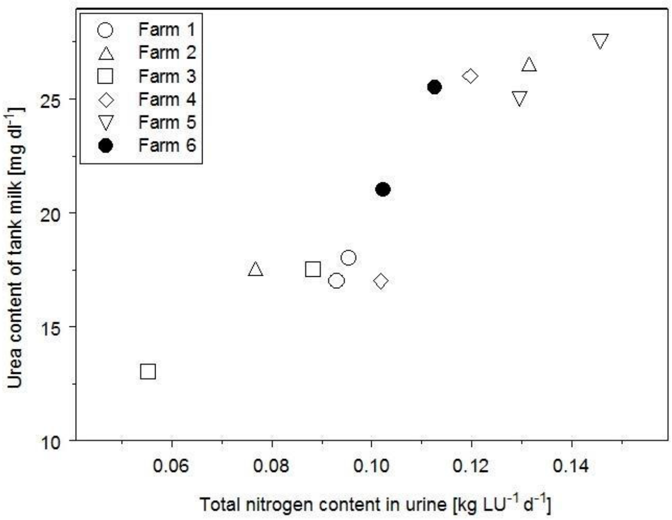

Figure 2: Urea content of tank milk [mg dl^{-1}] as a function of the total nitrogen in urine [kg LU^{-1} d^{-1}], shown by farm

A linear mixed-effects model revealed outside temperature ($p < 0.001$), wind speed in the housing ($p < 0.001$) and tank-milk urea content ($p = 0.0446$) to be significant variables influencing NH_3 emissions. The linear mixed-effects model reflects the significant relationship between tank-milk urea content and NH_3 emissions (coefficient: 0.79). As a reliable indicator of nitrogen utilisation and the nitrogen level for the whole herd [3], tank-milk urea levels appeared to be useful for understanding differences within and between farms. As with our results, tank-milk urea content was shown in the literature to be closely linked to NH_3 emissions [3; 4; 5].

For the model-based calculation, individual-animal milk urea levels from the monthly milk recording issued by the Swiss Brown Cattle Breeders' Federation, the Swiss Fleckvieh Cattle Breeders' Federation and the Swiss Holstein Cattle Breeders' Association were used. Average calendar-year levels ranged from 22 to 29 mg dl^{-1}. Whereas the nitrogen level was lower during the winter, milk urea content reached maximum values in the summer feeding period.

The calculated NH_3 emission factors ranged between 22 and 25 g $LU{\cdot}d^{-1}$, depending on altitude level and wind speed (Tab. 2). The NH_3 emission factors therefore, illustrate regional differences in climate and feed level, and reflect the influence of wind speed. Owing to the higher temperatures with the same wind speed, the NH_3 emission factor is larger for the plain region than for the mountain region. Within the same altitude zone, the emission factor based on the higher wind speed is always greater than that based on the lower wind speed. The differences between the individual variants are small. The modelled emission factors are lower than the NH_3 emission factor for the cubicle loose-housing system in Germany of 40 g animal place $^{-1}$ d^{-1} [6] and for the housing including outdoor exercise area in Portugal of 86 g LU^{-1} d^{-1} [7].

Table 2: NH_3 emission factors [g LU-1 d-1] for dairy farming in naturally ventilated cubicle loose housing with solid floors and an outdoor exercise area alongside, with reference to the model-based calculation for both mountain and plain regions and for two wind speeds.

	Emission factor: arithm. mean (95% confidence interval) [g LU^{-1} d^{-1}]	
Variants: Altitude	**Wind speed: 0.3 m s^{-1}**	**Wind speed: 0.5 m s^{-1}**
Plain region	22.7 (12.0; 37.9)	24.5 (13.4; 40.6)
Mountain region	21.8 (12.3; 37.5)	23.4 (13.0; 39.8)

4. Conclusions

Both the measurement concept and the tracer ratio method have proven their worth in practical use in naturally ventilated dairy housing with an outdoor exercise area, and are applicable to other housing systems for cattle as well as other livestock categories. Wind speed in the housing, outside temperature and tank-milk urea content were shown to be significant influencing variables for NH_3 emissions, giving an indication of appropriate mitigation approaches for needs-based, balanced feeding as well as structural and climatic aspects. Milk urea content is a good indicator of an animal's nitrogen supply, and is strongly correlated with total urine nitrogen content. Furthermore, milk urea content is closely linked to NH_3 emissions. Using the model-based calculation approach, we were able to determine regionally differentiated NH_3 emission factors based on widely available underlying data of high temporal and spatial resolution, thereby highlighting differences in climatic conditions and nitrogen levels. A further differentiation of emission factors by flooring design, dung-removal method and the taking into account of grazing remains to be developed.

Acknowledgements

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References

[1] Schrade, S.; Zeyer, K.; Gygax, L.; Emmenegger, L.; Hartung, E.; Keck, M. (2012): Ammonia emissions and emission factors of naturally ventilated dairy housing with solid floors and an outdoor exercise area in Switzerland. Atmospheric Environment (47), 183–194

[2] Vallimont, J.E.; Dechow, C.D.; Daubert, J.M.; Dekleva, M.W.; Blum, J.W.; Barlieb, C.M.; Liu, W.; Varga, G.A.; Heinrichs, A.J.; Baumrucker, C.R. (2010): Genetic parameters of feed intake, production, body weight, body conditions score, and selected type traits of Holstein cows in commercial tie-stall barns. Journal of Dairy Science 93(10): 4892–4901.

[3] Van Duinkerken, G.; Smits, M. C. J.; André, G.; Šebek, L. B. J.; Dijkstra, J. (2011): Milk urea concentration as an indicator of ammonia emission from dairy cow barn under restricted grazing. Journal of Dairy Science 94, 321–335.

[4] Van Duinkerken, G.; André, G.; Smits, M.C.J.; Monteny, G.J.; Šebek, L.B.J. (2005). Effect of rumen-degradable protein balance and forage type on bulk milk urea concentration and emission of ammonia from dairy cow houses. Journal of Dairy Science 88, 1099–1112.

[5] Burgos, S.A.; Embertson, N.M.; Zhao, L.; Mitloehner, F.M.; dePeters, E.J.; Fadel, J.G. (2010): Prediction of ammonia emission from dairy cattle manure based on milk urea nitrogen: Relation of milk urea nitrogen to ammonia emissions. Journal of Dairy Science 93, 2377–2386.
[6] Döhler, H.; Eurich-Menden, B.; Dämmgen, U.; Osterburg, B.; Lüttich, M.; Bergschmidt, A.; Berg, W.; Brunsch, R. (2002): BMVEL/UBA-Ammoniak-Emissionsinventar der Deutschen Landwirtschaft und Minderungsszenarien bis zum Jahr 2010. Umweltbundesamt (Eds.), Texte 05/02, Berlin, 192 pp.
[7] Pereira, J.; Misselbrook, T.H.; Chadwick, D.; Countinho, J.; Trindade, H. (2010): Ammonia emissions from naturally ventilated dairy cattle buildings and outdoor concrete yards in Portugal. Atmospheric Environment 44, 3413–3421

TC-O_16 Assessment of the through-flow patterns in naturally ventilated dairy barns – Three methods, one complex approach

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1. Objectives

Our aim is to understand the air flow patterns in naturally ventilated barns and relate them to typical emission rates [1,2]. The highly variable, spatially heterogeneous flow determines the air exchange rate and is crucial for the transport of pollutants, humidity and heat [3,4]. The observed patterns depend on the inflow and the building design [5]. Only by combining different methodologies (measuring and modelling) the complex flow characteristics can be mapped with sufficient accuracy [6,7].

2. Methodology

In a three-column approach we modeled and measured the through-flow of a naturally ventilated barn:

- Long-term on-farm measurements: A commercial dairy barn with building dimensions 96.15m x 34.20m x 4.2m at the sides and up to 10.7m height at the gable peak was studied [1]. Air was introduced through adjustable curtains in the sidewalls and space boards and doors in the gable walls. The long side wall was facing the prevailing wind direction. Relevant parameters of barn climate were simultaneously monitored at four locations (Fig. 2). Gill ultrasonic anemometers (UA) recorded wind profiles and an ANSYCO GASMET CX4000 Fourier-transformed infrared (FTIR) measurement device monitored gas concentrations - both with 1Hz resolution.
- Experiments in a boundary layer wind tunnel (20m x 3m x 2.3m): The through-flow of a barn model of 1:100, hereafter referred as “scale model”, was studied with laser-light-section and with a Dantec 2D laser-Doppler anemometer (LDA) under controlled lab conditions [6].
- Computational fluid dynamics (CFD) simulations: A quasi-2D large eddy simulation (LES) with a modelling domain of 300m x 1m x 90m (30cm average mesh width) was performed with OpenFoam 2.3.0 to draw conclusions on the air flow patterns for a simplified geometry with open side walls [6]. In addition, 3D-steady-state simulations of the scale model with dimensions 0.962m x 0.342m x 0.107m were conducted with a Reynolds-averaged Navier Stokes (RANS) solver in ANSYS CFX 14.5 [5]. The computational domain was 5.06m x 6.06m x 2.03m and the geometry was mapped in detail.

3. Results and discussion

3.1 Air flow pattern

Studies in the boundary layer wind tunnel and various numerical simulations with an orthogonal inflow (Figure 1b-d) revealed a meandering flow with lower wind speed in the back half of the barn [5,6]. Moreover, there is a recirculation close to the roof. Simulations revealed that there is a vortex under the roof which yields a reverse flow in the upper half of the barn. The interaction between this region and the lower air volume in the barn is small, i.e. the respective air volume is almost not taking part in the air exchange.

As shown in Figure 2 the condition of nearly orthogonal inflow is a typical one. In order to validate our model we consider measurements of vertical profiles of the air flow in the real barn during orthogonal inflow (Figure 1a). The vector field averaged over 10 minutes shows a similar pattern as the models: The direction of the air flow in the lower part of the barn is following the direction of the inflow (left to right in Fig. 1), with a downward component close to the inlet and outlet and a upward component in the middle of the barn. In the upper part of the barn the measured air flow is opposed to the direction of the inflow.

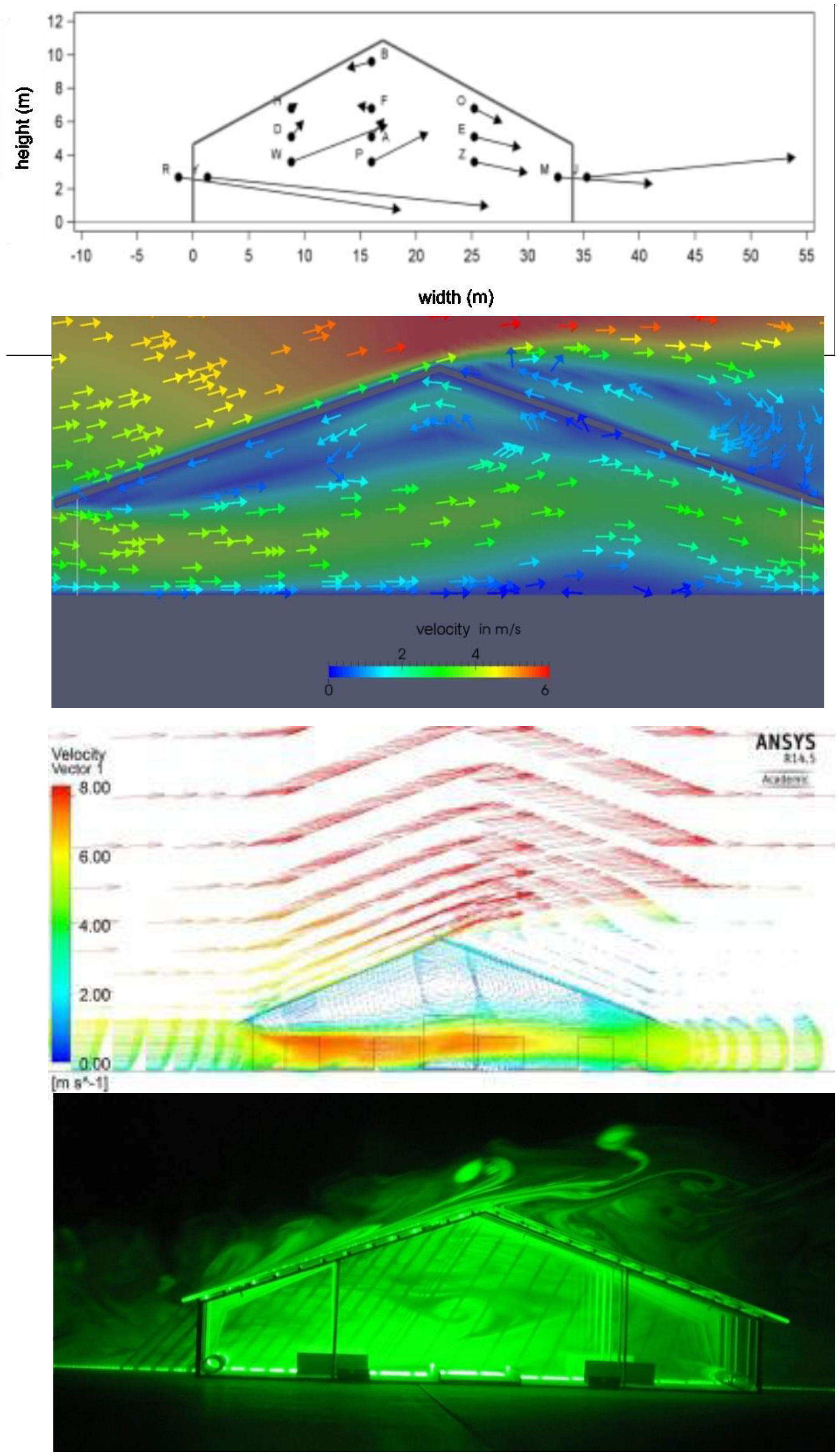

Figure 1: (a) Vector field of a vertical wind profile measured in April 2014 with ultra sonic anemometers during orthognal inflow. Vectors are orthogonal to the long side and scaled by factor 10. An average of 10 minutes is shown. (b) Quasi-2D LES run implementing only the effect of the roof of the barn. The atmospheric boundary layer is parametrized by a logarithmic wind profile with 6.25m/s wind speed in 90m height and roughness length 0.01m. The velocity is averaged over 10 minutes. (c) Vertical profile of the air flow based on a 3D RANS simulation of the scale model. (d) Instanteous flow in the wind tunnel for the scale model with orthogonal, turbulent inflow.

In this way, the measured air flow pattern is in good agreement with the modelled air flow although the model is characterised by strong simplifications.
Since the resolution of the model data is much higher than that of the field measurements we are able to identify also fine scale structures. In particular, we get information on the air flow close to the floor which is difficult to access with measurements due to animal activity. In the front half of the barn the models show high wind speed values in the animal occupied and the emission active zone while wind speed in the back half of the barn is typically much lower. Particularly, there is region about 25m behind the inlet where it is almost calm (Figure 1).

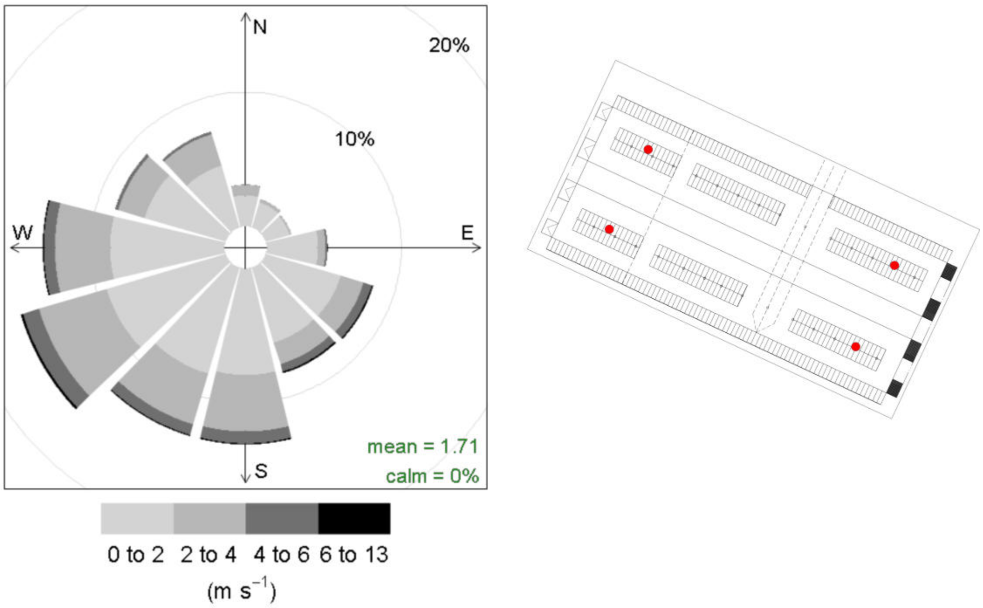

Figure 2: Typical distribution of speed and direction of the incoming wind exemplarily for August 2014 and the cor- esponding alignment of the investigated barn. The red dots indicate measurement points where air flow, gas concentrations, temperature and humidity were monitored simultaneously.

3.2 Distribution of gas concentrations

The observed characteristic air flow pattern indicates that gases which are emitted in the lower back half of the barn will stay for a long time. The air volume in the lower front half of the barn, on the other hand, is exchanged rather fast which reduces the gas concentrations over time.

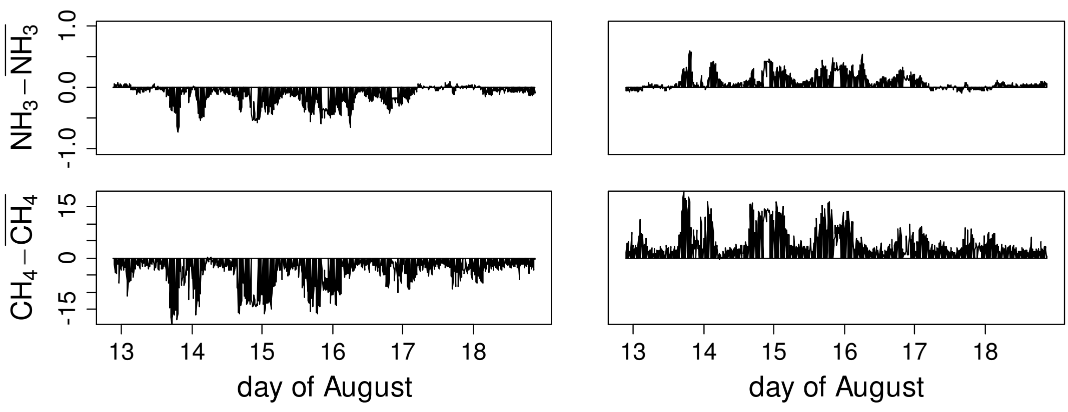

Figure 3: Deviations of partial spatial averages X for front (left subfigures) and back (right subfigures) half of the barn from the total spatial average of the entire barn $\bar{X}$. Here, X refers to concentrations of ammonia and methane, espectively. Temporal averages over 10 minutes are shown.

In order to validate this implication, we compare averages of ammonia and methane concentrations at the measurement positions shown in Figure 2 – partial spatial averages for the front part and the back part of the barn and the spatial average for the entire barn.

Deviations of the two partial spatial averages from the total spatial average reflect the observed air flow pattern: Gas concentrations in the back half of the barn are usually significantly higher than the total average while they are lower in the front half (Fig.3). The deviations are typically more pronounced for methane than for ammonia, which might be related to the typically larger emission rates of methane in dairy buildings. The time series of deviations in the gas concentrations vary with time reflecting the not stationary behavior of the air flow.

4 Conclusion and outlook

Combining different methodologies yields maximal information output in order to determine typical through-flow patterns in naturally ventilated barns. Physical and numerical modeling is crucial to obtain sufficiently high resolved data and to study the influence of boundary conditions. Field measurements are crucial to validate the models and assess the flow patterns regarding their effect on gas concentrations. With the three-column-approach we can combine the advantages of the methodologies and mutually validate the results to improve the understanding of the air flow dynamics which relate to the transport of gases and particles out of and climate in the barn. Even with significant approximations in the models we were able to reproduce the dominating air flow pattern for defined boundary conditions. The results revealed that, for an almost symmetric building regarding inlets and outlets, air flow and gas concentrations are spatially heterogeneous in the barn and not stationary. On average, the front half of the investigated barn is characterized by high wind speed and low gas concentration while animals in the back half may suffer from low wind speed and higher gas concentrations.

The compiled information about air flow patterns, air exchange rate and gas concentrations will be used in further experiments for the estimation of emission rates.

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References

[1] Saha, C.K., Ammon, C., Berg, W., Fiedler, M., Loebsin, C., Sanftleben, P., Brunsch, R., Amon, T. (2014): Seasonal and diel variations of ammonia and methane emissions from a naturally ventilated dairy building and the associated factors influencing emissions. Science of the total environment, 468-469, 53-62

[2] Schrade, S.; Zeyer, K.; Gygax, L.; Emmenegger, L.; Hartung E.; Keck, M. (2012): Ammonia emissions and emission factors of naturally ventilated dairy housing with solid floors and an outdoor exercise area in Switzerland. Atmospheric Environment, 47, 183-194

[3] Saha, C. K.; Ammon, C.; Berg, W.; Loebsin, C.; Fiedler, M.; Brunsch, R.; von Bobrutzki, K. (2013): The effect of external wind speed and direction on sampling point concentrations, air change rate and emissions from a naturally ventilated dairy building. Biosystems Engineering, 114(3), 267-278

[4] Fiedler, M., Berg, W.; Ammon, C.; Loebsin, C.; Sanftleben, P.; Samer, M.; von Bobrutzki, K.; Kiwan, A.; Saha, C. K. (2013): Air velocity measurements using ultrasonic anemometers in the animal zone of a naturally ventilated dairy barn. Biosystems Engineering, 116(3), 276-285

[5] Saha, C.; Fiedler, M.; Amon, T.; Berg, W.; Amon, B.; Brunsch, R.(2014): Assessing effects of different opening combinations on airflow pattern and air exchange rate of a naturally ventilated dairy building - A CFD approach. In: Proceedings. International Conference of Agricultural Engineering AgEng 2014. Zürich, p. 1-4

[6] Fiedler, M.; Fischer, J.; Hempel, S.; Saha, C.; Loebsin, C.; Berg, W.; Amon, B.; Brunsch, R.; Amon, T.(2014): Flow fields within a dairy barn - Measurements, physical modelling and numerical simulation. In: Proceedings. International: Proceedings. International Conference of Agricultural Engineering AgEng 2014. Zürich, p. 1-5

[7] Wu, W.; Zhang, G.; Bjerg, B.; Nielsen, P. V. (2012): An assessment of a partial pit ventilation system to reduce emission under slatted floor - Part 2: Feasibility of CFD prediction using RANS turbulence models. Computer and Electronics in Agriculture, 83, 134-142

TC-O_17 Influence of deep litter on gaseous emissions and microbial composition in a dairy farm

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1. Objectives

Deep litter is an animal housing system based on the repeated spreading of straw or sawdust in indoor booths. As animal dejections are dropped, new layers of litter are regularly added and mixed, so that the bed is kept under aerobic conditions and the composting process is prompted. Deep liter was originally developed for poultry but it is being applied successfully to dairy cows, where it has been associated to some specific benefits:

- Savings in the amount of required litter,
- Improved management of manure and organic fertilization,
- Better animal health due to lower incidence of hoof lesions and mastitis.

While dairy cattle farms are a major source of greenhouse gases (GHG) and ammonia (NH_3) [3], data on the influence of the bedding structure on such emissions is scarce. A previous study was carried out in a dairy farm managed with deep litter in north Europe [8], but no comparative data is available under Mediterranean conditions. Also, information on the effect of deep litter to microbial community dynamics is lacking.
The present work was aimed at characterizing the gaseous emissions that are associated to deep litter when implemented in a Catalan commercial dairy farm, in relation to the conventional management of regular bed replacement. Physicochemical and microbial parameters from bed samples were also studied.

2. Methodology

The study was carried out at the dairy farm "El Trèvol" (Vilobí d'Onyar, Catalonia, Spain), and encompassed five similar barns (65 – 85 cows each). Three barns were handled with deep litter and two were kept as conventional management controls. Deep litter consisted in the daily bed mixing and addition of a thin layer of sawdust, while control beds were set in one time and replaced weekly. Measurements of gaseous emission rates and bed sampling were carried out in spring, summer and winter for including climatic variability. The implemented methods are summarized as follows:

- Emission rates: A dynamic chamber (Lindvall) was used for gaseous sampling [4]; CH_4 was measured by GC-FID [5], and CO_2 and N_2O by GC-ECD [9]. NH_3 was determined in-situ with an electrochemical sensor (VRAE, RAE Systems Inc.) and SH_2 was adsorbed onto an activated coconut shell charcoal tube (with a PTFE prefilter), for subsequent chemical analysis [2].
- Physicochemical characterization: Bed samples were characterized in terms of total solids (TS), volatile solids (VS), total nitrogen (TN) and total ammonia nitrogen (TAN), electrical conductivity and pH, according to the Standard Methods for Wastewater [2].
- Microbial characterization: Micobiome analysis was carried out by high throughput sequencing of ribosomal *16S rRNA* gene libraries from the *Eubacteria* and *Archaeobacteria* domains [1]. Total *16S rRNA* genes from the *Eubacteria* and methyl coenzyme M reductase A (*mcrA*) genes from methanogenic *Archaea* were quantified by qPCR [7].

3. Results and discussion

3.1 Gaseous emissions

The average temperature of the bed ranged from a minimum of 13°C in winter to a maximum of 45°C during the summer. These temperatures were well below 60°C that are common during the termophilic phase of a standard composting process. In addition, differences in bed temperature between barns managed with deep litter and the controls were not significant (n=6, α=0.05). A relatively similar linear correlation (r^2>0.95) was observed in all cases between the average bed temperature and the daily average environmental temperature, so that the bed was consistently about 15°C warmer than the surrounding air.

Different gaseous emission patterns were observed in the bedding: emission rates of GHG were generally lower in deep litter (compared to the controls) during the mild and cold seasons, but the situation reversed during the summer (Figure 1). Average emission rates in deep litter correlated strongly with the bed temperature ($r^2 \geq 0.99$), which points to the importance of the microbial aerobic, anoxic and anaerobic activity resulting in the emission of CO_2, N_2O and CH_4, respectively. Such temperature/emission correlation was not observed in the controls.

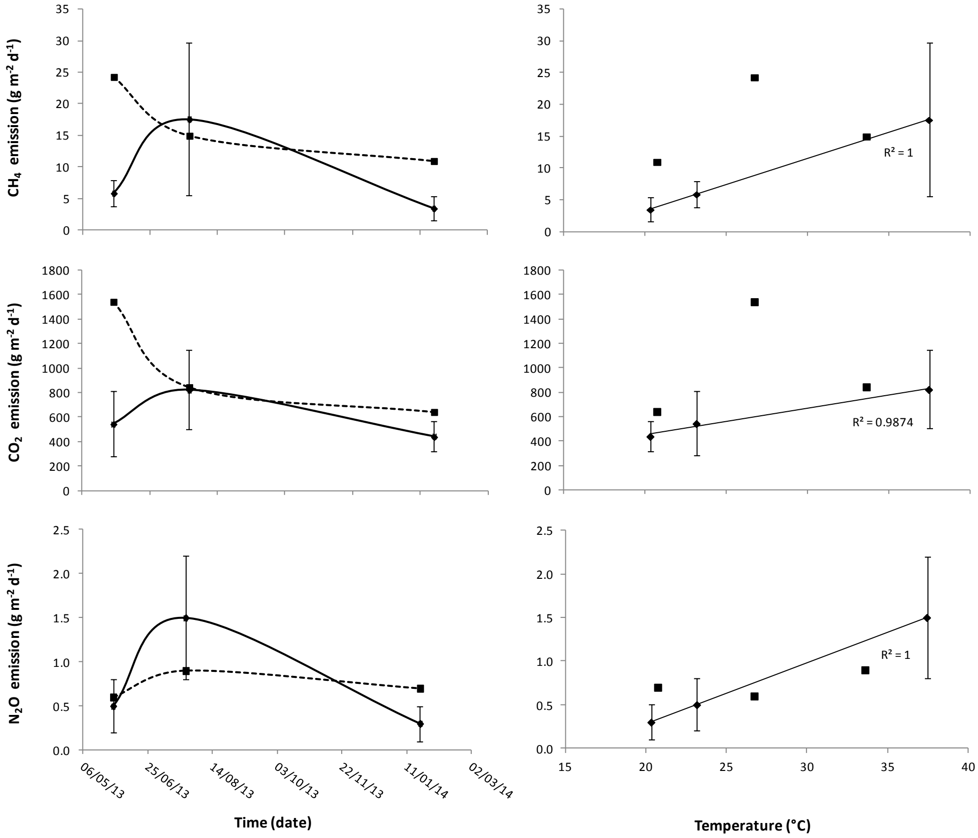

Figure 1: Left: time-course evolution on the GHG emission rates in deep litter (rhombi) and conventional management controls (squares). Right: correlation of these emissions with the average bed temperature (*n*=6). Error bars correspond to the average and confidence interval of three independent barns managed with deep litter (*n*=3, α=0.05); squares are the average of two control management barns.

Conversely, the emission of acidifying and toxic gases (NH_3 and SH_2) showed poor or no correlation with bed temperature (r^2<0.6, not showed) and were always significantly lower in barns managed with deep litter than in the controls. The highest emission reductions were observed for NH_3, with emission rate values that remained below 5 g m^{-2} d^{-1} in deep litter barns during the whole trial, while emission rates ranged from 10 to 30 g m^{-2} d^{-1} in the control barns (Figure 2).

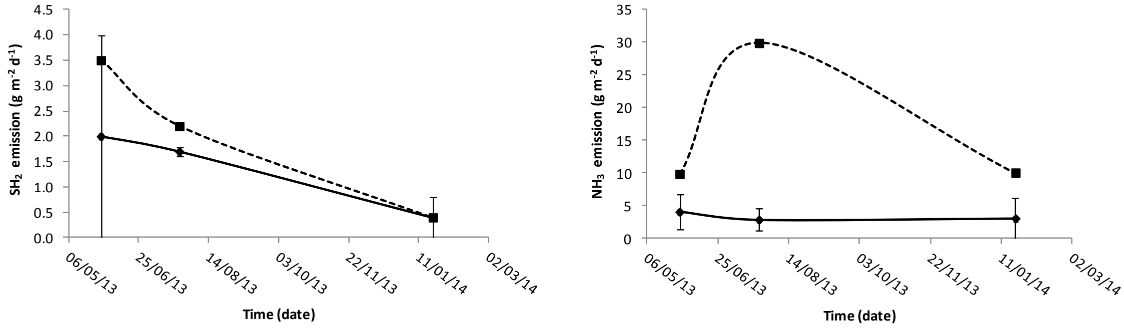

Figure 2: Time-course evolution on the emission rate of acidifying and toxic gases in deep litter (rhombi) and conventional management controls (squares). Error bars correspond to the average and confidence interval of three independent barns managed with deep litter (*n*=3, α=0.05); squares are the average of two control management barns.

3.2 Chemical characterization

Significant differences between deep litter and control beddings were also observed on its physicochemical composition (Table 1). On one hand, TS and VS tended to increase during the warm season in deep litter bed samples, most likely because of water evaporation due to the higher environmental temperatures, while the solids remained rather constant in the continuously renovated control bedding. Yet, at the end of the deep litter process in the winter season, no significant differences with the control were found in terms of TS and VS.

Table 1: Time-course evolution of selected physicochemical parameters of bed samples from deep litter (DL) and conventional control management (CM). Presented values correspond to the average and confidence interval (n=6, α=0.05).

Sampling date	TS (%)	VS (%)	TN (%)	TAN (mg kg^{-1})	EC (dS m^{-1})	pH
DL – Spring (29/05/13)	31±0.13	28±0.10*	0.97±0.08	732±36*	2.86±0.79	8.49±0.15
DL – Summer (24/07/13)	36±0.19*	32±0.33*	1.11±0.23	735±23*	2.69±0.38	8.70±0.13
DL – Winter (21/01/14)	32±0.01	29±0.01	1.11±0.04*	1,020±12*	1.60±0.12	8.71±0.06
CM – Control (pooled)	32±0.37	29±0.42	0.85±0.11	1,435±50	2.67±2.10	8.17±0.52

* Significant difference of DL in relation to CM.

On the other hand, nitrogen (TN) accumulation was higher in deep litter than in the conventional management control but, interestingly, ammonia (TAN) showed the opposite trend as concentrations were significantly higher in the control. This phenomenon points to a more intensive ammonia oxidation and assimilation in deep litter, probably because of the prolonged retention of animal dejections and the higher microbial aerobic activity in the composting deep litter bed. The lower amounts of free ammonium in the bed with deep litter would also explain the observed reduction in ammonia emissions (Figure 2). Finally, the bed pH increased slightly in the deep litter barns in relation to the control but, as with the EC, differences were not significant.

3.3 Microbial characterization

Micobiome analysis of the *Eubacteria* and *Archaea* domains are summarized in Figure 3. With respect to *Eubacteria*, a predominance of the phylum *Proteobacteria* was evidenced followed by representatives of the *Bacteroidetes*. However, during the warm season and only in deep litter samples, the second consistently increased their relative abundance at the expense of the former.

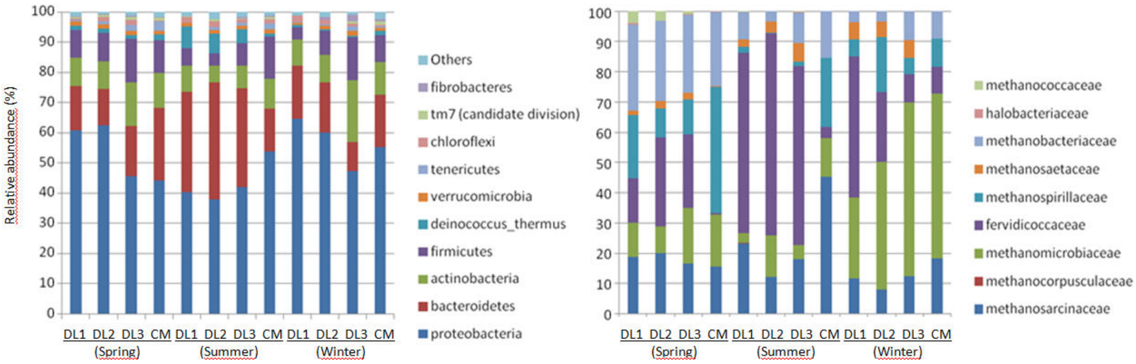

Figure 3: Relative abundance of 16S rRNA sequences from *Eubacteria* (right, at phylum level) and *Archaeobacteria* (left, family level) in bed samples from a dairy farm. Samples were taken from three barns managed with deep litter (DL1, DL2, and DL3) and one control barn with conventional management (CM).

The observed differentiation of bacterial populations was also seen in relation to the *Archaea*. In particular, representatives of the *Fervidococcaceae* family became significantly enriched during de summer period and only in deep litter bedding. This bacterial group has recently been described only recently in anaerobic environments that are characterized by relatively high temperatures and poor availability of organic matter, such as hot springs [6]. It was therefore an unexpected finding to detect these microbes in such prevalence in deep litter samples, fact that is currently being investigated.

A quantitative assessment by qPCR of the relationship between bacterial *16S rRNA* genes (indicator of the overall eubacterial population), and the *mcrA* functional gene (specific from methanogenic archaea) was performed (Figure 4). These results indicate that eubacteria were about two to three orders of magnitude more abundant than methanogens. This ratio is similar to those of anaerobic digesters [7], despite the prevailing aerobic conditions in deep litter. Such high counts on methanogens are probably caused by the microorganisms that are continuously excreted from the cows' rumen. However, it is interesting to note the consistent trend of archaea to increase their number in almost one order of magnitude during the warm season, when the highest methane emission rates were also measured (Figure 2).

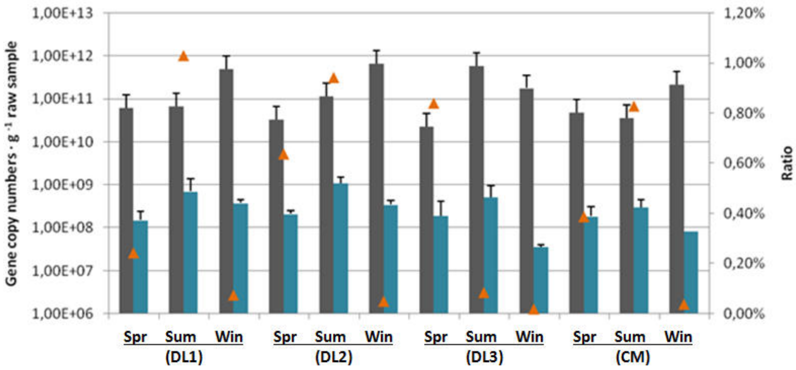

Figure 4: Gene counts of ribosomal *16S rRNA* (total *Eubacteria*, grey bars) and functional *mcrA* (methanogenic *Archaea*, blue bars) as determined by qPCR, in deep litter samples from three barns (DL1, DL2, and DL3) and in one control management barn (CM). Samples were taken during spring, summer, and winter. The ratio between mcrA/16S rRNA has been indicated in triangles.

4. Conclusion and outlook

The emissions reported here in a dairy farm managed with deep litter were comparable to those determined previously in North Europe [6]. These earlier determinations were claimed to be in the lower side, though no comparative data was provided. In the present study, gaseous emissions from deep litter were generally lower than those of conventional bed management, especially for NH_3 (80% reduction in average). Microbial community structure and GHG emission rates were primarily related to bed temperature, but environmental conditions rather than metabolism appear to be the main contributor to the bed heating and, ultimately, to the emission of GHG. A more profound analysis of the generated genomic databases is currently underway, in search for relevant bacterial species (i.e. potentially pathogenic microorganisms involved in bovine mastitis such as *Staphylococcus aureus*).

Acknowledgements

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References

[1] Affes, R.; Prenafeta-Boldú, F.X.; Flotats, X.; Battimelli, A.; Godon, J.J.; Steyer, J.P.; Palatsi, J. (2013): Microbiome dynamics during the anaerobic treatment of slaughterhouse wastes. In: Study of Methods for the Improvement of the Anaerobic Digestion of Lipids and Long Chain Fatty Acids. Doctoral Thesis, UPC, Barcelona

[2] APHA; AWA; WEF (2005). Standard methods for the examination of water and waste water, 21th ed. American Public Health Association/American Water Works Association/Water Environment Federation, Washington, DC, USA

[3] Gerber, P.J.; Steinfeld, H.; Henderson, B.; Mottet, A.; Opio, C.; Dijkman, J.; Falcucci, A.; Tempio, G. (2013). Tackling climate change through livestock – A global assessment of emissions and mitigation opportunities. Food and Agriculture Organization of the United Nations (FAO), Rome

[4] Lindvall, T.; Noren, O.; Thyselius, L. (1974). Odour reduction for liquid manure systems. Transactions of the ASAE 17, 508-512

[5] Palatsi, J.; Illa, J.; Prenafeta-Boldú, F.X.; Laureni, M.; Fernandez, B.; Angelidaki, I.; Flotats, X. (2010). Long-chain fatty acids inhibition and adaptation process in anaerobic thermophilic digestion: Batch tests, microbial community structure and mathematical modelling. Bioresource Technology 101, 2243-2251

[6] Perevalova, A.A.; Bidzhieva, S.K.; Kublanov, I.V.; Hinrichs, K.U.; Liu, X.L.; Mardanov, A.V.; Lebedinsky, A.V.; Bonch-Osmolovskaya, E.A. (2010). *Fervidococcus fontis* gen. nov., sp.nov., an anaerobic, thermophilic crenarchaeote from terrestrial hot springs, and proposal of *Fervidococcaceae* fam. nov. and *Fervidococcales* ord. nov. International Journal of Systematic and Evolutionary Microbiology 60. 2082-2088

[7] Rodríguez-Abalde, A., Guivernau, M., Prenafeta-Boldú, F.X., Flotats, X., Fernández, B. (2013). Characterization of microbial community dynamics and its correlation with physic-chemical parameters during the anaerobic codigestion of thermally pre-treated slaughterhouse wastes. In: Anaerobic Digestion of Animal by-Products: Pretreatments and co-digestion. Doctoral Thesis, UPC, Barcelona

[8] Sommer, S.G.; Dahl, P.; Rom, H.B.; Moeller, H.B. (1998): Emission of ammonia, nitrous oxide, methane and carbon dioxide during composting of deep litter. In: Ramiran 98: Proceedings - International Conference on Management Strategies for Organic Waste Use in Agriculture. Martinez, J.; Maudet, M.-N. (eds.). FAO, Rome

[9] Wang, C. (2010). Simultaneous Analysis of Greenhouse Gases by Gas Chromatography. Technical Note. Agilent Technologies, Inc. Shanghai (China)

TC-O_18 Reduction of ammonia emission from broiler houses by use of a heat exchange system

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1. Objectives

A heat exchange system is an air exchange system that recovers the thermal energy of air leaving a housing system to heat incoming air by a counter-current heat exchange system. The air exchange system is utilised to recover the thermal energy of the air drawn out from broiler houses to heat up the inflowing air and by that reduce the energy cost of heating. The use of a low cost heating system may influence the dryness of the broiler litter and thereby abate the ammonia emission of broiler production. The aim of the study was to investigate to what extend the use of a heat exchange system effects the emission of ammonia from broiler houses.

2. Methodology

The Dutch company Agro Supply has developed an Agro Clima Unit (ACU) for broiler houses (Figure **6**). The ACU system reduces the energy requirement for heating broiler houses by means of counter current heat exchange system [1]. However, the use of the ACU system may also influence the emission of ammonia from the broiler production as the inflowing ACU-heated air dry the broiler litter mat. The lower humidity of the broiler mat reduce the conversion of excreted uric acid into ammonia, and thereby the emission of ammonia from the broiler house.

The ammonia effect of the ACU heat exchange system was tested by simultaneous measurement of the ammonia emission from approximately 30.000 broilers housed in a mechanically ventilated commercial broiler house attached the ACU unit (case), and 30.000 broilers housed in a similar housing system without the ACU unit (control). Apart from the ACU system the two broiler houses were identical. Both houses were ventilated by traditional wall inlets and classic negative pressure ventilation system. The ammonia emission was continuously measured in three production cycles, each lasting approximately 30 days. The emission was quantified by continuous measurements of air exchange and ammonia concentration of in- and outflowing air.

The air exchange was continuously on-line measured by air anemometers (Stienen, d = 600mm) situated in ventilation ducts. The concentrations of ammonia in ingoing and outgoing air were continuously on-line measured by use of an on-line automatic photo-acoustic multigas analyser (INNOVA, 1412, Lumasense, Denmark), connected to an automatic mulitipoint sampler (INNOVA, 1307).

The emission of ammonia from the test houses was quantified by the following equation:

$$E_{NH3_t} = \sum_{i=1}^{i=3} \sum_{t=1}^{t=n} V_{i_t}\, x\, (C_{in_{i_t}} - C_{out_t}) \qquad \text{Eq. 1}$$

where
E_{NH3_t}= Emission of ammonia from the housing systems at time t, mg NH_3 h^{-1}
i = Type of ventilation (roof ventilation, Agro Clima Unit ventilation, gable wall ventilation)
n = the number of measurements during the sampling period
V_i = Air flow, m^3 air h^{-1}
C_{in_t} = Ammonia concentration in outgoing air at time t, mg NH_3 m^{-3} air
C_{out_t} = Ammonia concentration in ingoing air at time t, mg NH_3 m^{-3} air
t = measurement event

Figure 6: Picture of the Agro Clima Unit situated outside a broiler house. Part of the ventilation air to and from the broiler house are drawn through the Agro Clima Unit by a countercurrent principle to utilise the heat content of outgoing air to heat up ingoing air.

3. Results and discussion

The ammonia emission from broilers produced in equal commercial test houses with and without the ACU heating system was quantified continuously from the start of the production cycles until a fraction of the broilers was taken out of the production system approximately 30 days later. The ammonia effect of the ACU system was quantified for three subsequent broiler production cycles.

Table 1 shows the time of study, and the number of broilers housed in the test sections during the three test periods.

Table 1: Start and end of production cycles and the number of broilers at start and end of the production cycles.

Test period	Test section	Start of production cycle	End of measurement period	No. of broilers at start	No. of broilers at end
1	ACU	16.08-2012	17.09-2012	30.100	29.516
1	Control	16.08-2012	17.09-2012	30.900	30.260
2	ACU	05.10-2012	05.11-2012	31.900	31.409
2	Control	05.10-2012	05.11-2012	31.900	31.412
3	ACU	22.11-2012	24.12-2012	30.500	29.729
3	Control	22.11-2012	24.12-2012	30.400	29.748

The daily measured ammonia emission from the broilers housed in broiler houses with and without the ACU system is shown in Figure 2.

Levels of emission were found to increase during the production cycle for all three production periods (Figure 2). In test period 1 the ammonia emission was found to be lower from the broilers produced in the test section attached the ACU system for the first 20 days of the production cycle. In test period 2 lower emission from the test section attached the ACU was observed during the full period of measurement. In test period 3 reduced ammonia emissions were observed from the case section during the first 25 days.

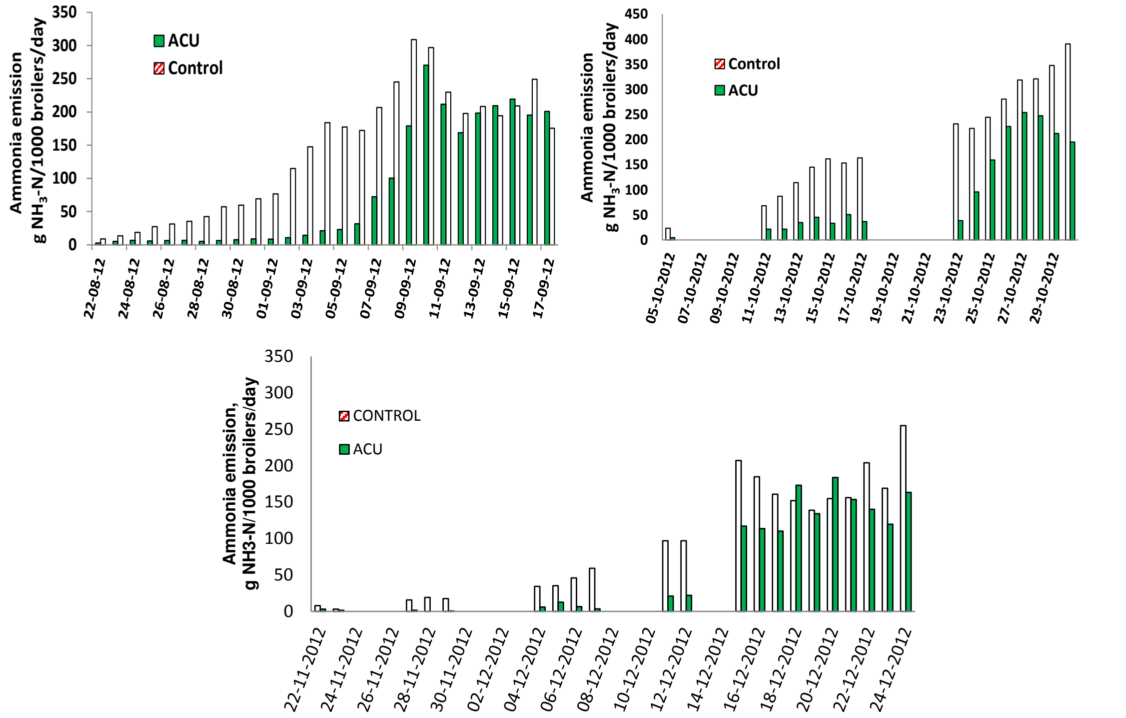

Figure 2: Measured daily emission of ammonia nitrogen (NH_3-N) from broilers produced in a broiler house attached the Agro Clima Unit (ACU) and a broiler house without the use of the Agro Clima Unit (Control). Ammonia volatilezation is shown in g of ammonia nitrogen (NH_3-N) per 1000 broilers per day. The top left diagram shows results for broilers produced during August and September. The top right diagram shows the results during October and November. The lower diagram shows results for broilers produced during November and December. Missing data is a consequence of measurement equipment malfunction.

The ventilation requirement of newly hatch broilers is low. Due that, the majority of the ventilation requirement took place by use of the ACU system in the start of the production cycles, whereas the majority of the total air exchange was performed by the roof ventilation system later in the production cycle, when the ventilation requirement of the broilers is higher (Figure 3).

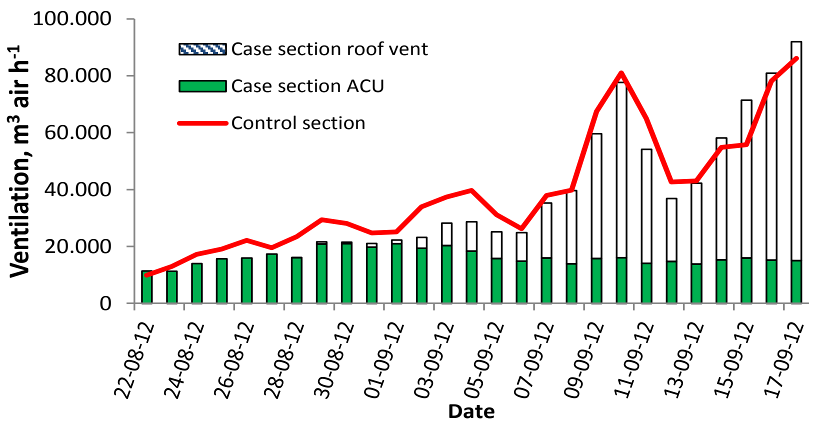

Figure 3: Ventilation by the different types of ventilation systems in period 1. The case section was ventilated by a combination of the ACU and the roof ventilation system (Roof vent). The control section was ventilated only by roof ventilation system. Gable fans were not used in any of the three measuring periods.

The highest reduction of ammonia emission was observed when the majority of the ventilation of the case section was performed by the ACU system. In general, it was observed, that the higher proportion of the total ventilation requirement performed by the roof ventilation system, the closer were the ammonia emission levels of the test and case sections (Figure 2).

The total ammonia emission per 1000 broilers produced in housing systems with and without use of the ACU system can be seen in Table 2. The study found differences between measured ammonia emission for the different test periods both for the control and the case section. The observed differences is expected to be due differences in the external climatic conditions, as air humidity influences the moisture content of the litter mat and thereby the ammonia emission potential. In average the ammonia emission was found to be 41% lower from the broiler section with the ACU unit attached.

Table 2: Measured ammonia (NH_3) emission from a control and a case broiler house attached a Agro Clima Unit (ACU), and the ammonia reduction effect of the ACU system per 1000 broilers during a 30 days production period.

Test period	Period	Control Kg NH_3/1000 broilers	Case (ACU) Kg NH_3/1000 broilers	Difference Kg NH_3/1000 broilers	Difference %
1	Aug – Sep	5.07	2.96	2.11	41.6
2	Oct – Nov.	7.01	3.59	3.41	48.7
3	Nov – Dec	3.84	2.58	1.26	32.7
Mean	Aug – Dec	5.30	3.04	2.26	41.0

4. Conclusion and outlook

The ammonia emission from broiler houses with and without an ACU heat exchange system was measured for three different broiler production cycles. The use of the ACU heat exchange system was found to reduce the ammonia emission from broiler houses by 41 %.

The ACU system is a promising environmental technology for broiler poultry production, as it both acts as an energy saving technology, and reduces the emission of ammonia from the production. The ammonia reduction effect of the ACU system appears to be depending on the proportion of the total ventilation requirement that can be performed by the ventilation capacity of the ACU system.

As the total ventilation requirement of broiler houses is higher in summer periods than in winter periods, the ammonia effect of the ACU system can be expected to be depending on when the study is performed. The present study was performed during autumn and early winter periods. To investigate the ammonia effect of the ACU system during other seasons, a new comparative study running over 12 month has been initiated to quantify the ammonia and odour effect of the ACU systems during a full year.

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References

[1] Adamovský, R. Adamovský, D., Kára, J. Neuberger, P. (2008):
Verification of the effects of the secondary heat recovery from ventilation air in an animal house for the fattening of broilder chickens. Infrastruktura I ekologia Terenow Wiejskich. Infrastructure and ecology of rural areas, 6, 51-61. Journal 1, 12-18

TC-O_19 Greenhouse gas and ammonia emissions from different dairy production systems in Germany

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1. Objectives

Dairy production is predicted to strongly increase in Europe, as milk quotas are going to be abolished. This will also affect the greenhouse gas and ammonia emissions. It is necessary to identify low emission production systems and mitigation options. Their effects on the whole process chain and their applicability into regional settings need to be considered. Therefore, dairy farms representing the existing range in production systems have been modelled and emissions of NH_3 and GHG have been calculated.

2. Methodology

Five dairy farms were modelled with average milk yields from 5200 to 11850 kg ECM 5 200 and 11 850 kg ECM (cow · a)$^{-1}$. Thus the existing range of production intensities in Germany was covered. These farm models are derived from surveys of existing dairy farms in Germany and Italy (table 1). Global values like feedstuff properties as well as standard values are taken from various sources.

Table 1: Main characteristics of dairy farms.

Farm model		MV1	MV2	MV3	MV4	MV5
Milk yield	kg ECM · a^{-1}	5 195	9 322	8 820	11 848	8 496
Size of herd	Number of dairy cows	47	151	48	55	67
Breed		VW 80 % RH 20 %	HF	BV	HF	FL
Lactation periods per cow		3.2	2.4	2.9	3.4	2.7
acreage for on-farm fodder	Arable	9 %	72 %	0 %	87 %	40 %
	Grassland	91 %	28 %	100 %	13 %	60 %
N-input (on-farm fodder)	kg N · GU^{-1}	3.6	2.8	4.5	3.3	3.0
Region		Black Forest	South Tyrol	Allgäu	Lower Rhine	Central Franconia

ECM-energy corrected milk; VW-Vorderwälder Vieh; RH-Rotes Höhenvieh; HF-Holstein Friesian; BV-Braunvieh; FL-Fleckvieh; GU-grain unit

The modelling approach applied integrates the following compartments: feed production (emissions from soil and cultivation), enteric fermentation and excretion, housing systems and manure management. Nutrient and product-flows between these compartments were considered and resulting GHG and NH_3 emissions calculated. The compartments are linked via mass flow of products. The calculation of GHG and ammonia emissions is based on the methodology of the National Inventory Report (NIR) [1]. Since the NIR approach is source related and highly aggregated on regional and national level it is not suitable for farm level modelling and therefore needed to be adopted for farm level modelling, e. g. by including pre-chain emissions for utilities.. Emission calculations on farm level have to illustrate relationships and effects in much more detail. The dairy system is modelled within the system boundaries of a typical dairy farm. Rearing of calves and heifers for replacement are part of the system, whereas excess calves are regarded to be sold and leave the system. Results of emission calculations are given as kg CO_2e for GHG and kg NH_3 per kg energy-corrected milk.

Dairy farms are multiple output systems with milk and beef as main products. Thus, the calculation of product related GHG emissions depends on an allocation of emissions between products. The GHG emissions are allocated according to the IDF-methodology guidelines for carbon foot printing in the dairy sector which is based on the physiological fodder energy demand for milk and beef production [2].

3. Results and discussion

3.1 Greenhouse gas emissions

When allocated completely to milk, GHG emissions range between 0.8 (MV4) and 1.9 kg CO2e · (kg ECM)$^{-1}$ (MV1). About half of the GHG emissions are caused by field activities (41 % for MV5 to 54 % for MV3), in particular N fertilizing. The second most important source is enteric fermentation, which contributes between 34 and 40 % to overall GHG emissions. The specific GHG emissions are strongly related to average milk yield per cow (r = -0.89). Grass and maize pellets in the feed ration like in MV3 increase GHG emissions due to the high energy demand to produce this forage. While grass and maize pellets account only for 4.6 % (dry matter related) of the total feed ration, they contribute to 13.5 % of the GHG emission from total feed production (on-farm and purchased) (figure 1).

Depending on the housing system and manure management (manure removal intervals, storage type, and storage time) great differences in emissions of GHG and ammonia from barn and storage can occur. Transport of roughage that cannot be grown in the vicinity of the farm only marginally contributes to the total emissions. Even transport distances of 250 km for the whole roughage in MV2 increase overall emissions marginally by 4 % (figure 1).

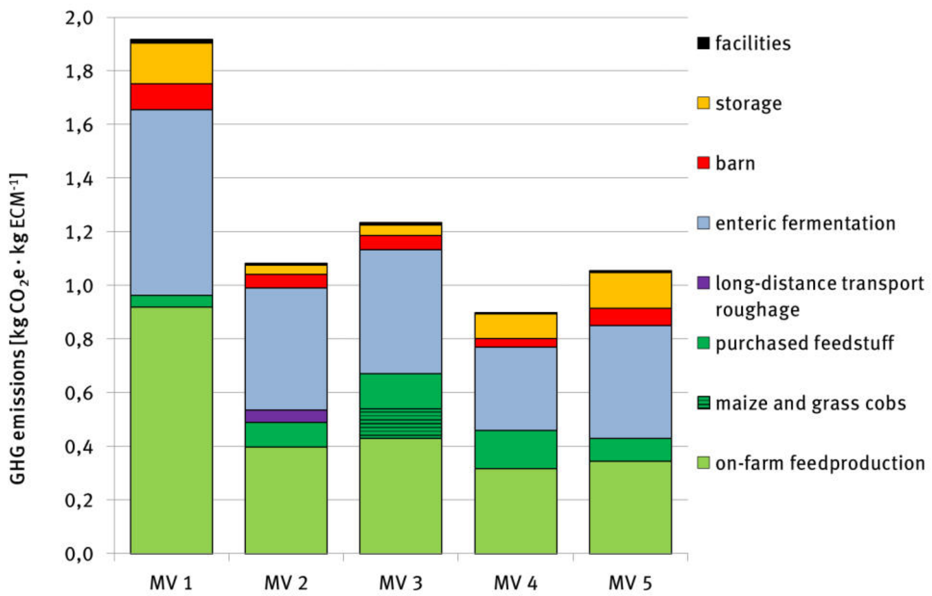

Figure 1: Sources of greenhouse gas emissions per kg ECM in the investigated dairy farms (allocation 100 % to milk).

N input per grain unit for on farm feedstuff production is higher in farms with high proportions of grassland (table 1) (r = 0.71), which is partially a result of higher losses of climate effective N compounds from manure application on grassland. Therefore, high shares of grassland are positively related to overall GHG emissions (r = 0.68). The same applies to NH_3 emission, due to less efficient slurry application techniques used on grassland on the modelled farms (r = 0.86). The positive relationship between grassland and GHG emission is reduced by the fact that nitrate leaching and therefore indirect N_2O emission from nitrate leaching is more pronounced on arable fields than on grassland [3].

GHG emissions from slurry storages range from 8.9 to 56.6 kg CO_2e per m³ slurry. The most important component of these GHGes is methane, which is modelled as a function of slurry volume, storage time and temperature. The effect of storage time is pronounced (r = 0.89) (table 2). However, long storage time in winter is recommended to enable slurry application adapted to nutrient demands of crops. This is crucial to achieve high N-efficiencies. One promising measure to reduce GHG emission from slurry storages is anaerobic digestion and capturing of methane.

Table 2: Influence of slurry storage[a] technique on GHG and NH_3 emissions.

	Farm model	MV1	MV2	MV3	MV4	MV5
Store 1	Cover type	(slurry crust)	tent	concrete	concrete	(slurry crust)
	Homogenization p. a.	6	5	12	4	3
	Storage time [d]	183	73	91	120	150
	Stored volume p. a. [m³]	980	4316	220	1163	1220
	GHG emission [kg $CO_2e \cdot m^{-3}$]	33.9	8.9	10.8	33.9	34.1
	NH_3 emission [kg $NH_3 \cdot m^{-3}$]	0.32	0.019	0.27	0.049	0.18
Store 2	Cover type			(slurry crust)	none	(slurry crust)
	Homogenization p. a.			12	4	2
	Storage time [d]			91	120	210
	Stored volume p. a. [m³]			963	238	521
	GHG emission [kg $CO_2e \cdot m^{-3}$]			13.8	36.1	56.9
	NH_3 emission [kg $NH_3 \cdot m^{-3}$]			0.64	1.37	0.18

[a] MV1 and MV2 has one slurry store each, MV3, MV4 and MV5 has two slurry stores each

3.2 Ammonia emissions

Ammonia emissions range between 6.7 and 11.3 g $NH_3 \cdot (\text{kg ECM})^{-1}$ (figure 2). Differences are mainly caused by manure management, but also by milk yield per cow. Ammonia emissions from on-farm feed production is the most important source in all five farm models. Differences are mainly caused by different manure application techniques. MV1, MV3 and MV5 use broadcast application technique on grassland as well as on arable soil with slurry incorporation within 4 hours where possible, causing higher NH_3 emissions (6.1, 5.3 and 4.4 g $NH_3 \cdot kg^{-1}$ ECM) than in MV2 and MV4 (4.4 and 3.0 g $NH_3 \cdot kg^{-1}$ ECM) which use trailing hose technique.
NH_3 emissions from manure storage in MV1, MV3 and MV4 are significantly higher (1.57, 1.72 and 1.48 g $NH_3 \cdot kg^{-1}$ ECM, respectively) than in MV5 und MV2 (both 0.73 g $NH_3 \cdot kg^{-1}$ ECM). NH_3 emissions from manure storages depend mainly on contact area between manure and air and air exchange rate. Therefore, NH_3 emissions from manure storage are especially high in open tank storages. Reduced slurry crusting due to slurry homogenization or lacking slurry crusts enhance emissions. For example, in MV4 with two slurry storages (storage 1 with concrete cover, storage 2 with no cover and no slurry crust) volume related emissions in storage 2 are 28 times higher than in storage 1 (1.37 vs. 0.049 kg $NH_3 \cdot m^{-3}$ slurry) (table 2).
Differences in NH_3 emissions related to kg ECM from barns are mainly caused by differences in milk yield per cow. NH_3 emissions from barns related to animals instead of kg ECM are in a rather narrow range of 18.1 kg NH_3 to 19.4 kg NH_3 per cow (incl. replacement animals) and year. Ammonia emissions from purchased feed are low, because they are produced mainly with mineral fertilizers.

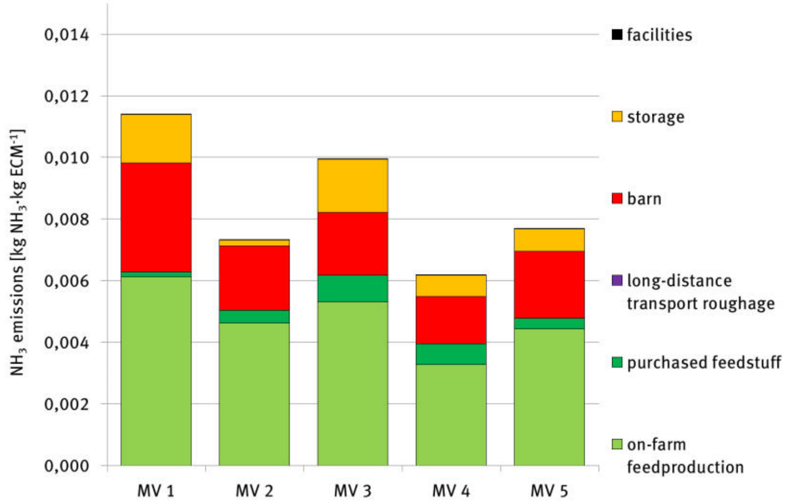

Figure 2: Sources of ammonia emissions per kg ECM in the investigated dairy farms (allocation 100 % to milk)

3.3 Allocation

If GHG and NH_3 emissions are allocated between milk and beef production according to the IDF methodology, systems with high amounts of produced beef are in advantage compared to those with less beef. Especially dairy farms with dual-purpose breeds (milk and beef production) like MV1 (Vorderwälder Vieh and Rotes Höhenvieh), MV5 (Fleckvieh), and less pronounced MV3 (Braunvieh) account lower GHG emission to milk than those with Holstein Friesian cattle. Differences range between 12 and 30 % per kg ECM with the higher values for those farms with dual-purpose breeds (table 3).

Table 3: Influence of IDF allocation on GHG emissions.

Farm model		MV1	MV2	MV3	MV4	MV5
beef	g · (kg ECM)$^{-1}$	73.20	32.99	33.99	24.44	45.01
100 % on milk	g CO_2e · (kg ECM)$^{-1}$	1.92	1.08	1.23	0.90	1.06
IDF allocation	g CO_2e · (kg ECM)$^{-1}$	1.34 (-30 %)	0.86 (-20.4 %)	1.01 (-17.9 %)	0.79 (-12.2 %)	0.81 -23.6 %)

4. Conclusion

The most promising measure to reduce NH_3 emissions is a well-balanced nitrogen management, like utilization of N efficient manure application techniques and covers for manure storages. This measure also efficiently reduces GHG emissions. Since methane from enteric fermentation and manure storage are main components of GHG emissions from dairy farms, it is important to increase milk and beef production efficiency and to avoid release of methane from manure storage into the atmosphere. Since methane emissions from enteric fermentation mainly depend on the number of animals, increased milk and beef yield per animal decreases product related emissions.

Acknowledgements

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References

[1] Rösemann, C. et al. (2013): Calculations of gaseous and particulate emissions from German agriculture 1990 - 2011. Report on methods and data (RMD). Thünen-Report 1. Johann Heinrich von Thünen-Institut. Braunschweig. 394 p.

[2] IDF (International Dairy Federation; 2010): A common footprint approach for dairy. Bulletin of the IDF 445/210. IDF, Brussels, Belgium.

[3] Vandré, R. et al. (2014): Weiterentwicklung der integrierten Stickstoff-Bilanzierung als Grundlage für landwirtschaftliche Minderungsstrategien zur Unterstützung der deutschen Nachhaltigkeitsstrategie. Umweltbundesamt (Hg.) Schlussbericht. FKZ 3711 92 236, Darmstadt (in preparation).

TC-O_20 Excretion of volatile solids by livestock to calculate methane production from manure

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1. Objectives

Methane contributed in 2013 10% to the Dutch Greenhouse gas emissions of which 63% came from agriculture. Methane is produced by enteric fermentation (72%) and by fermentation of volatile solids from the excreted manure (28%). Methane emission from manure management is estimated from the excretion of volatile solids (VS) by livestock, the potential methane production (B_o) and the Methane Conversion Factor (MCF) [1, 3]. In the Netherlands, the excretion of volatile solids (VS_{EX}) for animal categories representing a significant share of the Dutch emissions, is calculated as a product of 1) estimates of manure production and 2) measured data on average volatile solids concentrations in manure samples analyzed in commercial laboratories [2].This method may result in inaccurate estimates because of inaccurate estimates of manure production and of inaccuracy of the average concentration of VS in manure. Besides, the latter is biased because it concerns measurements before application of manure to the field. During storage in housing and outside, volatilization of mineral nitrogen compounds, organic odorous compounds, dust particles and water already occurred. This paper suggests to revise the Dutch method with a more robust approach based on the excretion of volatile solids (VS) calculated with data of intake, digestibility of organic matter (DOM) and excretion of Total Ammoniacal Nitrogen (TAN). Additionally the outcomes are compared to the excreted VS calculated with the Tier 2 method in the IPCC 2006 Guidebook [3].

2. Methodology

In the Netherlands comprehensive data are available on feed intake, feed composition and digestibility of key-categories of livestock. In such situation, the IPCC [3] recommends to use a Tier 2 method to calculate annual VS excretion (VS_{EX}) according Eq. 1.

$$VS_{EX} = (GE\times(1\text{-}DE\%/100)+(UE\times GE))\times((1\text{-}ash)/(\frac{GE}{1000}))\times 365) \qquad \text{Eq.1}$$

In which, GE is the gross energy intake (MJ/d); DE% digestible energy; UE is urinary energy as fraction of GE; ash is the ash content as fraction of dry matter intake. However, the CVB Feed Tables (chemical composition and feed values are assembled for all species of livestock animals) [4] provide information on GE, ME and ash of feed components but lacks the values for DE and UE (UE=DE-ME). This could be overcome by using the DE values of diets for various livestock categories suggested by the IPCC [3]. However, it is unknown if these values are accurate for feed components and animal diets that are typical for the situation in the Netherlands. This uncertainty urged the need to develop a new method to estimate VS_{EX} of various key-categories of livestock given the extensive information on feed intake and feed digestibility available in the Netherlands.

The excretion of VS in manure is the sum of VS excreted with feces, as a result from the (un)digestibility of the organic matter (DOM) in the feed ($VS_{EX\text{-}DOM}$) and urine ($VS_{EX\text{-}TAN}$) as a result from the digestibility of nitrogen. $VS_{EX\text{-}TAN}$ is defined as urea (cattle and pigs) or uric acid (poultry) and thus relates to TAN. The excretion of TAN is already used in the Tier 3 TAN-flow model of the Netherlands, NEMA (National Emission Model for Agriculture) to estimate the national ammonia (NH_3) and Nitrous oxide (N_2O) emissions. TAN is calculated from intake of crude protein nitrogen (N), digestibility of N and N retained in animal products [5, 6]. To do so, the Dutch Central Bureau of Statistics (CBS) and Wageningen UR are commissioned by the government to collect the required data. These data are the number of animals, the yields of animal products, the production and imports of home-grown feeds and forages, and the volumes of purchased feeds (concentrates and by-products) for each livestock category and sub-categories. The ingredients of the feed and the by-products are derived from data from the feed industry. Feed and nutrient digestibility's can thus be calculated with the CVB Feed Tables [4, 7]. These comprehensive data not only enables the calculation of $VS_{EX\text{-}TAN}$, but is also the framework for calculating the excretion of $VS_{EX\text{-}DOM}$ by knowing the organic matter content en the DOM of the compounds in the feed.

The total VS_{EX} is the sum of $VS_{EX\text{-}TAN}$ and $VS_{EX\text{-}DOM}$. The excretion of $VS_{EX\text{-}TAN}$ is already provided by NEMA and is calculated as follows:

$$VS_{EX\text{-}TAN} = \left(\left(\sum_{i=n}^{n} (DMI_i \times (CP_i \times CPD_i/6.25)\right) - N_{RET}\right)/P_N \qquad \text{Eq. 2}$$

In which, DMI_i is the dry matter intake (kg/year) of feed component i (i = n the number of feed components in the diet), CP_i is the crude protein concentration in feed component i, CPD_i is the digestibility coefficient of CP in feed component i; N_{RET} is the nitrogen retention (in animals, milk and eggs), P_N is the proportion of N in the excreted $VS_{EX\text{-}TAN}$. The values of DMI_i are obtained from compilations by the data of [7, 8] The values CP_i and CPD_i are derived from the CVB Feed Tables [4]. In cattle and swine, nearly all $VS_{EX\text{-}TAN}$ is excreted as urea (CH_4N_2O; molecular mass 60 g/mol). Therefore, in cattle and swine P_N = 28/60, with 28 being the molecular mass of two molecules of nitrogen. In poultry, nearly all $VS_{EX\text{-}TAN}$ is excreted as uric acid ($C_5H_4N_4O_3$; molecular mass 168 g/mol) and therefore, in poultry P_N = 56/168, with 56 being the molecular mass of four molecules of nitrogen.

The fecal excretion of VS ($VS_{EX\text{-}DOM}$) is calculated from the organic matter intake and its digestibility calculated as:

$$VS_{EX\text{-}DOM} = \sum_{i=n}^{n} (DMI_i \times (1000\text{-}ash_i)/1000)\text{-}(DMI_i \times 1000\text{-}ash_i)/OMD_i) \qquad \text{Eq. 3}$$

In which, DMI_i is the dry matter intake (kg/year) of feed component i (i = n the number of feed components in the diet), ash_i is the ash content (g/kg DM) of feed i and OMDi is the digestibility coefficient of organic matter of feed i. derived from the CVB Feed Table [4]. Note that DOM is the product of OM and OMD, with OM being the organic matter content of the feed.

3. Results and discussion

Table 1 presents the estimated VS_{EX} of various key-categories of livestock calculated according to three methods: 1) IPCC 2006 Tier 2 method using the range of DE values as suggested by IPCC; 2) Current method used in the Netherlands (NL_current) [1, 6] and 3) the revised method presented in this paper based on excretion of TAN and fecal excretion of organic matter (TAN+DOM).

Estimation of VS_{EX} with Eq. 1 is sensitive to errors in DE values: 10% error in estimating DE will be magnified to 20 to 45% for VS_{EX} [3]. The IPCC suggests a broad range of DE values of animal diets, which therefore could introduce large inaccuracies with differences of VS_{EX} calculated with the DE_{max} being 30-60% lower than calculated with DE_{min} (Table 1).

Except for the rearing heifers and the broilers all VS_{EX} calculated currently in the NL were lower than calculated with the IPCC's upper limit of the DE%, which varied from 75% for ruminants to 90% for the fatteners. A higher DE% results in a lower VS_{EX}. This suggests that the NL_current values were too low, since a higher DE than the upper DE is not likely, in particular because restriction of animal products in feed would have had a decreasing effect on the organic matter and the energy digestibility of feed.

In general the suggested TAN+DOM method in this paper had higher VS_{EX} compared to the NL_current, which is coherent with the fact that the NL_current would be too low because volatilization of VS during storage in the house and outside was not accounted for. For heifers and dairy cows the differences were small and even lower TAN+DOM values occurred. This was probably due to higher volumes of manure, rather than high estimates of the VS concentration [2, 8; not presented in this paper].

Except for dairy cows and broilers, the TAN+DOM calculations resulted in estimates of VS_{EX} within the range of the estimates of the IPCC method. The lower estimates of dairy can be explained by the ration of Dutch dairy cows with high quality forage and a high amount of concentrates. The higher estimates of the VS_{EX} of broilers by the TAN+DOM method is due to a lower energy digestibility of Dutch feed than estimated by the IPCC's DE with a lower limit of 85%. For Dutch broilers this is more close to 70-75% [4].

Except for pigs and broilers the calculated VS_{EX} with TAN+DOM were close to those calculated with the highest DE%. This is coherent with the perception that Dutch livestock feed has high digestible energy values. VS_{EX} of the pigs were also within the range of the DE's, but the values were closer to the average DE than to the max.

Table 1: Excretion of Volatile Solids (VS_{EX}) in manure (kg/year per animal) of key categories of livestock in the Nethe lands based on the IPCC Tier 2 method using three levels of digestible energy (note a, b and c) within suggested ranges of DE as % of GE, the current method of estimation VS_{EX} (NL_current) and the revised method based on excretion of Total Ammoniacal Nitrogen and organic matter intake and digestibility (TAN+DOM)

			VS_{EX} (kg/year per animal)				
		Feed	IPCC (Tier 2)			Netherlands	
	Year	Intake[a]	DE_{AVG}[b]	DE_{max}[c]	DE_{min}[d]	NL_current	TAN+DOM
			Cattle				
Dairy Cows	1995	5540	2051	1525	2576	1518	1436
	2013	6628	2372	1764	2981	1664	1712
Rearing calves	1995	1459	523	389	657	330	393
<1 year	2013	1485	527	389	659	320	394
Rearing heifers	1995	2763	978	727	1129	759	746
>1 year	2013	2895	1010	751	1269	800	782
			Swine				
Sows+piglets	1995	1698	437	234	556	182	379
	2013	1994	376	254	632	128	319
Rearing gilts	1995	725	192	100	243	45	228
	2013	804	152	103	256	33	171
Fattening pigs	1995	741	174	102	238	75	158
	2013	745	126	94	256	47	110
			Poultry				
Broilers	1995	30.0	3.7	2.5	4.8	4.6	7.6
	2013	35.0	4.3	3.0	5.6	4.6	8.0
Rearing broiler breeders	1995	23.8	6.0	4.9	7.1	3.4	5.2
	2013	21.6	5.5	4.5	6.5	3.4	5.2
Broiler breeders	1995	60.5	14.4	11.7	17.0	8.6	12.5
	2013	60.1	14.5	11.8	17.2	8.6	12.2
Rearing laying hens	1995	16.2	4.1	3.3	4.8	2.7	3.9
	2013	17.8	4.5	3.7	5.3	2.7	4.0
Laying hens	1995	41.9	9.8	8.0	11.7	6.6	8.1
	2013	42.5	10.0	8.1	11.8	6.6	8.5

[a] Feed intake based on data of the Central Bureau of Statistics [8]; [b] Calculated based on the an average DE value, using GE, ME and ash values based on feed composition [7]; [c] VS_{EX} estimated using the upper value of DE as suggested by the IPCC [3]; [d] VS_{EX} estimated using the lower value of DE as suggested by the IPCC [3].

4. Conclusion and outlook

The revised Dutch methodology to estimate TAN extended with data of OM intake- and digestibility (TAN+DOM), provides a good and useful framework for estimation of VS in manure for methane emissions from manure management. It proves to be more accurate for the Dutch situation than the current Dutch method (NL_current), and because of the uncertainty concerning the DE, also than the IPCC Tier 2 method. However, further research is needed to validate the method and to assess the sensitivity of the estimation for variation in input data.

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References

[1] Coenen, P.W.H.G., et al., Greenhouse gas emissions in The Netherlands 1990-2012 : national inventory report 2014. 2014, Bilthoven: National Institute for Public Health and the Environment (RIVM).

[2] Den Boer, D.J., Reijneveld, J.A., Schröder, J.J., van Middelkoop, J.C., 2012. Mestsamenstelling in Adviesbasis Bemesting Grasland en Voedergewassen. Wageningen UR Livestock Research, Commissie Bemesting Grasland en Voedergewassen, Lelystad.

[3] IPCC, 2006. IPCC guidelines for national greenhouse gas inventories: Agriculture, forestry and other land use, in IPCC guidelines for national greenhouse gas inventories. 2006, Institute for Global Environmental Strategies (IGES): Hayama.

[4] CVB, 2011. Veevoedertabel 2011 [CVB Feed Table 2011]. Centraal Veevoederbureau, Lelystad, Nederland. http://www.pdv.nl/english/Voederwaardering/cvb_products/cvb_feedtable.php, last visited 6-7-2015.

[5] Velthof, G.L., C. van Bruggen, C.M. Groenestein, B.J. de Haan, M.W. Hoogeveen & J.F.M. Huijsmans (2012). A model for inventory of ammonia emissions from agriculture in the Netherlands. Atmospheric Environment 46 (2012) p. 248-255.

[6] Van Bruggen, C., A. Bannink, C.M. Groenestein, B.J. de Haan, J.F.M. Huijsmans,H.H. Luesink, S.M. van der Sluis, G.L. Velthof & J. Vonk, 2014. Emissies naar lucht uit de landbouw in 2012 : berekeningen van ammoniak, stikstofoxide, lachgas, methaan en fijn stof met het model NEMA. WOt technical report;3. 2014, Wageningen: Wettelijke Onderzoekstaken Natuur & Milieu.

[7] Bikker, P., van Krimpen, M.M., Remmelink, G.J., 2011. Stikstofverteerbaarheid in voeders voor landbouwhuisdieren : berekeningen voor de TAN-excretie. Wettelijke Onderzoekstaken Natuur & Milieu, Wageningen.

[8] CBS, 2013. Dierlijke mest en mineralen 2012 [Animal manure and minerals 2012]. Centraal Bureau voor de Statistiek, Heerlen / Den Haag Nederland.

TC-O_21 Animal delivery of a nitrification inhibitor via feeds to urine patches

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1. Objectives

In grazed pastures, urine patches are the main source of nitrate leaching and nitrous oxide loss to the environment. The nitrification inhibitor dicyandiamide (DCD) can reduce these losses. Our objective was to investigate the efficacy of feeding cows with DCD (mixed with three types of supplementary feeds) to achieve targeted delivery of DCD to urine patches only (as an alternative to broadcast application).

2. Methodology

The experiment was set up in a Latin Square design, where three groups of five non-lactating dairy cows were subject in rotation to one of three feeding treatments during three successive phases of 21 days. Each phase was divided between pre-conditioning (4 days), sampling (5 days) and resting periods (12 days). During pre-conditioning and sampling, groups were evening-fed daily one of three treatments (3 kg dry matter cow^{-1} of grass silage GS, maize silage MS or barley concentrate BC) that was manually mixed with DCD (30 g cow^{-1}). During resting periods, cows were fed with GS without DCD. In the sampling periods, groups were brought out daily to graze fresh pasture. A representative soil sample (31 cores / sample on average) was collected from each urine patch (n=292) after marking the edge, using a 1 × 1 m grid and a soil corer (10 cm deep) [1]. Soil samples were sieved, extracted with 2 M KCl and filtered. Some urine samples (n = 26) were also collected (and filtered) from the DCD-fed cows. Soil extracts and urine samples were analysed for DCD concentration by HPLC analysis [2]. Urine samples (unfiltered) were also analysed for total N (TN) concentration. DCD concentrations from urine patches were converted into equivalent DCD application rates [1]. An analysis of the Latin Square structure was carried out to examine the feed effect (0.05 significance level) on the log-transformed equivalent DCD application rate. The relationship between urinary TN and DCD concentrations was investigated by linear regression analysis.

3. Results and discussion

3.1 Relationship between DCD and TN excreted

DCD concentration in dairy cow urine displayed a wide range of concentrations with values ranging from 0.264 to 2.517 g L^{-1}. Likewise, urinary TN (corrected to exclude DCD-nitrogen) varied between low (1.544 g L^{-1}) and high concentrations (12.396 g L^{-1}). Both variables were significantly and positively correlated (R^2 = 38%, P = 0.0008) (Figure 1). A similar result was reported in a sheep study [3]. This result suggests that in grazed pastures, the highest DCD loading in urine patches should be found at the highest equivalent TN application rate, thereby ensuring adequate DCD in high N patches prone to greatest N losses.

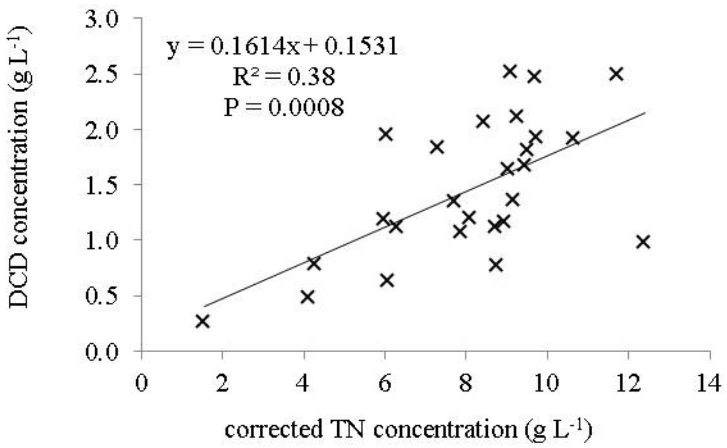

Figure 1: Scatterplot of TN (corrected to exclude DCD-nitrogen) and DCD concentrations (g L^{-1}) with linear regression line, measured in urine samples collected from three herds of five cows fed with DCD (30 g cow^{-1} d^{-1}) that was man ally mixed with 3 kg dry matter of either grass silage, maize silage or barley concentrate.

3.2 DCD excreted in urine patches

Following grazing, the equivalent DCD application rate for the whole experiment was very variable, with values ranging from 0.2 to 195 kg ha^{-1} (Figure 2).

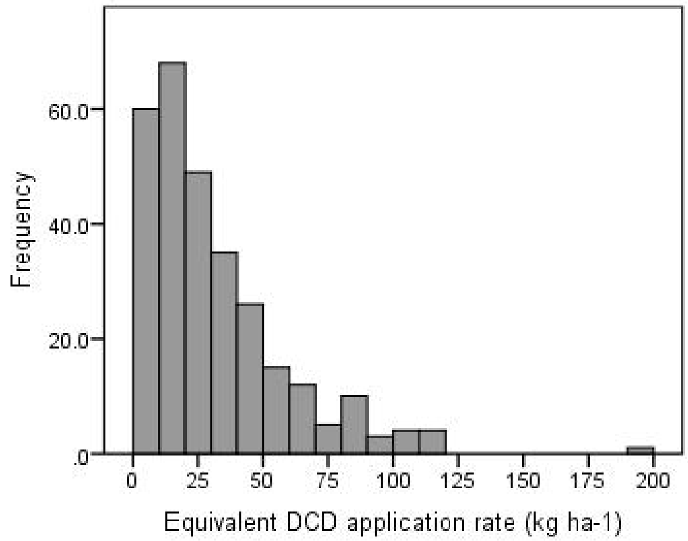

Figure 2: Histogram (n = 292) of the equivalent DCD application rates (kg ha^{-1}) in urine patches measured after three herds of five dairy cows were fed with DCD (30 g cow^{-1} d^{-1}) that was manually mixed with 3 kg dry matter of either grass silage, maize silage or barley concentrate.

The median equivalent DCD application rates in the three feeding treatments varied between 18.8 and 34.6 kg ha^{-1} (Figure 3), i.e. well above the 10 kg DCD ha^{-1} rate that had been used for DCD broadcasting in New Zealand. There was no significant effect (P = 0.4) of treatment on the log-transformed equivalent DCD application rate. This experiment shows that mixing DCD with different feeds should be similarly efficient at delivering high DCD rates to urine patches where high TN levels can lead to large N losses.

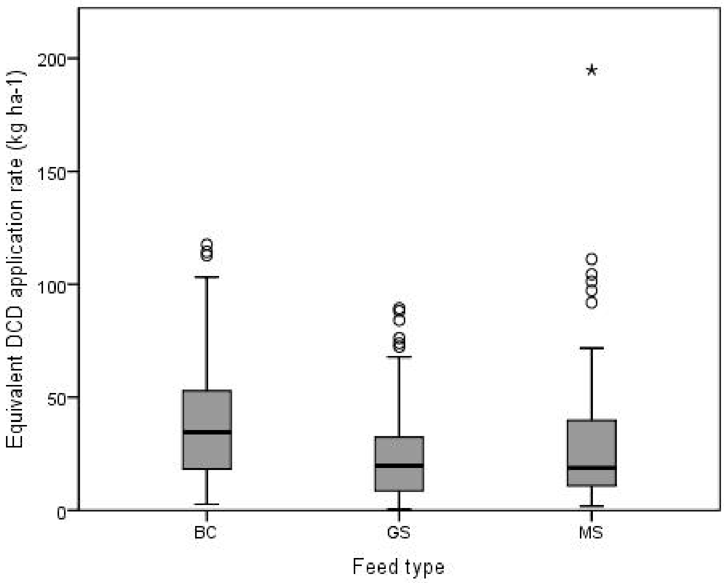

Figure 3: Boxplot of the equivalent DCD application rates (kg ha^{-1}) in urine patches from three herds of five dry cows fed with DCD (30 g cow^{-1} d^{-1}) that was manually mixed with 3 kg dry matter of either grass silage (GS, n = 101), maize silage (MS, n =101) or barley concentrate (BC, n = 90) (the three horizontal lines of the box are the 25th, 50th and 75th percentiles; lower and upper whiskers connect the smallest and largest values that are not outliers; circles are outliers (i.e. > 1.5 box-length above the box); stars are extreme cases (i.e. > 3 box-lengths above the box)).

4. Conclusion and outlook

This experiment showed that DCD and TN concentrations in excreted urine from DCD-fed cows were positively and significantly correlated. Incorporating DCD with supplementary feeds should therefore be effective at delivering high DCD rates to urine patches where high TN levels can lead to large N losses. No significant feed effect was detected on the equivalent DCD application rate measured in urine patches. This result suggests that any of the three feeds tested (grass silage, maize silage and barley concentrate) would be similarly effective at quantitatively delivering DCD to urine patches. DCD could easily be incorporated into animal feeds and would reduce costs due to its targeted delivery in urine patches (compared with DCD blanket application) as well as decreasing N losses to air and water.

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Disclaimer

DCD has been withdrawn from use on farms in New Zealand and its use is now restricted to research. Protocols to ensure that DCD use in research does not enter the food chain are supported by soil and plant testing for DCD residues.

References

[1] Welten, B.G.; Ledgard, S.F.; Schipper, L.A.; Waller, J.E.; Kear, M.J.; Dexter, M.M. (2013): Effects of prolonged oral administration of dicyandiamide to dairy heifers on excretion in urine and efficacy in soil, Agriculture, Ecosystems and Environment 173, 28–36

[2] Turowski, M.; Deshmukh, B. (2004): Direct chromatographic method for determination of hydrogen cyanamide and dicyandiamide in aqueous solutions, Analytical Letters 9, 1981–1989

[3] Ledgard, S.F.; Menneer, J.C.; Dexter, M.M.; Kear, M.J.; Lindsey, S.; Peters, J.S.; Pacheco, D. (2008): A novel concept to reduce nitrogen losses from grazed pastures by administering soil nitrogen process inhibitors to ruminant animals: A study with sheep, Agriculture, Ecosystems and Environment 125, 148–158

TC-O_22 Development of system to reduce ammonia emission and leaching of nitrate from slurry application

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1. Background

With the increasing industrialization of agriculture and the trend towards ”clusters” of animal production in many countries, the concentration and animal density, has become a problem to the surrounding environment. So far, this has resulted in the EU nitrate directive and the EU National Emission Ceiling, limiting the amount- and emission of nitrogen. In some cluster areas such as Denmark and The Netherlands, national legislation is more advanced and this has resulted in an accelerated implementation of the directives compared with the rest of EU. This has led to development of new environmental technologies such as the SyreN system discussed in this article.

2. Objectives

The national legislation in both above-mentioned countries, specifically addresses soil injection of slurry. Injection of slurry significantly reduces the ammonia emission from slurry, but it is also an expensive method compared to surface application and has drawbacks such as:

- Damage to growing crops
- Operational problems in heavy- and/or undulated grounds, and under dry soil conditions
- Risqué of pollution swapping in the form NH_3 and N_2O
- Increased traffic on fields and compaction of soils

Thus, a project in cooperation with Aarhus University was initiated to prototype a system using acid to control the ammonia emissions without the drawbacks of injection.

3. Results

The SyreN system uses a known technique – lowering the pH of the slurry, known as acidification [1] – to reduce the ammonia emission. Concentrated sulphuric acid is used to lower the pH of the slurry during application in the field. The novelty of using acidification in the field is a significant reduction in use of acid compared with storage or barn acidification. This effect is a result of treating only the liquid fraction of the slurry, avoiding consumption of acid by the dry matter. In addition, treatment in the field is flexible and can be adjusted to changing weather conditions, which greatly influences emissions. The dose rate of sulphuric acid can be adjusted to crop need for sulphate fertilizer with good economic results.

A special feature of the SyreN system is using the transport container for sulphuric acid (IBC - Intermediate Bulk Carrier) as an acid tank with the system. This method creates a safe to use system, as there is no refilling / handling of the acid at the user end but only a dry coupler must be used. In addition, the IBC is standard approved for public road transportation under the ADR rules. The system is positioned in the front hitch of the tractor, which facilitates loading of the IBC tank, and offsets weight from the slurry tanker to the tractor.

Figure 1: Loading system

3.1 Economy

The trial results from 4 years of tests in Denmark and Germany has revealed that the amount of nitrogen from the reduction in ammonia emission, in combination with a unique way of applying needed sulphur as a fertilizer, is profitable. That means that there is a market for the technology with application of all slurries regardless of environmental concerns or not [2]. In the period from 2010 to 2014, + 50 trials with bandspreading have shown an average yield increase of 4% in grass and 2,4 kg/ha-1 in wheat. The economic effect of acidification during application is influenced by the application technology with which it is used. Broadcasting of slurry is known to have a very high emission rate and the use of the system in combination with this technology has the highest effect with an average estimated +30 kg nitrogen pr. Ha. [7]. The variation ranges from +10 kg to +60 kg nitrogen / Ha. The combination of bandspreading and acidification is well documented with a VERA test protocol for emissions from fertilizers [3]. It reduces the ammonia emission between 40 – 60%, equivalent of +15 kg nitrogen pr. Ha, based on slurry pH of 6.4. The combination of acidification with open slot injection reduces the effect to an estimated +5 kg pr. Ha. A large number of trials by Aarhus University and Leuphana Universität have compared the effect of open slot injection system to acidification and has largely concluded that the two systems are similar concerning emission and yield increase. Authors have also tried to determine the ideal pH level of the slurry, where the conclusion is that the economical optimal level is pH 6.4, with pH 6.0 having the better emission reduction. Acidification also makes more P available for plant uptake [1]. The wider working width using acidification compared to injection as well as increased capacity of slurry application makes acidification more profitable than injection.

In addition, different additives can be added to the slurry from the SyreN additive tank system. Currently iron sulfate is used for reduction of odor from slurry. Manganese sulfate- or nitrate in combination with a low pH in the slurry is used in areas with manganese deficiency as well as all dissolvable micro nutrients can be added to the slurry. Finally, nitrogen inhibitors are used to reduce the nitrification process of the ammonium from slurry. This is especially recommended for maize, beets or all autumn applications, where an increased risk of leaching is present. A research study by SEGES, Denmark [5] – shows an estimated reduction in leaching from nitrogen with an average 6 kg ha-1. from maize and spring crops.

Increasing the use efficiency of organic fertilizer has a significant GHG abating effect. A recent LCA study on acidification during field application revealed a reduction of 200 kg CO_2 eq pr. Ha [6] under Danish conditions using SyreN with bandspreading technology.

The economics of using acidification with SyreN system has a long range of variable parameters. In order to accommodate user requirements for estimates of profitability, two sets of software has been developed – Alfam model and SyreN Estimater. The core of the software is the Alfam model developed by Aarhus University [7] to estimate the emission from slurry under a given set of climatic conditions in combination with slurry types and application techniques. The software is free to use available at www.biocover.dk. The SyreN Estimater calculations are based on a fixed price pr. m^3 of treated slurry, as most users are custom applicators. To recommend a charge pr. m^3, the Danish custom applicator union [8] has conducted a study among its members (102 units operational in Denmark pr. 2015). As expected, the cost is highly variable depending on number of m^3 slurry offered to the system, where treatment of 20.000 m^3 recommends a price of 0.50 Euro pr. m^3, excluding variable cost for sulphuric acid. For an individual farmer / user, the cost is less, as they typically do not include profits for use of system in their calculations, but rely on the offset of nitrogen cost and / or yield increase to make use of the system in a profitable manner. A farm estimate based on 1500 Ha swine / dairy farm show a farm income of 32.000 Euro and 19.000 Euro respectively based on application by a custom applicator [9].

3.2 Sulphuric acid as fertilizer

Sulphur deposition in Denmark has dropped from over 300 kg/ha in the early 80ties to just 3.7 kg/ha in 2013 [10]. This has introduced a growing need for sulphate as fertilizer. From the now 5 years of operation, we have seen a huge variation in the use of the system depending on local needs - the higher the crop need for sulphur, the better the economics. It is not unusual to see yield increases of ½ ton in rape, as it is hard to find fertilizers that satisfy the high consumption of sulfate. A common way to address the need for sulphur in rape, is to replace expensive fertilizers such as ammonium sulfate NS 21-24 with a combination of sulphuric acid and conventional fertilizers such as Nitrostar NS 28-5. This reduces the purchase price more than 25 Euro / ha - and enables increased use of sulfate to reach the recommended target rate.
Sulphur deficiency is also common with grass. Many farmers are using slurry as the only type of fertilizer- or just a low NS supplement is common, resulting in sulphur deficiencies. This kind of problem is automatically solved with SyreN system as the level of sulphate as fertilizer is often the same level for the targeted pH 6.4 used by the system, where 1.5 liter sulphuric acid / Ha is average requirement:

	S-need, kg pr. ha	Typical amount of slurry, ton pr. ha	Needed kg S pr. ton	Liter H_2SO_4 pr. ton slurry
Winter wheat, clay soil	15	30	0,5	0.9
Spring barley, sandy soil	10	30	0,3	0.6
Winter rape, clay soil	35	30	1,2	2.1
Silage grass, irrigated sandy soil	30	40	0,8	1.3

3.3 Economy in adjustment of N:P rate

The latest addition to the SyreN system (2013) is the ability to inject anhydrous ammonia into the slurry during filling of the slurry tanker. With the extreme pumping capacity available, it is possible to distribute the gas absorption to liquid over a very large area, avoiding overheating and potential problems with heat- and pressure building. As the slurry tanker is also a closed container system, 100% absorption of the ammonia is a reality.

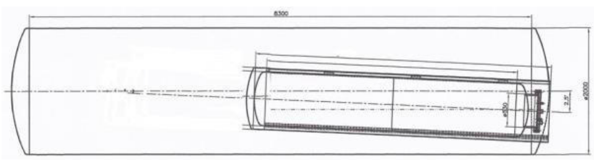

Figure *2*: Ammonia pressure tank integrated in the slurry tanker

The combined system, now under the name of SyreN+, adjusts the N:P ratio through addition of anhydrous ammonia during the filling of the slurry tanker. The added ammonia is changed to ammonium in the slurry, resulting in a sharp increase in pH value. Only 1 kg / m^3 is needed to increase the pH above 8.0 [11]. During application, sulphuric acid is added at a rate of 1.6 litter / m^3 acid to 1 kg ammonia to offset the pH value increase. This system reduces traffic on the field with one passing.

A major economic benefit is avoiding over application of phosphorus by being able to adjust the N:P ration to a recommended level for crop uptake for both N and P. The reduced use of phosphorus is both a major economic and environmental benefit. A SyreN Estimater example [9] on

500 Ha rape, showed a net increase of 6.580 Euro using the SyreN+ anhydrous ammonia system compared to using just acid and conventional fertilizer.

Benefits with SyreN+:

- Adjustment of nitrogen contends in slurry – reduced traffic on field
- Surface application or Soil injection
- Reduced purchase prize of fertilizer
- All N fertilizer as ammonium – reduced leaching of nitrogen
- Increased effect from nitrogen inhibitor
- Liquid surface application of nutrients – fast plant response
- Ideal system for slurry application in growing crops

The effect is creating a balanced NPKS nutrient value in slurry, an up to 80% reduction in ammonia emission and binding the nitrogen as ammonium in the soil, reducing leaching of nitrogen to aquatic environment. Further, a reduction in CO_2 emission through reduced traffic on fields and in the fertiliser value chain through using industrial raw materials direct at the end user level. The SyreN Estimater software also allows the user to quantify amounts of nutrients to be used and to identify the economy in using the system to optimise the effectiveness of organic fertilisers.

4. Future

The ability to increase and decrease the pH of slurry based fertilizers, gives a unique potential to adjust the nutrient values in slurry. SyreN Crystal is a current project that uses precipitation to reduce / adjust the amount of phosphorus in slurry – whereby excess phosphorus from high animal intensive areas may be economically redistributed to areas of low animal intensity. The slurry tanker is used as a tank for precipitation of phosphorus as MAP or struvite. With a slurry tanker and the SyreN system, there are no additional abilities needed. $MgCl_2$ is injected into the slurry in combination with increasing the pH with anhydrous ammonia. This causes the phosphorus to precipitate. Removing the precipitate material enables adjustment of the N:P ratio. The precipitated material is gathered in a separate tank within the slurry tanker. When enough struvite material is accumulated, the tank is filled with roughly a 1:1:1 ratio of sulphuric acid, struvite and water. This causes the crystals to dissolve and the sulphuric acid is altered to phosphorus acid. Following, the anhydrous ammonia system is use to increase the nitrogen level in the now liquid fertilizer. Ideally, the pH is raised to a 4-6 level and increasing the NPKS value to 17-4-0-9. The liquid is filled back into the transport IBC´s and it is now a commercial grade starter fertilizer for maize based on organic P. This process has a potential to reduce the consumption of phosphorus in agriculture with 20%. As there are no capital cost for precipitation equipment, it is expected to be a profitable solution.

References

[1] Acidification of animal slurry a review David Fangueiro a, *, Maibritt Hjorth b, Fabrizio Gioelli
[2] Trail work from Danish Landsforsøg Annette Vestergaard.
[3] VERA Verification Statement 001 - SyreN, 02-10 2012 for WEB
[4] Reduction of ammonia losses after spreading of cattle slurry to grassland by acidification and injection. Achim Seidel[1*], Andreas Pacholski[2], Tavs Nyord[3], Annette Vestergaard[4], Henning Kage[1]
[5] Use of nitrogen inhibitor with acidification – Leif Knudsen, SEGES Denmark
[6] Estimation of the Green House Gas (GHG) emission from the Manure application system SyreN+, Steen Gyldenkærne
[7] Ammonia volatilization from field-applied animal slurry—the ALFAM model, H.T. Søgaard, S.G. Sommer, N.J.
[8] Cost calculation for SyreN system pr. m3. DM&E Denmark
[9] Examples of SyreN Estimater software calculations of farm income. Morten Toft, BioCover A/S
[10] DEC (Danish center for environment and energy) repport 2014
[11] Notat om SyreN+. Author Tavs Nyord, Aarhus university 19.03.2013. Ref. TANY

References available on www.biocover.dk - articles

TC-O_23 GHG emissions from temperate paddy fields under different straw and water managements

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1. Objectives

Straw management is a key factor affecting sustainability of rice cropping systems. Furthermore, rice straw is a potential source for energy production. Straw and water management techniques strongly impact greenhouse gas (GHG) emissions from paddy fields through changes of carbon availability and soil redox conditions that regulate the production and release of both methane (CH_4) and nitrous oxide (N_2O) [1]. Three complementary experiments were established to evaluate the implications of alternative straw and water management techniques on CH_4 and N_2O emissions from paddy cultivation during cropping and intercropping periods.

2. Methodology

Methane and N_2O emissions from temperate rice-cropped soil were measured applying the closed-chamber technique [2] on a series of three experiments established within the Italian rice area (Pavia, Vercelli, and Crescentino, NW Italy) in the period 2012-2014. Three different crop residue management techniques (autumn or spring straw incorporation into soil, straw removal) and three water management practices (continuous flooding, dry seeding and delayed flooding, rotational irrigation) were evaluated. Main details concerning the set-up of the three experiments are provided in Table 1.

Table 1: Main characteristics of the three experiments and short description of treatments.

Experiment	Soil	Type	Duration	Straw Management	Water Management	Fertilization
Castello d'Agogna	Silt-loam, average content of C and N	Field	2 whole years	SPR	Std_FLD	MIN
				SPR	Late_FLD	MIN
				SPR	Rot_IRR	MIN
Vercelli	Silt-loam, average content of C and N	Field	1 cropping season	AUT	Std_FLD	MIN
				SPR	Std_FLD	MIN
				SPR	Late_FLD	MIN
				REM	Std_FLD	MIN
Crescentino	Silt-loam, high content of C and N	Pot	1 cropping season	SPR	Std_FLD	MIN
				REM	Std_FLD	MIN
				SPR	Std_FLD	DG+MIN
				REM	Std_FLD	DG+MIN
				SPR	Std_FLD	DG_SF+MIN
				REM	Std_FLD	DG_SF+MIN

SPR- spring incorporation; AUT- autumn incorporation; REM- removal; Std_FLD- continuous flooding; Late_FLD- dry seeding and delayed flooding; Rot_IRR- rotational irrigation; MIN- mineral; DG- digestate (from maize silage, triticale silage and 5% rice straw); DG_SF- solid fraction of digestate.

3. Results and discussion

3.1 Castello d'Agogna

In Figure 1 yearly cumulative fluxes of CH_4 and N_2O from Castello D'Agogna Experiment, as average of values of 2012 and 2013 campaigns, are shown.

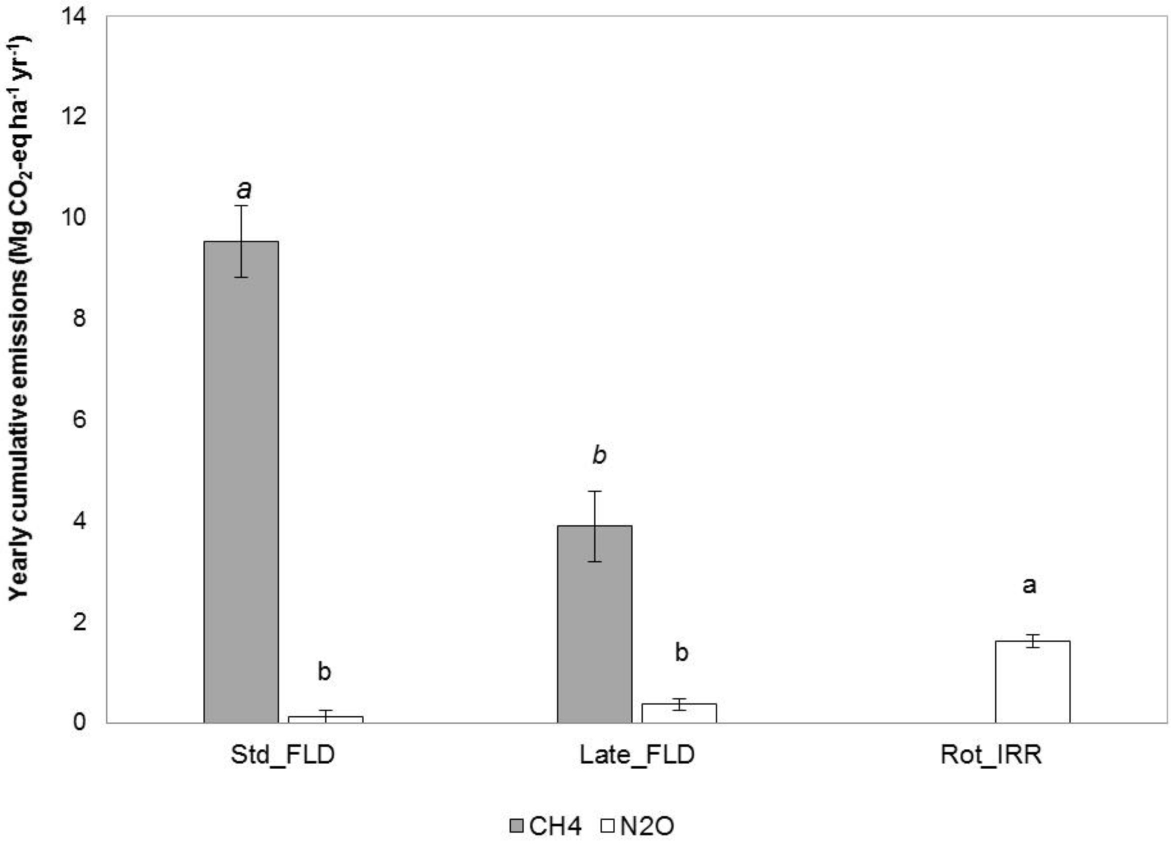

Figure 1: Yearly cumulative emissions of CH_4 and N_2O from Castello D'Agogna Experiment (average 2012-2013).

Reducing conditions, due to flooding, enhanced CH_4 fluxes, leading to the highest emissions in treatments seeded in water and flooded for the most part of cropping cycle [3]. Delaying flooding by approximately one month, not only delayed the beginning of CH_4 emissions but also, significantly reduced their intensity when present, since incorporated straw was subjected to a period of aerobic decomposition and mineralization before flooding. Rotational irrigation on one side prevented any CH_4 loss as oxic conditions prevailed during the whole cropping system, but on the other side induced high N_2O emissions, otherwise limited during flooding due to an inhibition of nitrification and to complete denitrification to N_2. Combining fluxes in the GWP indicator evidenced that CH_4 rather than N_2O is the main contributor to GWP, indicating that continuous flooding had a greater impact on climate change with respect to the alternative water management options.

3.2 Vercelli

Under continuous flooding conditions, autumn incorporation of straw reduced CH_4 emissions with respect to spring incorporation. This is due to the fact that crop residue incorporated in autumn were more degraded (aerobically) with respect to those incorporated in spring at the time of field flooding. The level of emissions produced by treatment AUT was equal to that recorded for REM, showing that early incorporation of straw or its removal was not significantly different for CH_4 budget. Methane emissions being equal, both management options disclose other potentially opposite environmentally friendly properties. In AUT, in fact, incorporated straw can be effective in maintaining proper levels of SOC, while in REM, removed straw could be addressed to a virtuous use of its C for energy production.

Nitrous oxide was produced for all treatments during drainage events, both for favorable redox conditions and for mineral N availability, since all drainage events corresponded to a fertilization

[4]. Yearly cumulative emissions were the highest when straw was removed from field, while were the lowest for autumn incorporation. This behavior suggests straw influences C and N availability that in turn plays a pivotal role in controlling N_2O emissions: on one hand, residue provides a labile C substrate for microbial utilization that can induce N_2O emissions when associated to N fertilization (as happened for SPR-Std_FLD treatment, with respect to AUT); on the other hand soil incorporation of straw induces N immobilization partially preventing N_2O emissions [5].

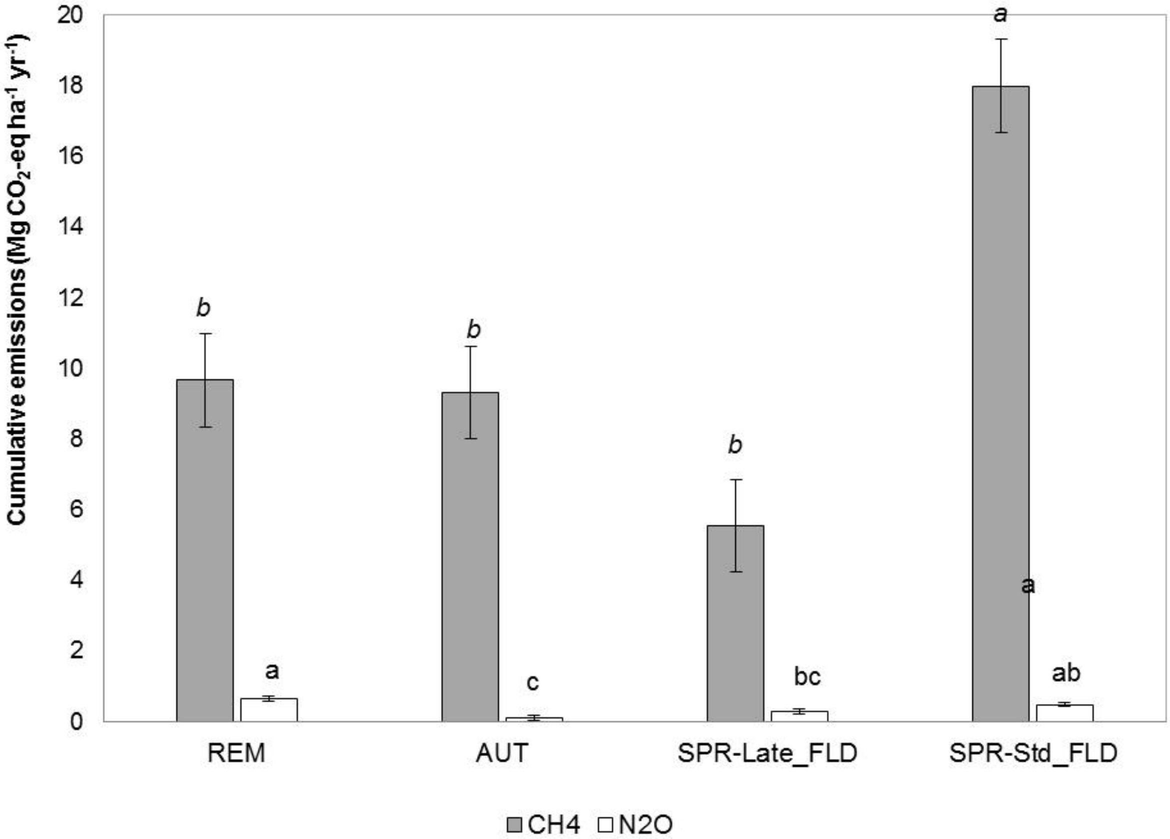

Figure 2: Cumulative emissions of CH_4 and N_2O from Vercelli Experiment (campaign 2014).

3.3 Crescentino

Results showed a significant effect of straw incorporation on CH_4 emissions. When residues were incorporated emission were significantly higher than the corresponding treatment with straw removal, except when only mineral fertilizer was used. The highest emissions occurred in the treatment receiving both straw and solid fraction of digestate, meaning thatCH_4 was driven by the amount of added C [6].

Since the experiment was conducted following a strict protocol and working in pot conditions allowed an optimal control of soil management, flooding was established immediately after fertilization. This resulted in the absence of N_2O emissions during the whole cycle. Combining this information with that arisen from the other two experiments, it appears that correct water management which involves flooding immediately after fertilization can be effective in strongly limiting N_2O emissions.

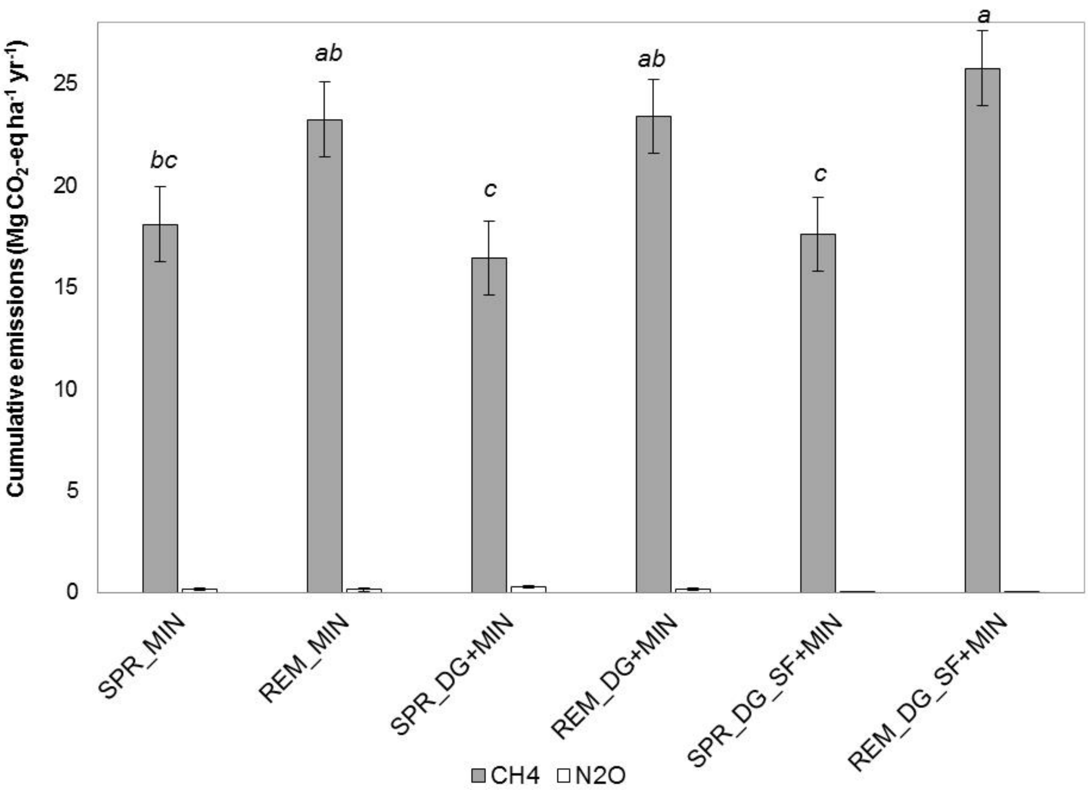

Figure 3: Cumulative emissions of CH_4 and N_2O from Crescentino Experiment (campaign 2014).

4. Conclusion and outlook

The best straw practice for reducing GHG losses was the spring incorporation associated with dry seeding and delayed flooding. Straw removal was effective in reducing field emissions of CH_4. Continuous flooding, although preventing N_2O fluxes, was the water management option showing the worst performance in terms of GHG production, because of enhanced CH_4 emissions.

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References

[1] Lu Y.; Wassmann R.; Neue H.U.; Huang C (2000): Dynamics of Dissolved Organic Carbon and Methane Emissions in a Flooded Rice Soil, Soil Science Society of America Journal, 2011-2017.

[2] Hutchinson, G.L.; Livingston. G.P (1993): Use of chamber systems to measure trace gas fluxes. In: L. Harper, et al., editors, Agricultural ecosystem effects on trace gases and global climate change. ASA Spec. Publ. 55. ASA, CSSA, SSSA, Madison, WI. p. 63–78.

[3] Pittelkow, C.M.; Assa, Y; Burger, M.; Mutters, R.G.; Greer, C.A.; Espino, L.A.; Hill, J.E.; Horwath, W.R.; van Kessel, C.; Linquist, B.A. (2014): Nitrogen management and methane emissions in direct-seeded rice systems, Agronomy Journal, 968-980.

[4] Kudo Y.; Noborio K.; Shimoozono N.; Kuriha R. (2011): The effective water management practice for mitigating greenhouse gas emissions and maintaining rice yield in central Japan, Agriculture, Ecosystems & Environment, 77-85.

[5] Said-Pullicino, D. ; Cucu, M.A.; Sodano, M.; Birk, J.J.; Glaser, B.; Celi, L. (2014): Nitrogen immobilization in paddy soils as affected by redox conditions and rice straw incorporation, Geoderma, 44-53.

[6] Wang J.; Jia J.; Xiong Z.; Khalil M.A.K.; Xing G. (2011): Water regime–nitrogen fertilizer–straw incorporation interaction: field study on nitrous oxide emissions from a rice agro ecosystem in Nanjing, China; Agriculture, Ecosystems & Environment, 437 - 460.

TC-O_24 Life cycle assessment of a combined manure system optimized for phosphorous utilization

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1. Objectives

As part of an ongoing effort in identifying improved systems for future management of manure, an environmentally-optimised system was designed with focus on increased utilisation of phosphorus (P) in manure. The optimised system comprises a combination of techniques: In-house source-separation, biogas production, separation of the digestate and thereby obtaining a phosphorus-rich organic fertilizer, which can be transported to agricultural areas with phosphorous deficiency. The objective was to evaluate the environmental consequences of the optimised system [1].

2. Methodology

The evaluation of the environmental impacts of the manure management system was based on the consequential Life Cycle Assessment (LCA) approach, explained in [2]. The environmental aspects have been evaluated following the manure chain:

- Source-separation of manure from fattening pigs, in-house, directly after excretion
- Storage of the liquid fraction from the source-separation at the farm and applied to fields locally for fertilising
- Transport of the solid fraction from the source-separation to biogas plants
- Biogas production, based on a mixture of the solid fraction from the source-separation and raw pig slurry in order to obtain the optimal % dry matter input for biogas production
- Separation of the digested pig manure after the biogas plant
- Transport of the solid fraction (digested manure) to agricultural areas with P deficiency
- The biogas is used for energy production, thus replacing and avoiding the energy production based on fossil fuels
- As the manure is applied to fields, the phosphorous content replaces P in mineral fertiliser, and the production of the "avoided amount of P mineral fertiliser" is subtracted from the system. The amount of Nitrogen (N) and Potassium (K) in the manure, replacing mineral fertilisers, is also included.

The optimised system was compared to the typical management of raw pig slurry in Denmark today (in-house storage, outdoor storage and application to field), the "reference system". Flow charts for the two systems are shown in figure 1 and 2 below.

The LCA was carried out for manure from fattening pigs. Data represents Danish conditions, e.g. manure composition and management, crop cultivation, legislation, separation and biogas technology.

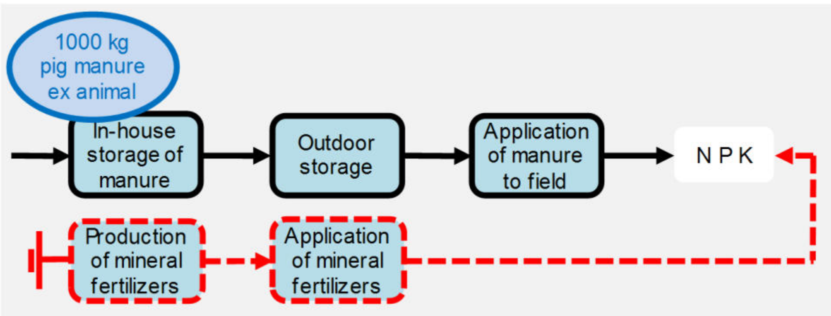

Figure 1: System Boundaries for the reference system (the typical management of pig manure in Denmark today). Dotted lines indicate avoided processes.

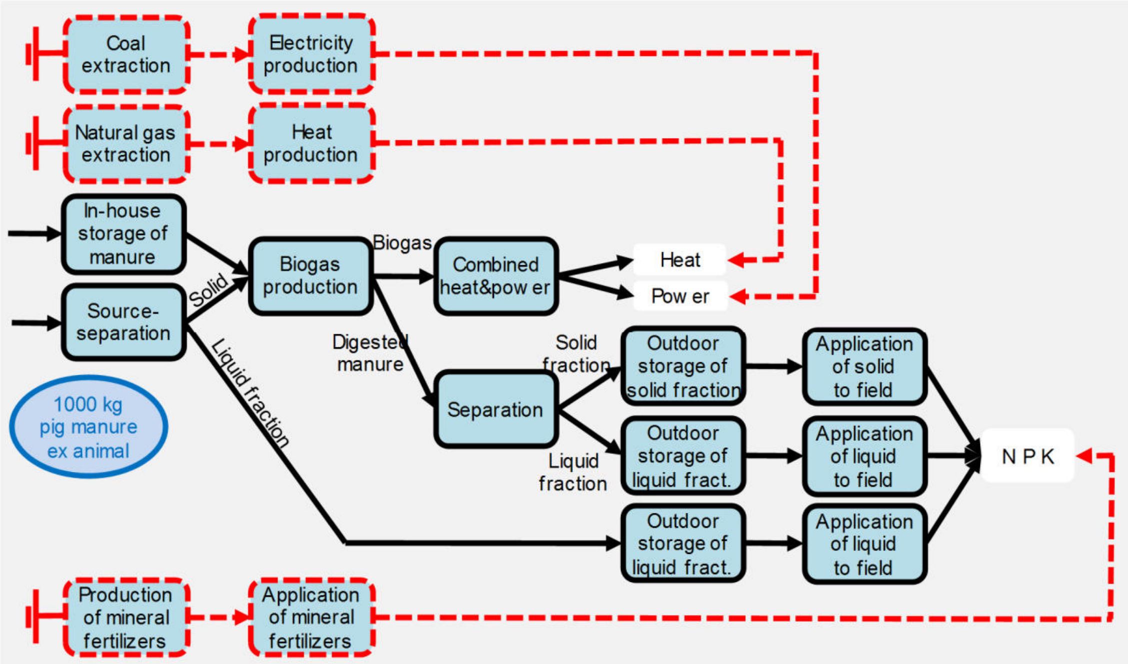

Figure 2: System Boundaries for the optimised manure management system. Dotted lines indicate avoided processes.

3. Results and discussion

The results of the LCA of the optimised manure management system compared to the reference system (the "usual management of raw pig slurry in Denmark today") showed that:

- The optimised system reduces greenhouse gases by 25-50% (50% when biogas substitutes fossil fuels; lower in the future, where energy production will be based on a mixture of fossil fuels and renewable energy sources).
- The optimised system cuts ammonia emissions by 23-30%.
- The separation of digested manure after the biogas plant leads to important savings of mineral phosphorous fertiliser. In the optimised system, the phosphorous can be transported to agricultural areas with phosphorous deficit.
- The energy consumption for transporting the phosphorous-rich solid fraction was insignificant for the overall results, implying that transportation distance would not hinder the environmental benefits of the overall system (100-300 km).
- There was no significant difference regarding nitrate leaching.

The results are discussed below, all based on the results in [1].

3.1 Global warming emissions

The optimised manure management system reduces the contributions to global warming compared to the reference system:

- One of the main reasons is that the produced heat and power based on biogas substitutes heat based on natural gas and power based on coal, and these have been subtracted from the system. Sensitivity analysis for the replaced energy production shows that for future energy systems, where heat and power is based on non-fossil sources (wind and solar power), the reduction is much lower. Although the benefits regarding global warming emissions will be lower in the future, the optimised manure management system will still be an advantage, as biogas production is necessary as energy storage in a future energy scenario [3].
- The CH_4 emissions during the outdoor storage are lower for the storage of the solid fraction plus storage of the liquid fraction compared to storage of the manure in the reference system, as the CH_4 emission factor per kg VS was lower for storage of the solid fraction than for the raw non-separated manure, and as it is assumed that the emission factor for the liquid fraction is identical to the emission factor for the non-separated manure. This conclusion is only true if the solid fraction from the digested manure is covered during storage. There is a significant uncertainty related to the emission factors for digested manure (separated as well as non-separated) due to lack of investigations in this field. Sensitivity analysis shows that the emis-

sions from the digested manure is crucial for the overall results; insufficient management of the solid fraction could lead to increasing the CH_4 and CO_2 emissions to a level where the optimised manure management will have the same overall level as the reference system.

- The energy consumption for the separation of the digested manure is insignificant for the overall contributions to global warming from the system.
- There are significant uncertainties regarding losses of CH_4 from the biogas plants. Losses corresponding to 1% have been assumed for the optimised manure management system. Losses up to 5% do not change the overall conclusions. If, however, the losses correspond to 10%, there is no significant difference between the greenhouse gas emissions from the optimised manure management system compared to the reference manure system.

3.2 Ammonia emissions and nitrate leaching

The optimised manure management system has lower NH_3 emissions than the reference system. The separation of the digested pig manure leads to increased NH_3 emissions compared to non-separated digested pig manure, caused by increased NH_3 emissions from storage of the solid fraction. Nevertheless, this is counterweighted by reduced NH_3 emissions after field application, as the separation leads to reduced NH_3 emissions from the liquid fraction during and after field application; due to the low content of dry matter, the liquid fraction infiltrates very fast in the soil, and hence, less ammonia is likely to volatilize as compared to non-separated slurry [4,5].

Separation of the digested pig manure after the biogas plant does not change the contributions to nitrate leaching significantly [1].

3.3 Phosphorous; resource consumption and aquatic eutrophication

In Denmark, the livestock production is very unevenly distributed. In the Western parts of Denmark, there is a high density of livestock production, resulting in a surplus of nutrients (relative to crop demands), and in contrast to this, there is a nutrient deficit at the farm areas in the Eastern parts of Denmark. Transport of the nutrient containing manure is costly, and separation might be a solution for this, reducing the transported volume.

The flows of phosphorous in the reference system and the optimised system are illustrated in figure 3 and 4 below. In the both systems, P is distributed to the agricultural areas in excess amounts. This is due to Danish law; In Denmark, the amount of slurry is applied according to the slurry content of N, regardless of the content of P. Accordingly, excess amounts of P is distributed to soils in farm areas with a high density of livestock production.

The purpose of separating the digested pig manure after the biogas plant in the optimised manure management system is to maximise the utilization of the phosphorous content of the manure. By separating, the phosphorous rich solid fraction can be brought to fields to agricultural areas with P deficit. Hence, when analysing the P content of the slurry, the slurry can be applied in amount corresponding to crop needs, defined as the fertilizer guidelines by Danish authorities. The amount of P distributed in excess amounts is significantly lower in the optimised system (30% of the P is excess in the reference system; this is reduced to 13% in the optimised system).

The separation type has been carefully selected in order to distribute maximum amounts of P to the solid fraction [1]. When implementing a system for optimising utilisation for phosphorous it is essential to identify the optimal separation method, recovering as much P as possible in the solid fraction, as there is significant difference between the numerous separation methods regarding the separation efficiency for P [6].

The application of manure to fields replaces mineral fertilisers (that is, for the amounts of P not applied in excess amounts), and the "avoided production of mineral fertilisers" is subtracted from the system. In the reference system, all P is applied to local agricultural areas; however, as P is applied in excess amounts, only part of the P actually replaces mineral P fertiliser. The production of 0.85 kg mineral P fertiliser is avoided in the reference system whereas 1.05 kg mineral P fertiliser is avoided in the optimised system (per 1000 kg manure ex animal). If optimising the separation after the biogas plant, it will be possible to avoid P being applied in excess amounts (in the model, calculations are based on "average technology", however, the separation efficiency is likely to improve in the future).

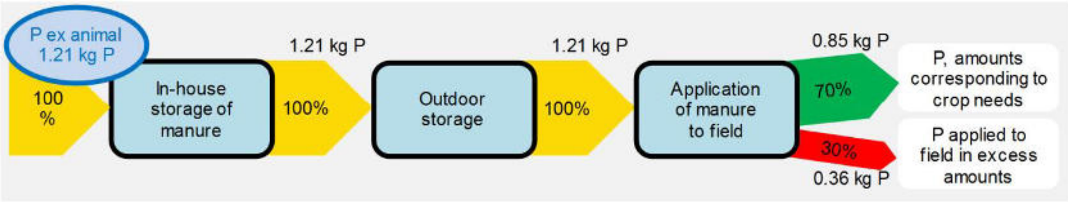

Figure 3: Flows of phosphorous in the reference system.

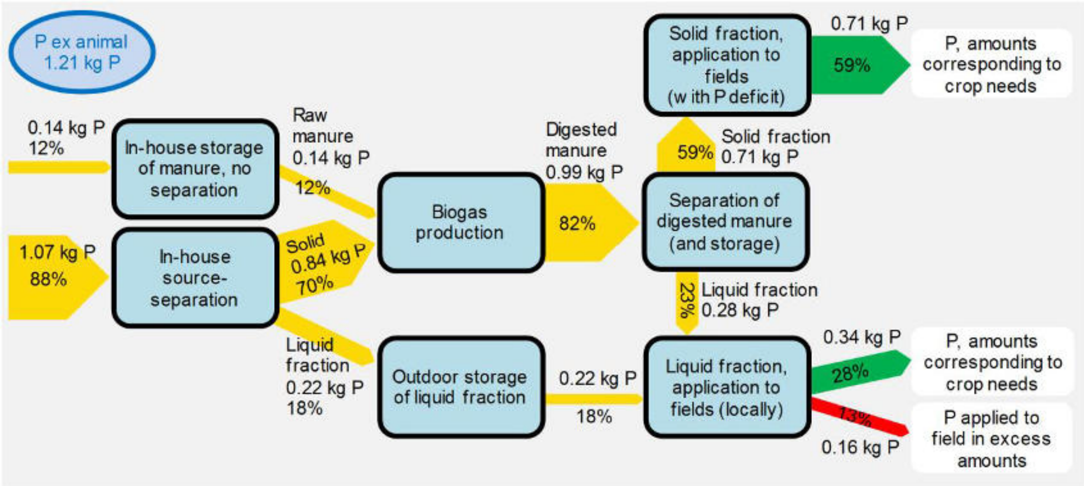

Figure 4: Flows of phosphorous in the optimised system.

Precise models on P losses from soil and long term consequences of application of P containing manure to fields are lacking. A simple model was used for evaluating the contributions to the impact category “Aquatic Phosphorous Eutrophication”, assuming that P losses correspond to 5% of the surplus P (i.e. P applied via manure minus plant uptake). More complicated models reflects that P losses are mainly corresponding to the P content of the soil (rather than the P applied in manure), and when using these, there is no significant reduction of the contributions to the environmental impact category “Aquatic Phosphorous Eutrophication”. Long term models are needed.

4. Conclusion

Introducing an optimised system for manure management in Denmark, comprising of a combination of techniques (source-separation, biogas production followed by separation of the digestate) could lead to significant reduction of the consumption of mineral P fertilisers in Denmark and reduce the total greenhouse gases as well as ammonia emissions from the management of pig manure. One key limitation was to assess the fate of phosphorus at the field, due to the lack of precise models on phosphorous losses from agricultural soils and on the long-term consequences of application of various amounts of organic phosphorous to fields.

References

[1] Wesnæs, M., Hamelin, L. & Wenzel, H. (2013): Life Cycle Inventory & Assessment Report: Separation of Digested Fattening Pig Slurry for Optimal P Concentration, Denmark. Syddansk Universitet. Institut for kemi-, bio- og miljøteknologi, Odense. Report, public available.

[2] Hamelin, L., Naroznova, I., & Wenzel, H. (2014): Environmental consequences of different carbon alternatives for increased manure-based biogas. Applied Energy, 114, 774-782.

[3] Hamelin L (2013): Carbon Management and environmental consequences of agricultural biomass in a Danish Renewable Energy Strategy. PhD thesis. Department of Chemical Engineering, Biotechnology and Environmental Technology. Faculty of Engineering. Odense, Denmark

[4] Hansen, M. N., Sommer, S. G., Hutchings, N., & Sørensen, P. (2008): Emissionsfaktorer til beregning af ammoniakfordampning ved lagring og udbringning af husdyrgødning (In Danish). Aarhus Universitet, Det Jordbrugsvidenskabelige Fakultet. (DJF Husdyrbrug; Nr. 84).

[5] Amon, B., Kryvoruchko, V., Amon, T. & Zechmeister-Boltenstern, S. 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agriculture, Ecosystems and Environment 112, 153-162.

[6] Hjorth M. K.V. Christensen, M.L. Christensen, S.G. Sommer (2009): Solid–liquid separation of animal slurry in theory and practice. A review. Agron. Sustain. Dev. 30 (2010) 153–180. INRA, EDP Sciences, 2009.

Thematic Area TC – Advances in emission prevention (Poster presentations)

Ina Körner: Trees, processed photograph

TC-P_01 In-house distribution of ammonia and greenhouse gas concentrations in a laying hen facility

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1. Objectives

Gaseous losses and air quality are areas of concern in laying hen facilities [1]. Standard method for estimating gas losses requires measuring them at exhaust fans. However, the composition of the exhaust air may be different from the inside air composition. It was aimed to assessing the spatial variability of ammonia (NH_3) and greenhouse gas (GHG) concentrations in a laying hen facility.

2. Methodology

The study was conducted in a commercial laying hen building from the Basque Country (northern Spain) adapted to Directive 1999/74/EC with 53,000 Lohmann-Brown hens. The house was 17 m wide and 66 m long and enriched cages were arranged in 6 rows of 9 tier cages. The lighting period was 17:7 (light:dark) hours per day. The farm was selected to be representative of the current egg production facilities in the Basque Country in terms of management practices.

Bird mortality, laying rate, productivity, feed intake and conversion efficiency was recorded on a daily basis by the producer and provided to the researchers to calculate bird inventory and productive parameters. Manure was stored on belts under the cages from 2 to 4 days before being discharged. The producer recorded manure removal frequency.

Ammonia (NH_3), nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) concentrations were continuously monitored from February to September 2014 by a photoacoustic analyser (INNOVA 1412, LumaSense, Denmark). According to the technical specification of the instrument, the detection limit of the measurement was 0.2 ppm for NH_3, 0.4 ppm for CH_4, 0.03 ppm for N_2O and 1.5 ppm for CO_2.Twelve sampling tubes were distributed into the exhaust fans, inside the laying hen cages, on manure belts and in the corridors. Air samples were taken from inlet (1 sampling point), outlet air (3 sampling points), cage level (3 sampling points), belt level (3 sampling points) and corridor level (2 sampling points).

An automated system regulated inside temperature through windows opening, cooling system and the activation of 18 fans (Tecno Poultry Equipment, Macronew 3, Italy) set up within a tunnel ventilation system. Five sensors were used to record air temperature and relative humidity in the building. One of them was placed outside the house, one of them by the air inlet, other two, close to the fans and the other one in a cage. Air temperature and relative humidity were monitored and recorded every 15 min using 5 data loggers (Onset, HOBO U12-013, USA) with a precision of ±0.35ºC and ±2.5% respectively.

Ventilation rate was continuously measured under the usual rearing conditions of the farm following the methodology described by Calvet et al. (2010). The average percentage of operation of each fan was obtained every 5 minutes. An electronic data logger system (Binary Devices S.L., Datalogger 244, Spain) converted every second the electric signal from each fan into digital data on fan status. All on-line sensors were unloaded every 10 days by using specific software. Each fan was individually calibrated for airflow rate at different pressure drops. During calibration, the air was ducted 30 cm from each fan and the air velocity was measured at 25 different locations in the section using a hot wire anemometer (Testo, 425, Germany) [3]. Pressure drop in the building was controlled and recorded every 5 min by a pressure drop meter (Veris, PX, USA).

Calculations

A ventilation performance curve (Eq.1) was obtained according to the average airflow rate associated to each pressure drop recorded in the building.

$$Y = -0.4405x + 43.909$$
$$R^2 = 0.7378 \qquad \text{Eq.1}$$

where,

Y = *Airflow rate* ($10^3 m^3 h^{-1}$)

X = Pressure Drop (Pa)

Total airflow rate for each hour was calculated by integrating the number of operating fans and the individual airflow rate as given in Eq.1 for each pressure drop recorded.
Gas emissions were calculated by means of a gas balance method in the volume determined by the building, according to Equation 2.

$$E = (C_{outlet} - C_{inlet}) \times V \qquad \text{Eq.2}$$

Where E is the emission (mg/h), C_{outlet} and C_{inlet} is the outlet and inlet gas concentration, respectively (mg/m^3), and V is the ventilation rate in the building (m^3/h).
Emission were expressed both per hen (hens present in the hen house on the measurement day) in mg $d^{-1}hen^{-1}$ or per animal mass as g h^{-1} 500 kg^{-1} LW (considering for each week the number of hens present in the building and the corresponding weight).

3. Results and discussion

Results showed that NH_3 concentration was significantly higher on manure belts (*P* < 0.05). Mean NH_3 concentration on these sites was 3.9 mg NH_3 m^{-3}. Threshold limit value for animal welfare, which is established at 25 ppm [4], was not reached during the experimental period.

Table 1: Ammonia and GHG concentration in different points, at two different seasons.

Gas concentration, mg m^{-3}	Belt	Cage	Corridor	Outlet	Inlet
NH_3					
Warm season	3.5	2.7	2.3	2.3	0.7
Cold season	4.4	3.8	3.7	3.6	0.7
CO_2					
Warm season	1573	1674	1630	1516	763
Cold season	2454	2692	2852	2169	891
CH_4					
Warm season	4.5	4.4	4.2	4.1	3.3
Cold season	3.3	3.1	3.3	2.7	1.3
N_2O					
Warm season	0.48	0.48	0.48	0.47	0.46
Cold season	0.67	0.69	0.69	0.65	0.58

Warm season: May - September; Cold season: February - April

Ammonia concentration at exhaust fans was overall 23% lower than those observed on the belts. However, a seasonality effect was observed between both sampling sites (*P* < 0.05). Ammonia concentration at fans was 34 % lower in warm season while the difference was 20% in the coldest months (Table 1).

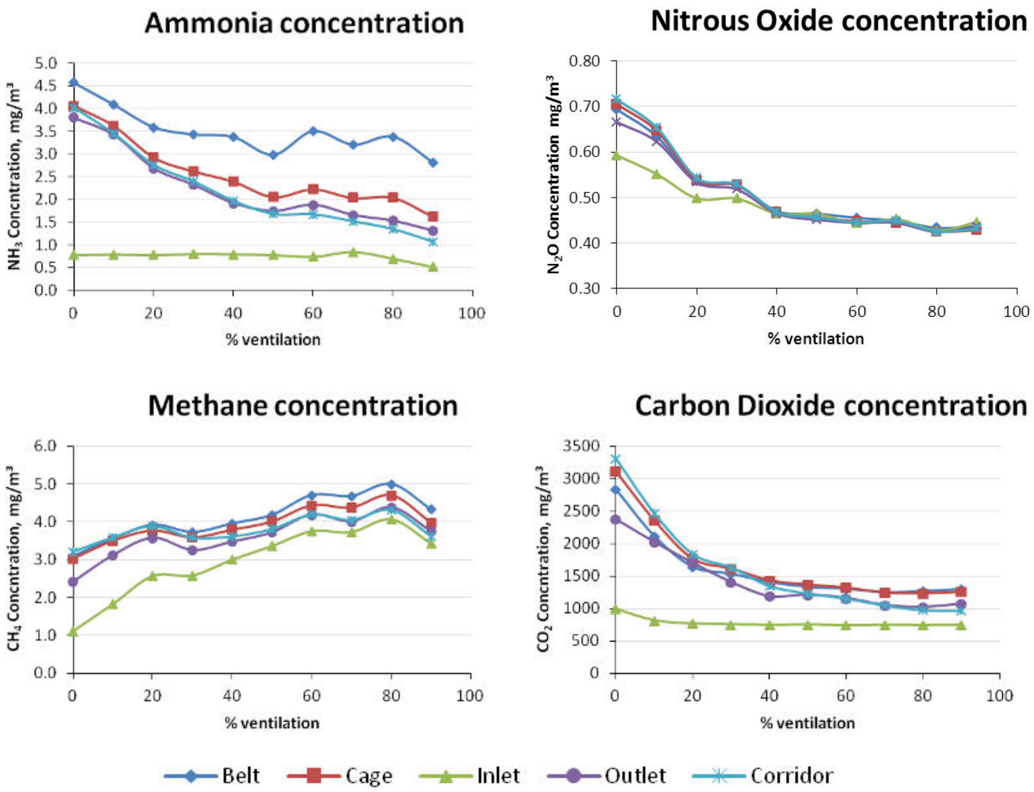

Figure 1: Ammonia and GHG concentration in different points, at different ventilation percentages.

Carbon dioxide emissions were significantly higher either inside the laying hen cages or in the corridors (P < 0.05). Carbon dioxide concentration averaged around 2,200 mg CO_2 m^{-3} in both areas. Carbon dioxide concentrations were also below the threshold limit estimated for animal welfare (3,000 ppm). Carbon dioxide concentration was 17% lower at exhaust fans, significantly affected by the difference observed in the coldest season (21%).

Methane and N_2O losses did not differ among the selected sampling sites and averaged 3.7 and 0.57 mg m^{-3}, respectively.

The low emission of N_2O made difficult the analysis of the obtained data. Regarding CH_4, gas concentration raised as the ventilation increased, what also happened in the inlet point. This suggests that the data are questionable because the inlet air concentration should not be affected (Figure1).

Table 2: Emission of NH_3 and GHG at different ventilation rates.

Gas	Unit	% ventilation*									
		0-10	10-20	20-30	30-40	40-50	50-60	60-70	70-80	80-90	90-100
NH_3	mg hen^{-1} d^{-1}	70	113	147	169	162	165	228	194	224	227
N_2O	mg hen^{-1} d^{-1}	1.7	3.3	3.5	3.8	2.9	3.1	3.3	3.7	4.6	5.8
CO_2	g hen^{-1} d^{-1}	32	53	73	71	63	79	81	71	73	93
CH_4	mg hen^{-1} d^{-1}	31	59	83	82	74	87	94	80	84	100

*100% ventilation = 656,167 m^3 h^{-1}

Although the concentration of all the gases dropped, we realized that the emission increased in all of them. The increase of NH_3 emission was substantial and reached a high emission earlier than other gases when ventilation rate was increased (Table 2). This was aligned with the more moderate decline suffered by the NH_3 concentration. This was more appreciated in the emission source, which was the manure accumulated on the belt (Figure 1).

4. Conclusion and outlook

Ammonia and carbon dioxide distribution is variable inside the laying hen facilities. Gaseous sampling locations must be consciously analysed if it is aimed at assessing the air quality in terms of animal welfare or estimating the gaseous losses at farm level. Seasonality must be necessarily considered in this kind of studies.

It should continue working on the optimization of the methodology of measurement of gases at the farm level. It has been appreciated a great difference between the measuring points, as well as at different moments. The differences in ventilation and management affected the emission rate. If those studies in which this was not take into account, there could be mistakes in reported values of emission. In addition, the correct determination of the measurement points and the conditions would make studies more comparable.

Acknowledgements

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References

[1] Nimmermark, S., Vonne, L., Gustafsson, G., Wijnand, E. (2009). Ammonia, dust and bacteria in welfare-oriented systems for laying hens. Annals of Agricultural and Environmental Medicine 16, 103-113

[2] Calvet, S., Cambra-López, M., Blanes-Vidal, V., Estellés, F., & Torres, A.G. (2010). Ventilation rates in mechanically-ventilated commercial poultry buildings in Southern Europe: Measurement system development and uncertainty analysis. Biosystems Engineering, 106, 423–432.

[3] ASHRAE (2001). ASHRAE Fundamentals Handbook e Printed edition. Atlanta, Georgia: American Society of Heating, Refrigerating and Air-Conditioning Engineers Inc.

[4] UEP. 2014. Animal husbandry guidelines for U.S. egg laying flocks (2014 Edition). United Egg Producers ed. (http://www.unitedegg.org/information/pdf/UEP-Animal-Welfare-Guidelines-2014.pdf10.3382/ps/peu076.html)

TC-P_02 The effect of slurry composition on methane and ammonia emissions from fattening pig slurry – A review of three nutrition assays

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1. Objectives

The aims of this study were to assess the effects of pig slurry composition, obtained from three nutrition assays, on the biochemical CH_4 potential (B_0) and NH_3 emission potential.

2. Methodology

A total of 84 slurry samples were obtained from fattening pigs (50-100 kg live weight) housed individually in metabolic cages. Animals were fed with 13 different diets that included different types and sources of fibre (orange pulp and carob meal, study 1), different sources of protein (soybean meal, sunflower meal and wheat DDGS, study 2) and different inclusion levels of fat and fibre (calcium soap of palm fatty acids distillate and orange pulp, study 3). Faeces and urine were collected separately during 3 days after 20 days of adaptation to the experimental diet, and stored at 4ºC. Slurry was later re-composed according to the original faeces:urine excretion ratio.

In vitro potential NH_3 emissions (E-NH_3) were determined in duplicate in the laboratory following the methodology described by Portejoie et al. [1] over 11 days. The ultimate CH_4 yield (B_0) of each slurry sample was determined in triplicate through biodegradability assays in 125 mL bottles over 100 days at mesophilic conditions (35ºC) according to Vedrenne et al. [2].

Slurries were also analyzed for dry matter (DM), volatile solids (VS), total Kjeldahl nitrogen (TKN), ammonium nitrogen (N-NH_3), pH and volatile fatty acids (VFA) (all expressed on a wet basis). Chemical analyses of samples were conducted in triplicate. Cellulose (CEL) and Hemicellulose (HEM) content of faeces were also determined. The shift of excreted nitrogen between faeces and urine (F_U-Ratio) was also calculated.

In order to explore the effects of slurry composition on gaseous emissions, correlations between slurry characteristics and emissions were determined. Individual and multiple linear regression analyses were also conducted to explore these relationships. All analyses were conducted using Statgraphics Centurion XVI.

3. Results and discussion

3.1 NH_3 emission potential

Ammonia emissions averaged 1.87 g N-NH_3/kg of fresh slurry. A high variability was found (CV > 20%) among different diets. Surprisingly, despite the variability recorded, when exploring the whole database from the three studies, no significant relationships were found with the main slurry composition parameters (DM, VS, TKN, N-NH_3, VFA or pH). At study level (study 2), pH and N-NH_3 content did influence NH_3 emissions, with negative and positive correlations, respectively.

For the complete database and at study level, NH_3 emissions were negatively correlated ($r=-0.6$, $p<0.05$) with the F_U-Ratio. This effect has been described before in the literature and is related to the nature of the nitrogen present in faeces and urine (mainly organic and mineral respectively). Despite this relationship being statistically significant, the linear regression did not provided a good prediction ($R^2<0.4$) when considering all data. Nevertheless, if exploring only data from study 1 (in which fibre sources of diet were modified) a linear relationship could be identified (Figure 1).

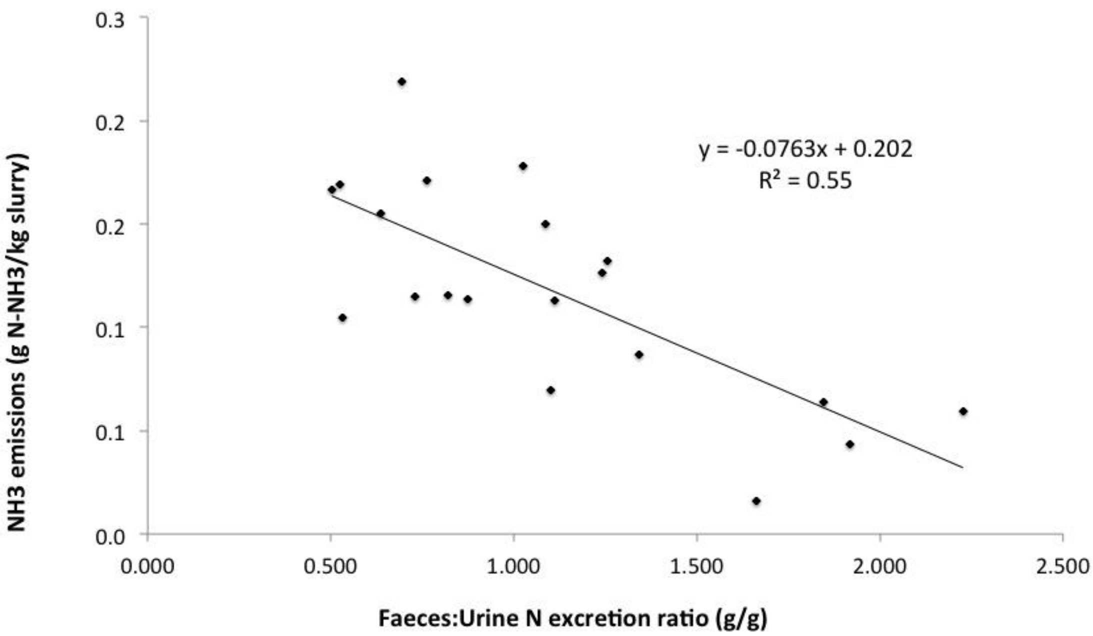

Figure 1: Relationship between ammonia emissions and faeces:urine N excretion ratio for study 1

Faecal content was positively correlated with CEL and HEM (r>0.6, p<0.05). Moreover, if considering the addition of both, cellulose and hemicellulose content in faeces, the correlation improved (r>0.8, p<0.05).

Among others (not presented in this paper, due to their low relevance), the following significant multiple linear regression model (R^2=0.67) was obtained:

$$NH_3\ (g\ N - NH_3/kg\_slurry) = -0.922 + 0.18 \times TKN(g/kg_{DM}) - 0.71 \times F\_Uratio \qquad \text{Eq. 1}$$

As already reported, the faeces:urine nitrogen partition seems to play a crucial role in ammonia emissions, as well as the amount of total Kjeldhal nitrogen available in the slurry.

3.2 CH_4 emissions

B_0 (g CH_4 per gram of VS) ranged between 256 and 430 among treatments and was conditioned by the type of diet (increasing with diets rich in fat and fibre). Slurry and VS excretion was also affected by diet and gave major variations in predicted CH_4 emitted (VS excreted times B_0) per animal and day (66-144 L CH_4/day). In both cases, diets rich in fat and fibre resulted in higher emissions.

Faeces and slurry pH were positively and strongly correlated with B_0 (r=0.72 and r=0.62, respectively, p<0.05). A slight relationship between faeces and slurry pH and VFA content of slurry was found, and could be an explanation for this correlation. Nitrogen content of faeces was positively correlated with B_0 (0.61, p<0.05) and CH_4 produced per animal (r=0.77, p<0.01). No other relevant correlations with effluent characteristics were found when analysing data from all experiments together.

Nevertheless, when looking at individual studies, a positive relationship between CH_4 emissions and fat and soluble fibre content of slurries was identified (study 1). Moreover a positive relationship was identified in study 2 between fibre content of slurries (both fermentable and non-fermentable) and CH_4 emissions. Finally, fat content of slurries played a key role (study 3) leading to higher CH_4 emissions when increased.

When looking again at the whole datasheet, the following significant regression models were obtained (R^2=0.52 and 0.75 for Eq. 2 and 3 respectively):

$$B_0\ (mL/g\ VS) = -458 + 119.4 \times pH \qquad \text{Eq. 2}$$

and

$$CH_4\ (L/animal\ day) = 78.5 - 0.82 \times UrineDM(g/kg) + 0.39 \times FaecesCP(g/kg) \qquad \text{Eq. 3}$$

Although no clear relationship was found between effluent composition and potential CH_4 emissions when looking at the complete datasheet, diet composition was a critical factor. The inclusion of different types of fibre and fat changed potential emission per animal by two-fold because of changes in B_0 of slurry and the amount of excreta produced.

4. Conclusion and outlook

Although no relevant relationships were found between effluent composition and potential emissions when dealing with the whole database, some interesting correlations were identified at study level. In general terms, the inclusion of different types and amounts of fibre and fat changed potential CH_4 emission from manure per animal by two-fold because of changes in B_0 of slurry and the amount of excreta produced. Regarding NH_3, the shift between nitrogen excreted in faeces and urine was the only factor that presented a consistent effect for all studies. Among strategies studied, it seems that modifying nitrogen sources in the diet resulted in the most effective strategy to reduce NH_3 emissions from pig slurry.

This work revealed the high variability that can be found in gaseous emissions from slurries depending on their composition and the interaction between factors. Several factors are involved in processes leading emissions and, when looking for prediction equations it is difficult to identify straight relationships between slurry composition and emissions.

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References

[1] Portejoie, S., Dourmad, J.Y., Martinez, J., & Lebreton, Y. (2004). Effect of lowering dietary crude protein on nitrogen excretion, manure composition and ammonia emission from fattening pigs. Livestock Production of Science, 91, 45-55.

[2] Vedrenne, F., Béline, F., Dabert, P., & Bernet, N. (2007). The effect of incubation conditions on the laboratory measurement of methane producing capacity of livestock wastes. Bioresource Technology, 99, 146–155.

TC-P_03 Potential for biogas production from anaerobic fermentation of vinasse in Iran

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1. Objectives

Biofuels have received much attention as alternative fuels, yet despite being an attractive energy source, their production causes environmental pollution. One of the most prominent biofuels is ethanol, which though presenting many advantages, there are several disadvantages associated with its waste disposal, such as the uncontrolled disposal of vinasse. The objective of this study is: To investigate the vinasse's potentials in biogas, produced by anaerobic digestion process in Iran.

2. Methodology

The following roadmap illustrates the study:

- Literature research (this research is a case study based on some relevant published data in literature and scientific articles. It has been conducted on the basis of theoretical data to calculate the potentials of methane production. The sources, consulted for the preparation of this paper, were Iranian governmental organizations and producers of sugar cane and ethylic alcohol in the world).
- Recognizing the position of Iran in terms of sugar cane cultivated area and daily production of fine ethanol
- Surveying physical and chemical parameters of vinasse.
- Calculation of biogas production capacity from vinasse in Iran and selection of the best case to use biogas, produced from anaerobic fermentation of vinasse.

3. Results and Discussion

3.1 Sugar cane production status in Iran

Iran is a sugar cane producer with nearly 60.4 thousand hectares of cultivated area [1].According to the last census in Iran, between 2009/2010 crop, 3.035 million tons of sugar cane were processed [2], producing 110 thousand liters of fine ethanol per day [3].In 2012, Iran was estimated to be a main producer of alcohol from sugar cane in Middle East.

3.2 What is Vinasse?

One of the most important but rarely discussed subjects, regarding the negative impacts of sugar cane, is vinasse, a by-product of ethanol industry (Fig.1), [4].

Vinasse is a contaminant in aquatic and terrestrial environment with high pollution potentials, approximately one hundred times more than household sewage. It has low pH, high corrosive impact, and high levels of biochemical demand of oxygen (BOD).It also has high organic matter content, causing a depletion of oxygen [4].

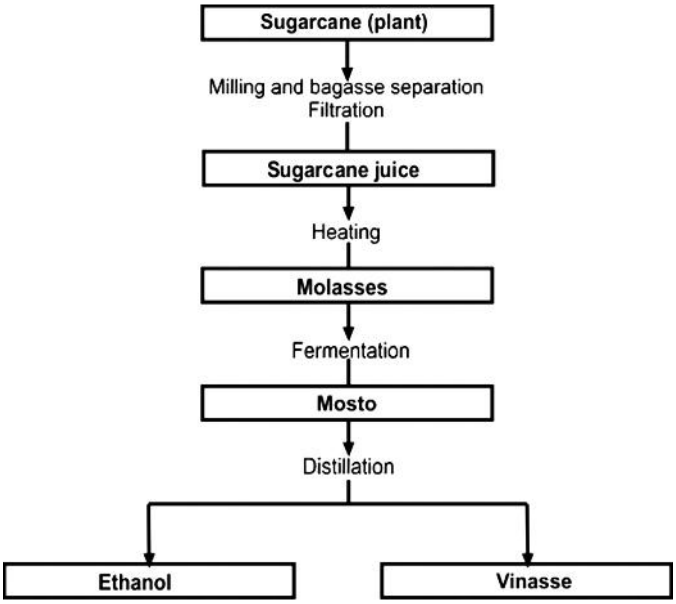

Figure1: Flowchart of ethanol production process and underproduction of sugar cane vinasse [4].

3.3 Physical and Chemical Parameters of Vinasse

In Brazil, ethanol plants use vinasse as a fertilizer, which is applied directly in the soil, as it is rich in potassium. On the other hand, environmental agencies impose some limits to its application [5]. In Iran, agricultural soils are salty, therefore, as a fertilizer vinasse cannot be used directly in the agriculture. Hence vinasse disposal has become an environmental problem for the plants. Vinasse storage in lagoons causes greenhouse gas due to its evaporation.
An alternative, increasingly on the rise in ethanol industry, is the anaerobic digestion of the organic load of vinasse. This process consists of the biodegradation of the organic load of vinasse to produce biogas and biofertilizer [4].
Before being disposed in the soil, vinasse should be treated in the Up flow Anaerobic Sludge Blanket (UASB) reactor. This kind of treatment will produce bio fertilizers and biogas. Table 1 shows the physicochemical parameters of vinasse (before and after biodigestion) in São Martinho Industry of Sugar and Alcohol [6].

Table 1: Physicochemical characteristics of vinasse feed and effluent in Sao Martinho sugar factory biodigestion plant

Parameter	Vinasse (before biodigestion)	Effluent (after biodigestion)
pH value	4.0	6.9
Chemical oxygen demand, COD in mg/L	29,000	9,000
Nitrogen total in mg/L	550	600
Ammonia nitrogen in mg/L	40	220
Phosphorus as P_2O_5 in mg/L	17	32
Sulfate in mg/L	450	32
Potassium K_2O in mg/L	1,400	1,400

According to table 1, the anaerobic digestion of the vinasse will reduce the organic load, but its power as a fertilizer will remain the same. It is very important to note that the potassium content of the digest after the biodigestion process of vinasse is the same as before, whereas readily-available nitrogen contents increase.
According to [5],biogas is composed of methane (40-75%), carbon dioxide (25-40%), and other compounds. It can be used as a biofuel in engines with internal combustion.

3.4 Calculation of Biogas production capacity of vinasse in Iran and selection of the best case in using biogas

To carry out an economical evaluation of biogas use as a fuel for electrical and steam generation or heat production, it is necessary to calculate its production capacity. According to [5], the production of 1 liter of alcohol discharges 10 liters of vinasse and with the digestion of 1 m^3 of vinasse approximately 14.6 m^3 of biogas is produced. These values were used to estimate the production of biogas and vinasse in the sugar cane plants, under study, and the total estimation of biogas production in Brazil.

But in Iran, Razi Yeast & Alcohol Co. produces 110 m^3/day of alcohol and 1,100 m^3/day of vinasse from sugar cane. It was calculated that the amount of vinasse, annually generated in the reference plant, is about 330,000 m^3/year and this amount of vinasse can produce 4,820,000 m^3/year biogas.

According to literature [5] biogas (55% methane) has an energy value of 20.8 MJ/m^3 while diesel has 36.4 MJ/L. In the composition of biogas, there is approximately 55% of methane (CH4), which has a specific mass of 0.714 kg/m^3, thus 1 m^3 of biogas is equivalent to 0.393 kg methane .Therefore, by running a biogas plant in Razi Yeast &Alcohol Co. 2,651,000 m^3/year of methane can be obtained. According to biogas conversion factor, equal to 0.45 m^3 of biogas/kg COD [6], anaerobic digestion of vinasse will reduce environmental risks in nature.

Vinasse can be a source to generate electricity by burning the generated biogas (9.88 kWh/m^3 methane) [7].Iran is potential to produce approximately 5,302,000 m^3/year of methane from vinasse in near future, which would result in 52.4 GWh of electricity per year.

Five alternatives could be presented for the use of the biogas, resulting from vinasse anaerobic digestion [6]:

- Case I: Electric power generation with RCE (Reciprocating Combustion Engines)
- Case II: Electric power generation with gas MT (Gas Micro Turbine)
- Case III: Electric power generation through bagasse co-firing with the biogas in the boilers
- Case IV: The sale of the substituted bagasse (for generation of electric power in another unit)
- Case V: The sale of dried yeasts (use of biogas as fuel in spray dryers) (Fig. 2):

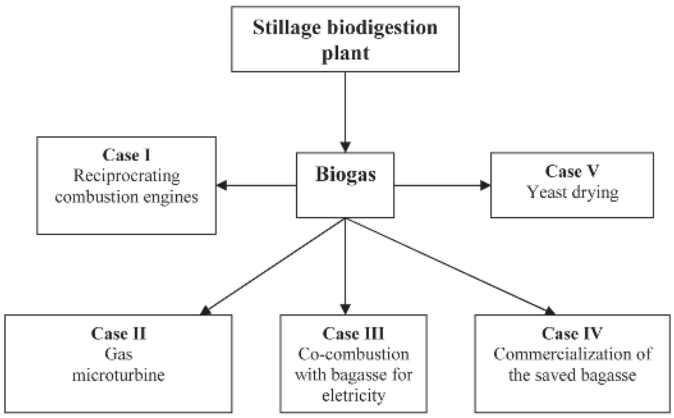

Figure 2: Presented alternative cases for the utilization of the biogas energy [6].

Regarding Iranian agriculture properties, combination of cases III and V (i.e. co-combustion with bagasse for yeast drying) is the most suitable option.

4 Conclusion and Outlook

Among the produced waste, sugar cane vinasse has received special attention, facing endless possibilities for its reuse and disposal. The anaerobic digestion process of vinasse in Iran, or the use of effluent of vinasse (after digestion) as biofertilizer in the agriculture and production of methane is one of the best alternatives for the sugar cane vinasse reuse and disposal. Evaporation of vinasse in the storage lagoons, producing greenhouse gas, is an obsolete manner. Among alternative cases for the utilization of the biogas energy, co-combustion with bagasse for yeast drying is one of the best alternatives for Iranian sugar cane industry.

Acknowledgments

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References

[1] Iranian Agricultural Ministry, (2011),Agricultural Statistics, Volume I, Available at:http://www.maj.ir/.

[2] Hemayati S.S.;.,Hamdi, H.; Taleghani D.F. and Amili H.,(2011): Strategic framework for sugar cane research, Iranian sugar cane research& training institute, sugar beet seed institute.

[3] RaziYeast & alcohol, Sugar Cane & By-product Development Co.2015, Available at: <http://www.yarazi.com/Alcohol.aspx

[4] Fontanetti, C.S,;Christofoletti, C.A.; Escher, J.P.; Correia, J.E and Marinho, J.F.U, (2013): Sugar cane vinasse (Environmental implications of its use), Waste Management, 33:2752-2761.

[5] De Souza S. N. M.; Santos R. F; Fracaro, G.P.M,(2011):Potential for the production of biogas in alcohol and sugar cane plants for use in urban buses in Brazil, In proceeding at World Renewable Energy Congress 2011-Sewden

[6] Salamon, K.R.; Lora, E.E.S.; Rocha, M.H and del Olmo, O.A., (2011): Cost calculations for biogas from vinasse biodigestion and its energy utilization, Sugar Industry4: 217-223..

[7] Santos, R.F.; Borsoi, A.; Secco, D.; de Souza, S.N.M and Constanzi, R.N., (2011): Brazil's potential for generating electricity from biogas from stillage, In proceeding at World Renewable Energy Congress 2011-Sweden

TC-P_04 Flux chamber measurements of ammonia and nitrous oxide emission from organic beddings in bedded pack dairy barns

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1. Introduction

A bedded pack barn is a different type of housing for dairy cows. The common free stall barns with cubicles and slatted concrete floors with slurry storage underneath is widely used among Dutch dairy farmers and is considered as the reference system. In 2007 the bedded pack barns were introduced in The Netherlands by a group of farmers who got inspired by similar alternative housing systems in the USA and Israel. Bedded pack barns are characterized by a combined walking and lying area. This area consists of soft and in most cases organic material that either absorbs or converts the deposited faces and urine. Because of this active manure management, bedded pack barns perform differently compared to the conventional, reference systems concerning sustainability aspects like economics, animal welfare, milk quality, manure quality and environment. A project commissioned by both the ministry of Economic Affairs (EZ) and the Dutch Dairy Board (PZ) aimed to evaluate those sustainability aspects of the bedded pack barn.
One of the aspects of the environmental impact of bedded back barns is the gaseous emissions of C- and N-compounds. These can emit in different forms depending on material composition, the quality of the degradation processes and external climatic parameters. Most important N- gases with environmental impact are ammonia (NH_3) that leads to acidification and eutrophication and nitrous oxide (N_2O) that is a greenhouse gas. Also N_2 might emit, but has no negative impact on the environment and only represents a loss of valuable nitrogen.

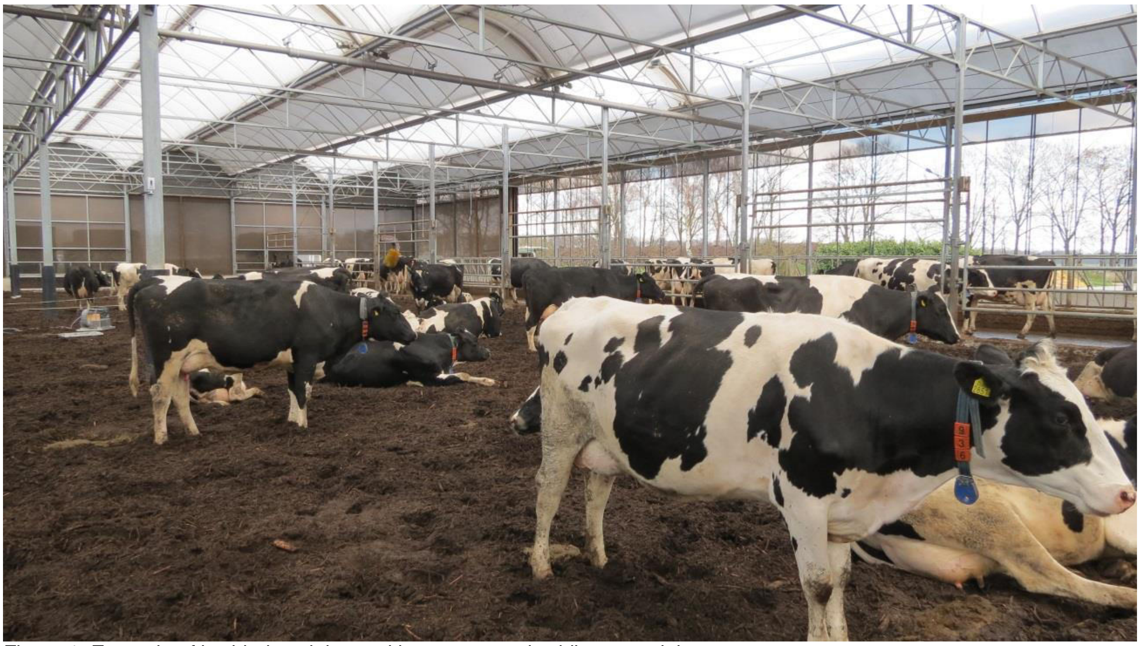

Figure 1: Example of bedded pack barn with compost as bedding material

Gaseous emissions of bedded pack barns have not been measured extensively before. However, emissions of deep litter systems for cows and pigs and other type of litter based systems for dairy and beef cows have been researched [1]. Also emissions during composting are reported by numerous authors. The combination of composting processes with a constant input of faces and urine and frequent tillage of the top layer influences the nutrient losses through gaseous emissions. This makes an assessment of these types of barns necessary.

The objective of this study was to provide insight into the differences of gaseous nitrogen losses between two of the most widespread bedding materials (compost and woodchips) applied in the Netherlands.

2. Methodology

Bedded pack barns in the Netherlands can be divided in two groups. One group uses organic materials, mostly compost, as bedding material. Urine is absorbed in the bedding and faces is mixed with the compost by daily tillage of the top layer using a cultivator or a similar device. The aim of this process is to keep the upper layer dry, by mixing the moist faces and urine with the lower bedding layers. The other group also uses organic materials but with the aim to trigger an (aerobic) composting process of bedding and manure in the bedding. This latter process should produce sufficient heat to evaporate excess moisture and keep the top layer of the bedding dry. Wood chips are the most popular bedding material used for this co-compost process. Urine is partly absorbed in the solid matrix and partly evaporated. The faces is also mixed with the material but the tillage of the top layer also aims at the stimulation of the composting processes. In most cases a tractor-driven device is used to intensify the loosening effect. In addition to tillage of the top layer farmers sometimes install a (forced) aeration system at the bottom of the bedded pack. These aeration systems use blowers either to blow outside air upwards through the bedding material or to create a downwards airflow through the bedding by sucking and thereby concentrating the air at a single emission point. Temperatures in these composting bedded pack barns can rise to above 60^{o}C.

Table 1: Farm data.

Farm #	**Days #**	**Material Bedded pack area**	**Bedded pack area m^2/cow**	**Material Walking area**	**Walking area m^2/cow**
1	4	Wooden chips (WC)	12,5	Concrete slats	5,0
2	2	Wooden chips (WC)	15,0	Concrete slats	4,0
3	2	Wooden chips (WC)	15,0	Concrete slats	4,0
4	4	Wooden chips (WC)	16,0	Solid asphalt	3,0
5	2	Wooden chips (WC)	8,5	Concrete slats	1,5
6	2	Compost (C)	18,0	Concrete slats	4,0
7	2	Compost (C)	22,0	None	0,0
8	1	Compost (C)	9,5	Concrete slats	7,0
9	2	Compost (C)	12,5	Concrete slats	4,0

The ammonia emission from the bedding was measured at nine farms (see table 1). Emission flux was measured using an open flux chamber several times (n) on different locations in the bedding at different days (see table 1 and 2). Ventilation was set at a constant value of 30% of the maximum capacity resulting in an average air velocity across the emitting surface of 0,57 m/s. This is within the range of possible air velocity in a typical naturally ventilated dairy barn. Both incoming and outgoing air were sampled during the last 15 minute of a 30 minute period at which the flux chamber was placed on the bedding. Air samples were led with a restricted flow (~1000 ml/min) through two glass impingers placed in serial and both put in an acid solution. The ammonia emission was calculated from the average ammonia concentration difference between ingoing and outgoing air and the ventilation rate in the sampling device.
The nitrous oxide emission was measured using the same flux chamber but closed. Nitrous oxide concentrations were measured (after filtering out CO_2 with soda lime) using a photo acoustic multi gas sampler (Innova 1312) at a sampling interval of 2 minutes. Emission was calculated using the method presented by [2] and [3]. Averages on farm level have been statistically analyzed with t-tests using Genstat 17 [4]

3. Results

The results of ammonia and nitrous oxide emission measurements are presented in the table below (SE: standard error). Averaged ammonia emission on farm level (SE between brackets) from wooden chips bedding was 186,3 mg/m^2/h (46,8) and from compost bedding 409,1 mg/m^2/h (151,8). Average nitrous oxide emissions from wooden chips bedding was 18,8 mg/m^2/h (4,6) and from compost bedding 14,0 mg/m^2/h (6,4).

Table 2: Average ammonia and nitrous oxide emissions per farm.

Farms	n	NH_3 emission (mg/m^2/h)	SE	n	N_2O emission (mg/m^2/h)	SE
1	16	154,2	83,8	13	7,3	1,9
2	9	163,8	38,4	4	8,3	3,6
3	9	207,6	39,7	6	25,4	9,2
4	16	346,5	65,7	7	23,6	5,9
5	7	59,6	28,0	2	29,6	28,1
6	10	356,2	105,7	38	27,5	14,2
7	10	837,8	152,7	1	1,4	
8	5	319,1	80,1	5	22,1	7,2
9	10	123,2	63,2	8	5,2	1,9
WC	5	186,3	46,8	5	18,8	4,6
C	4	409,1	151,8	5	14,0	6,4

4. Discussion

Flux chamber measurements of ammonia emission from a conventional concrete slatted floors barn with slurry pits resulted in an emission of 1200 mg/m^2/h [5]. Such a cubicle barn typically offers 4 m^2 emitting area per cow. According to the measurements, the ammonia emissions per m^2 of bedding did not differ significantly between compost and wood chips (p=0,16) and were both significantly lower than the emission per m^2 from the reference system (p=0,01). The relative emissions per cow of both materials were higher than the reference (figure 1). This means that both the number of m^2 per cow and the emission from the (concrete) walking area play an important role in the establishing of total emission from the barn (based on flux chamber measurements). To reduce ammonia emissions from a bedded pack barn one can therefor either try to influence the emission from the bedding or from the (concrete) walking area. One option to reduce ammonia emissions is to use a solid floor with a lower emission per m^2 as is used on farm 4.

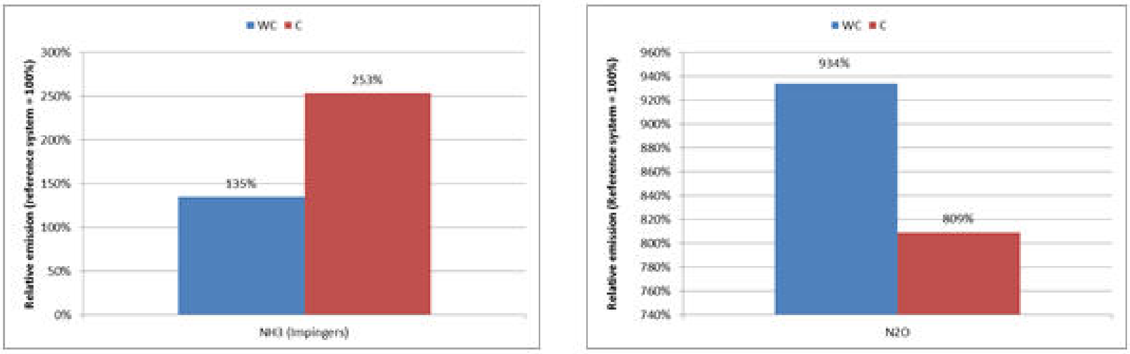

Figure 2: Relative emission per cow of ammonia (left) and nitrous oxide (right) of the two bedding materials.

The N_2O emissions per m^2 of bedding did not differ significantly between compost and wood chips (p=0,56). The calculated N_2O emission factor for a slurry based reference systems is 0,23 kg N_2O per cow per year [6]. The relative emission per cow is given in figure 1. All farms produced per cow more N_2O than the reference system, up to almost 19 times more but only for wood chips the average emission per cow was significantly higher (p=0,02). This higher emission of N_2O from organic bedding compare to a slurry based system is probably caused by the more intensive microbiological processes that take place in the bedding. In these processes,

N_2O is produced during both the nitrification an denitrification processes when both aerobic and anaerobic circumstances are simultaneously present in the composting mixture. Also temperature, pH and oxygen availability play an important role and may partly explain the small differences in N_2O emissions between wood chips and compost.

Increase of available area per cow and use of soft organic bedding material to improve cow welfare does lead to higher gaseous losses of nitrogen from housing both through ammonia and nitrous oxide. However, environmental assessment of these kind of housing systems should also include storage and application.

5. Conclusion

Ammonia emission per m^2 from a compost bedding is more than two times higher than from a wooden chips bedding but both significantly lower than from the reference system. Increase of available area to improve cow welfare but leads to higher ammonia emissions per cow compared to the slurry base reference system. Nitrous oxide emissions were 8 to 16 times higher than the reference system. Total nitrogen losses per cow from housing increased. Losses during storage and application should be evaluated in the future.

References

[1] Mosquera, J., J.M.G. Hol, J.W.H. Huis in ‘t Veld (2005) Emissions from a naturally ventilated deep litter barn for dairy cows [Onderzoek naar de emissies van een natuurlijk geventileerde potstal voor melkvee. I Stal] In Dutch. Rapport 324, Agrotechnology & Food Innovations, Wageningen

[2] Hutchinson, G.L. and A.R. Mosier (1981) Improved soil cover method for field measurement of nitrous oxide fluxes. Soil Science Society of America Journal 45 (2) p. 311-316.

[3] Mosquera, J. G.J. Kasper, K. Blanken, F. Dousma, A.J.A. Aarnink (2010) Development of a fast measurement method for the determination of ammonia emission reduction from floor related measures. [Ontwikkeling snelle meetmethode ter bepaling van ammoniakemissiereductie van vloergebonden maatregelen] In Dutch. Rapport 291, Wageningen UR Livestock Research, Wageningen.

[4] VSN International Ltd, GenStat Seventeenth edition

[5] Dooren, H,J.C.; P.J. Galama; M.C.J. Smits; W. Ouweltjes; F. Driehuis; S. Bokma (2012) Bedding material in loose housing systems for dairy cattle. In Dutch. Rapport 411, Wageningen UR Livestock Research, Lelystad

[6] Mosquera, J. and J.M.G. Hol (2012). Emission factors for methane, nitrous oxide and PM2.5 for livestock housing, including explanation. [Emissiefactoren methaan, lachgas en PM2,5 voor stalsystemen, inclusief toelichting] In Dutch. Rapport 496, Wageningen UR Livestock Research, Lelystad.

TC-P_05 Impact of cattle-slurry treatment by separation and acidification on gaseous emissions after soil application

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1. Objectives

Cattle-slurry management became a priority in many livestock farms and slurry treatment is used to increase the fertilizer value of slurry and/or minimize its environmental impact. Indeed, significant emissions of ammonia (NH_3) and greenhouse gases (GHG) as nitrous oxide (N_2O), carbon dioxide (CO_2) and methane (CH_4) can occur during and after slurry application to soil. Application of acidified slurry or liquid fraction (LF) obtained by solid-liquid separation are two alternatives to raw slurry application that have proven to be efficient to minimize ammonia emissions. However, few is known about its effect on GHG emissions.

The aim of the present work was to assess the efficiency of cattle slurry treatment by acidification and/or solid liquid separation to mitigate ammonia (NH_3) and greenhouse gases (GHG) emissions following surface application to a sandy loam soil.

2. Methodology

Raw slurry (S) from a commercial dairy farm located close to Lisbon (Portugal) was first treated by centrifugation to obtain a liquid fraction (LF). Half of S and LF were then acidified to pH 5.5 by addition of concentrated sulphuric acid leading to acidified slurry (AS) and an acidified liquid fraction (ALF). Slurries' composition is presented in Table 1. These slurries were then applied to a Portuguese sandy loam soil (39º 49' 17'' N 07º 27' 44'' W; Haplic Cambisol; 38% coarse sand, 38% fine sand, 15% silt and 9% clay) at a rate of 80 mg N kg^{-1} dry soil (equivalent to 240 kg N ha^{-1}) and an aerobic incubation was performed during 92 days at 25 ºC in 2 L kilner jars.

Table 1: Composition of the slurries used and amounts applied to soil (N = 3).

Parameters	Slurries			
	S	AS	LF	ALF
Slurry composition				
pH (H_2O)	7.2^a	5.5^b	7.2^a	5.5^b
Dry matter (g kg^{-1})	113^a	116^a	21^b	25^b
Total C (g C kg^{-1})	74.7^a	76.0^a	15.4^b	16.5^b
Total N (g kg^{-1})	3.8^a	3.3^b	2.5^c	2.2^c
NH_4^+-N (g N kg^{-1})	1.3^a	1.3^a	1.2^a	1.2^a
NO_3^--N (mg N kg^{-1})	0.1^b	1.6^{ab}	0.9^{ab}	2.1^a
Application rate				
mg C kg^{-1} dry soil	1570^b	1869^a	498^d	588^c

Within rows, values presented with different superscripts are significantly different (P < 0.05) by Tukey test.

Six treatments three times replicated were carried out: 1. soil only (Control); 2. surface application of S followed by soil incorporation (S-I); 3. band application of S (S-S); 4. band application of AS (AS-S); 5. band application of LF (LF-S); 6. band application of ALF (ALF-S).

Two independent incubations were performed to follow NH_3 and GHG emissions.

- NH_3 fluxes: Kilner jars (L = 100 mm, H = 210 mm) were filled with 0.75 kg dry soil (H = 70 mm) and treatments were then applied as above described. Water holding capacity was adjusted at 60% and controlled regularly. Ammonia fluxes were measured during the first 14 days of incubation using the acid traps, containing 50 mL of 0.05 M orthophosphoric acid. Acid traps in each jar were replaced after 5, 22, 29, 45, 69, 94, 166, 220 and 333 hours of the beginning of the experiment.

- GHG fluxes: a second independent set of Kilner jars was prepared to measure the GHG emissions in each treatment following the same procedure described for the measurement of NH_3 fluxes. Methane, N_2O and CO_2 were measured using the close chamber technique followed by quantification by gas chromatography, using a GC-2014 (Shimadzu, Japan). Gas measurements were carried out at days 1, 3, 4, 5, 8, then every two days up to day 36 and once a week up to day 92.

Cumulative emissions were estimated by averaging the flux between two sampling occasions and multiplying by the time interval between the measurements. More information about the experimental protocol can be obtained in [1].

3. Results and discussion

3.1 Nitrogen emissions

Significantly higher ($P < 0.05$) NH_3 emissions were observed in S-S treatment relative to all other amended treatments during all the measurement period (Figure 1). Incorporation of raw slurry led to a significant decrease of NH_3 emissions relative to S-S treatment but the lowest NH_3 emissions were observed in treatments amended with acidified slurry, LF or acidified LF. The cumulative NH_3 emissions from AS-S were significantly lower ($P < 0.05$) in 86% relative to S-S treatment (Figure 1). During the measurement period, the daily NH_3 fluxes from LF-S treatment were similar ($P > 0.05$) to those observed in Control. No differences ($P > 0.05$) were observed between ALF-S and LF-S treatment relative to NH_3 emissions indicating that LF acidification is not required since raw LF application already minimizes NH_3 emissions relative to raw slurry. As observed here, lower NH_3 emissions from LF amended soil than raw slurry were reported by other authors and they attributed this difference to a greater infiltration of LF in soil [2, 3]. Furthermore, a significant immobilization of NH_4^+ ions derived from LF might have occurred leading to a lower NH_3 volatilization [4]. As previously reported [5, 6], slurry acidification seems to have a strong potential to minimize NH_3 emissions after soil application.

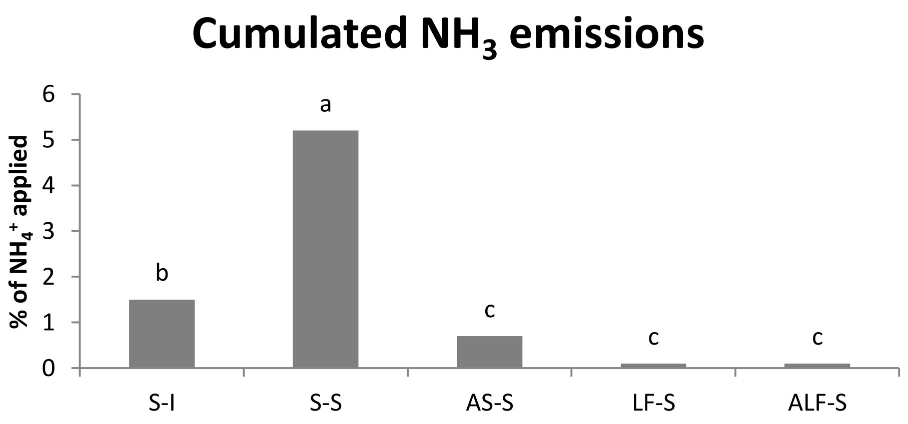

Figure 1: Mean value of the cumulated amount of ammonia emitted during the experiment ($N = 3$). Values quoted with different letters are significantly different ($P < 0.05$) by Tukey test.

Lower N_2O fluxes were observed in acidified treatments (AS-S and ALF-S) relative to non-acidified (S-S and LF-S, respectively) treatments. Significantly higher or similar N_2O emissions were observed in S-I treatments relative to S-S indicating that slurry incorporation might enhance N_2O emissions. The highest values of cumulative N_2O emissions were obtained in S-I, LF-S and ALF-S treatments (Figure 2). It is to refer that the total amount of N_2O released from ALF-S was significantly higher than in S-S and AS-S. Application of acidified slurry led to the lowest N_2O emissions, namely a significant ($P < 0.05$) decrease of N_2O emissions relative to S-I. This result is in line with a previous study [7] using acidified pig slurry and a delay or inhibition of the nitrification process might be the main reason for such decrease.

The N losses (NH_3 + N_2O) by gaseous emissions were significantly reduced by 59% in acidified cattle-slurry comparatively to non-acidified cattle-slurry. However, a combined treatment - separation + acidification - did not bring any benefit relative to NH_3 and N_2O emissions since similar results were observed in acidified LF and non-acidified LF. Soil application of acidified cattle-slurry rather than non-acidified LF might be motivated by the lower costs associated to acidification compared to solid-liquid separation.

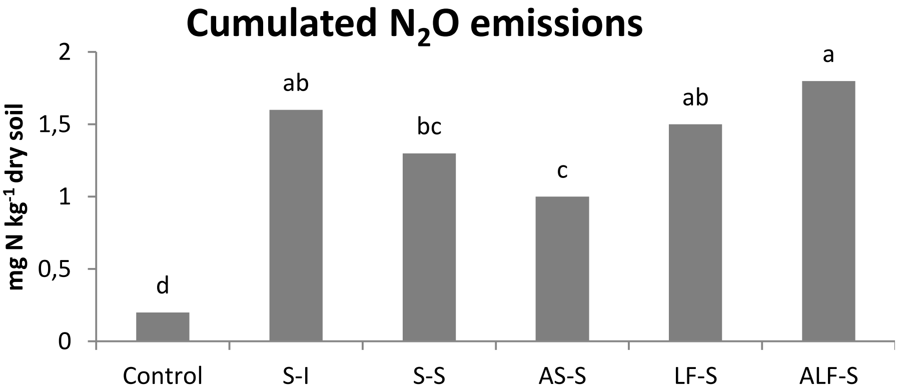

Figure 2. Mean value of the cumulated amount of nitrous oxide emitted during the experiment (*N* = 3). Values quoted with different letters are significantly different (*P* < 0.05) by Tukey test.

3.2 Carbon emissions

Higher CO_2 fluxes were observed over the first 8 days of the experiment in all treatments, with 26 and 45% of the total CO_2 emissions occurring during this period, followed by a reduction until the end of the measurements. Cumulative CO_2 emissions were lower in 32% when S-S was applied relative to LF-S (Figure 3). Also, the cumulative CO_2 emissions were lower ($P < 0.05$) in 52% when ALF-S was applied relative to LF-S amended treatment (Figure 3). More than 60% of the applied C was released as CO_2 in the LF-S treatment whereas in all the other amended treatments, less than 35% of the applied C was released.

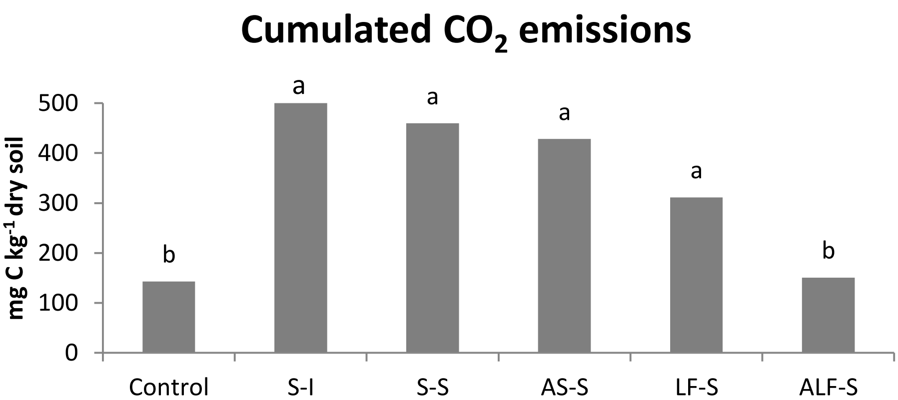

Figure 3. Mean value of the cumulated amount of carbon dioxide emitted during the experiment (*N* = 3). Values quoted with different letters are significantly different (*P* < 0.05) by Tukey test.

Cattle-slurry application increases the soil microbial activity and consequently CO_2 emissions after amendment due the organic matter decomposition. Compared with non-acidified slurries, slurry acidification reduced CO_2 emissions because it decreases microbial activity and consequently oxygen consumption, sulphate reduction and methanogenesis [8].

Methane emissions were only observed in S-I and S-S treatments during the first 4 days following soil application, being in agreement with previous studies [9] who reported that CH_4 emissions occurred during a short period. Furthermore, it is generally assumed that most of CH_4 released from slurry amended soil is due to the volatilisation of the CH_4 dissolved in the cattle-slurry (produced during storage). It has been seen that slurry acidification reduces the methanogenesis and consequently CH_4 production and emissions during slurry storage [6], what justifies the lower CH_4 emissions observed in AS-S relative to S-S. Most of the dissolved CH_4 initially present in S and AS should have been released during the separation process explaining the absence of CH_4 emissions in LF-S or ALF-S treatments.

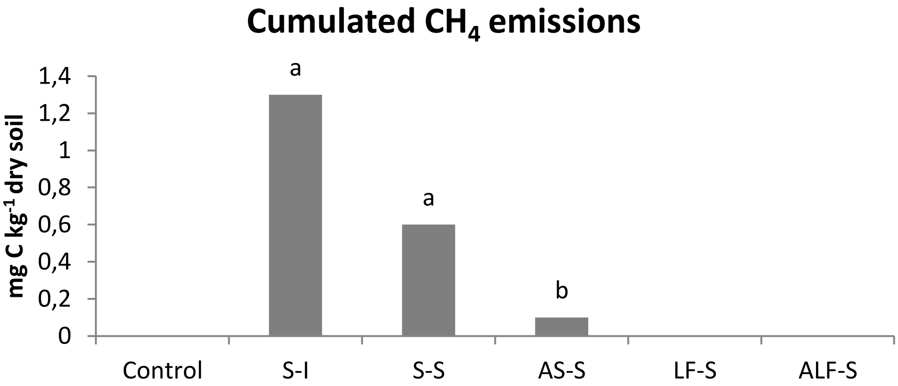

Figure 3. Mean value of the cumulated amount of methane emitted during the experiment (N = 3). Values quoted with different letters are significantly different (P < 0.05) by Tukey test

4. Conclusions

Our results showed that slurry acidification is a good solution to minimize NH_3, N_2O and CH_4 emissions following soil application. Furthermore, LF application rather than raw slurry led to a significant decrease of NH_3 emissions but has no impact on N_2O emissions. However, acidification of LF has no positive impact on gaseous emissions. Further studies at farm scale are required to validate our results and, on the other hand, a more safe and cost- effective solution for slurry acidification needs to be developed.

Acknowledgements

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References

[1] Fangueiro, D., Pereira, J., Bichana, A., Surgy, S., Cabral, F., Coutinho, J. (2015): Effects of cattle-slurry treatment by acidification and separation on nitrogen dynamics and global warming potential after surface application to an acidic soil. Journal of Environmental Management, 162, 1-8.

[2] Sommer, S.G., Hansen, M.N., Søgaard, H.T. (2004): Infiltration of slurry and ammonia volatilisation. Biosystems. Engineering 88, 359-367.

[3] Ndegwa, P.M., Hristov, A.N., Arogo, J., Sheffield, R.E., (2008): A review of ammonia emission mitigation techniques for concentrated animal feeding operations. Biosystems Engineering 100, 453-469

[4] Sanz-Cobena, A., Misselbrook, T., Camp, V., Vallejo, A. (2011): Effect of water addition and the urease inhibitor NBPT on the abatement of ammonia emission from surface applied urea. Atmospheric Environment 45, 1517-1524.

[5] Kai, P., Pedersen, P., Jensen, J.E., Hansen, M.N., Sommer, S.G., (2008): A whole-farm assessment of the efficacy of slurry acidification in reducing ammonia emission. European Journal of Agronomy 28, 148-154.

[6] Fangueiro, D.; Hjorth, M.; Gioelli, F. (2015): Acidification of animal slurry - a review. Journal of Environmental Management 149, 46-56

[7] Fangueiro, D., Ribeiro, H., Coutinho, J., Cardenas, L., Trindade, H., Cunha-Queda, C., Vasconcelos, E., Cabral, F., (2010): Nitrogen mineralization and CO2 and N2O emissions in a sandy soil amended with original or acidified pig slurries or with the relative fractions. Biology and Fertility of Soils 46, 383-391.

[8] Ottosen, L.D.M., Poulsen, H.V., Nielsen, D.A., Finster, K., Nielsen, L.P., Revsbech, N.P. (2009): Observations on microbial activity in acidified pig slurry. Biosystems Engineering 102, 291-297.

[9] Pereira, J.; Fangueiro, D.; Chadwick, D.R.; Misselbrook, T.H.; Coutinho, J.; Trindade, H. (2010): Effect of cattle slurry pre-treatment by separation and addition of nitrification inhibitors on gaseous emissions and N dynamics: A laboratory study. Chemosphere 79, 620-627.

TC-P_06 Greenhouse gas emissions and crop yields under different organic fertilizers and irrigation treatments in a Mediterranean maize field

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1. Objectives

In this experiment, we aimed to assess the effect of:

- Partial substitution of urea (U) by organic amendments at seeding: pig urine (PU), pig urine with the nitrification inhibitor 3,4 dimethylpyrazolephosphate (DMPP, PUI), compost from the solid phase of pig slurry (PCOM);
- Two different irrigation systems: sprinkler and drip;

on greenhouse gas (GHG) emissions, crop yield and yield-scaled N_2O emissions (YSNE) in a maize (*Zea mays* L.) crop.

2. Methodology

The study was carried out at the "El Encín" field station (Madrid, Spain). All the irrigation and fertilizer treatments were assigned in a three-replicated completely randomized design. All plots (except the control) were fertilized with 180 kg N ha^{-1} (120 kg N ha^{-1} as organic fertilizers at seeding – PCOM, PU, PUI- and 60 kg N ha^{-1} as urea at dressing, one month after seeding). In the case of urea, the entire N rate (180 kg N ha^{-1}) was applied at dressing (broadcasted on surface in sprinkler plots and by fertigation on drip irrigation plots). A control (C) with no N fertilization was also included. Maize was seeded on 7th May 2014 (resulting in a plant population of 7.5 plants m^{-2}) and harvested on 24th October 2014. Irrigation was applied in drip plots by a surface drip irrigation (emitters with nominal discharge of 4 L h^{-1}, 0.33 m apart, 48 irrigation events) and by an installed 12m x 12m sprinkler irrigation system at a height of 2.5 m, in the case of sprinkler plots (32 irrigation events). A total amount of 688 L m^{-2} of water was applied in both systems.

Greenhouse gas emissions were sampled by the static closed chamber method, and quantified by gas chromatography. Gas samples were taken twice per week during the first month after fertilization events. Afterward, gas sampling was performed weekly or every two weeks, until the end of the crop period. To minimize any effects of diurnal variation in emissions, samples were taken at the same time of day (10am –12 pm). Yield-scaled N_2O emissions were calculated by dividing total N_2O emission by N uptake in aboveground biomass [1].

3. Results and discussion

3.1 Greenhouse gas emissions

3.1.1 Nitrous oxide emissions

Nitrous oxide emissions ranged from -0.1 to 9.8 mg N_2O-N m^{-2} d^{-1} (U, 24th June) (Fig. 1a, b). Emission peaks were observed after fertilization, with N_2O fluxes subsequently decreasing until the end of the experiment. Cumulative fluxes were significantly reduced when U was partially substituted by organic amendments in sprinkler irrigation plots, but not when U was applied by fertigation. Considering all N-fertilized treatments, only PUI significantly decreased cumulative N_2O losses (60 % abatement, P<0.001) when compared to U. Even though the effect of application of pig urine with DMPP has been rarely tested, these results are consistent with the field studies of [2] or [3], who found an effective mitigation of N losses from cattle urine when it was applied with the nitrification inhibitor dicyandiamide (DCD). The use of DMPP instead of DCD could be considered as a more advisable alternative under high-temperature conditions, in which DCD has shown high instability [4].
Drip irrigation significantly reduced N_2O cumulative losses by 63% relative to sprinkler irrigation (P<0.001). The use of high water efficiency irrigation systems (e.g. surface drip irrigation) has

shown a potential for mitigating N_2O losses in previous studies as a result of lower denitrification rates [5, 6].

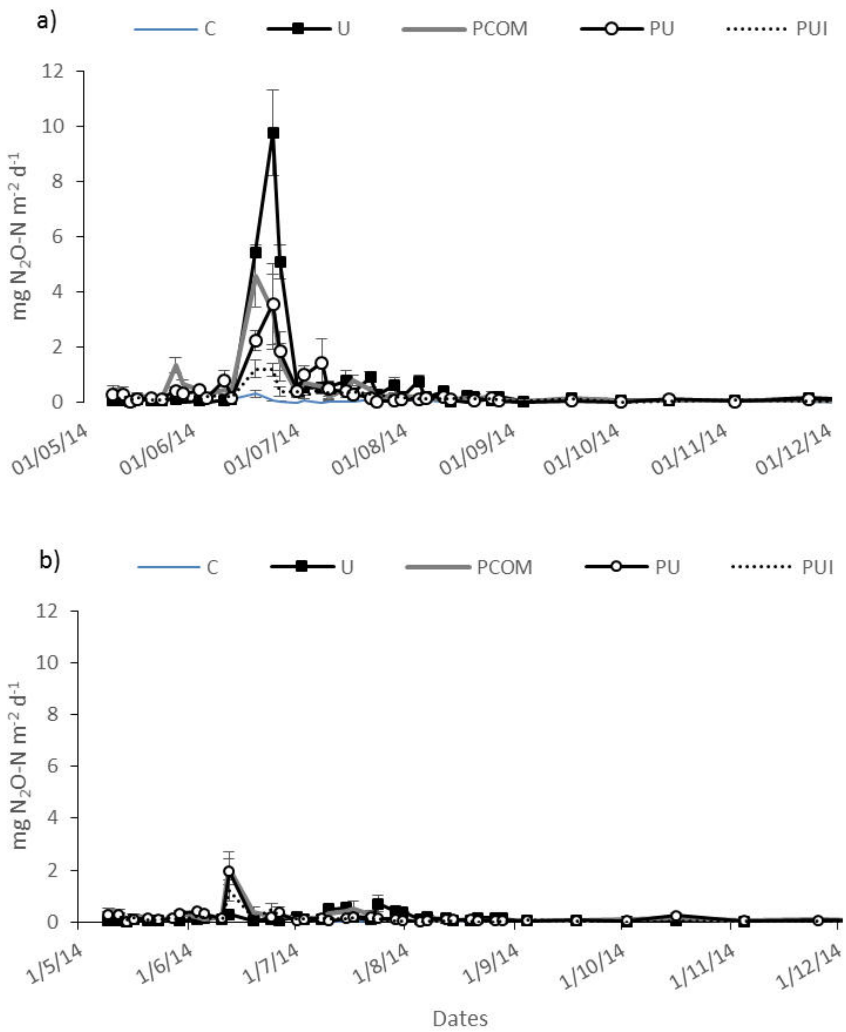

Figure 1: Nitrous oxide (N_2O) fluxes throughout maize cropping cycle in **a** sprinkler-irrigated plots and **b** drip-irrigated plots in the different fertilizer treatments (control, C, urea, U, compost, PCOM, pig urine, PU, pig urine with the nitrific tion inhibitor DMPP, PUI). Vertical bars indicate the standard errors of the mean.

3.1.2 Methane and carbon dioxide emissions

All treatments were net CH_4 sinks (Fig 2a). Uptake capacity was significantly increased in U and PUI treatments, and under sprinkler irrigation (*P*<0.001). These results showed that in spite of higher NH_4^+-N availability expected in U and PUI, no inhibitory effect of CH_4 oxidation was observed. The competitive inhibition of the enzyme responsible for CH_4 oxidation (CH_4 monooxygenase) with the NH_3 monooxygenase has been previously reported [7, 8]. Conversely, a positive correlation between soil NH_4^+ and CH_4 oxidation was observed by [9]. Regarding irrigation effect, soil methanotrophic activity increases to a value close to field capacity, then decreases when the water content increases [8]. This may explain why CH_4 uptake is higher in sprinkler irrigation plots. Carbon dioxide cumulative fluxes (Fig. 2b) were increased in sprinkler when compared to drip irrigation system (*P*<0.001). As expected, the fertilizer treatment that provided a C source (PCOM) led to greater CO_2 fluxes than the rest of treatments (*P*<0.001).

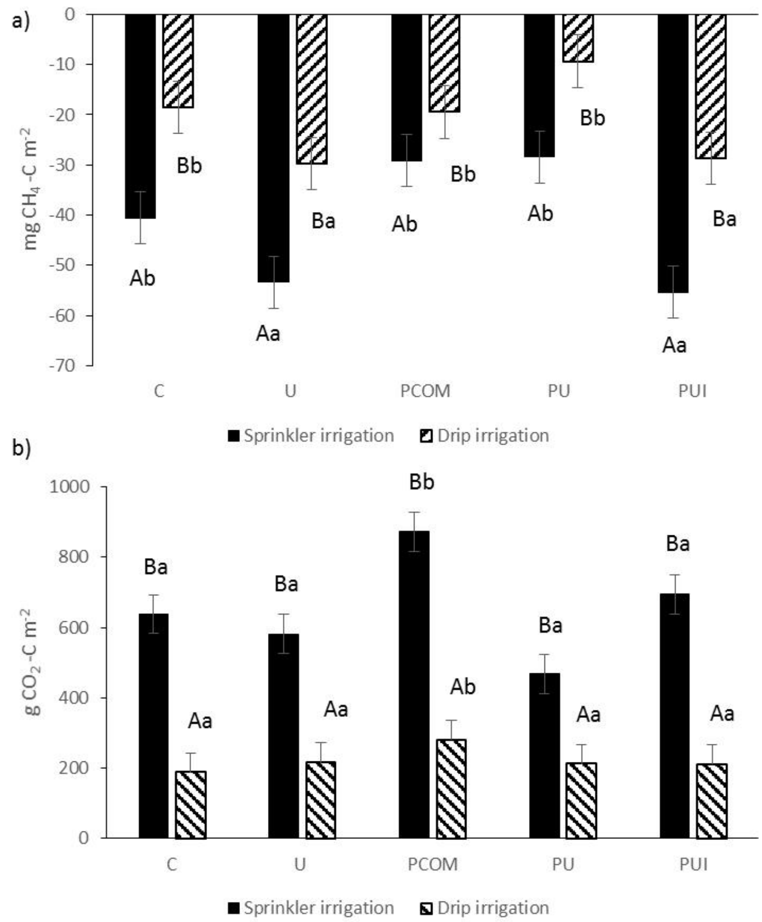

Figure 2: Cumulative **a** methane (CH_4) and **b** carbon dioxide (CO_2) fluxes in the different fertilizer (control, C, urea, U, compost, PCOM, pig urine, PU, pig urine with the nitrification inhibitor DMPP, PUI) and irrigation treatments. Different uppercase letters within columns indicate significant differences in the irrigation factor, while different lowercase letters within columns indicate significant differences in the fertilizer factor, by applying the Tukey's honest significance test at P<0.05. Vertical bars indicate the standard errors of the mean.

3.2 Maize yields and Yield-scaled N2O emissions

Considering all fertilized treatments, only PUI did not decrease grain yield when compared to U. By contrast, PCOM and PU decreased grain yield by 33 and 26%, respectively (P<0.001). Urea also resulted in the highest biomass production, which was 21, 34 and 31% greater than that of PUI, PU and PCOM, respectively. These results emphasize the need to accurately calculate the available N for the crop when organic amendments (e.g. compost) are applied. The use of DMPP with pig urine increased grain (15%, 0.05<P<0.10) and biomass yields (21%, P<0.05) compared to urine alone, which is consistent with the meta-analysis of [10]. The irrigation system did not affect grain yields, even though biomass production was 32% higher (P<0.001) in drip-irrigated plots.

The N_2O efficiency of a cropping system, in a context of increasing food demand, should be expressed in terms of YSNE [1] (Fig. 3). This index decreased when DMPP was used with pig urine compared to urine alone, while PCOM resulted in the highest N_2O-N emitted per kg of N uptaken. Higher biomass production and lower N_2O emission in the drip irrigation system contributed to significantly abate (P<0.001) the YSNE ratio in these plots. Partial replacement of urea by PUI could be considered as a sustainable management practice in terms of the abatement of GHG losses and YSNE, leading to crop yields potentially acceptable to farmers. In this sense, PCOM or PU did not result in an optimum mitigation-adaptation balance. The employment of an improved irrigation system (i.e. surface drip) showed a high effectiveness in the mitigation of N_2O losses and YSNE, without affecting grain yield and even enhancing biomass production. In spite of the lower CH_4 oxidation potential, this management practice resulted in a lower global warming potential than the conventional sprinkler irrigation system.

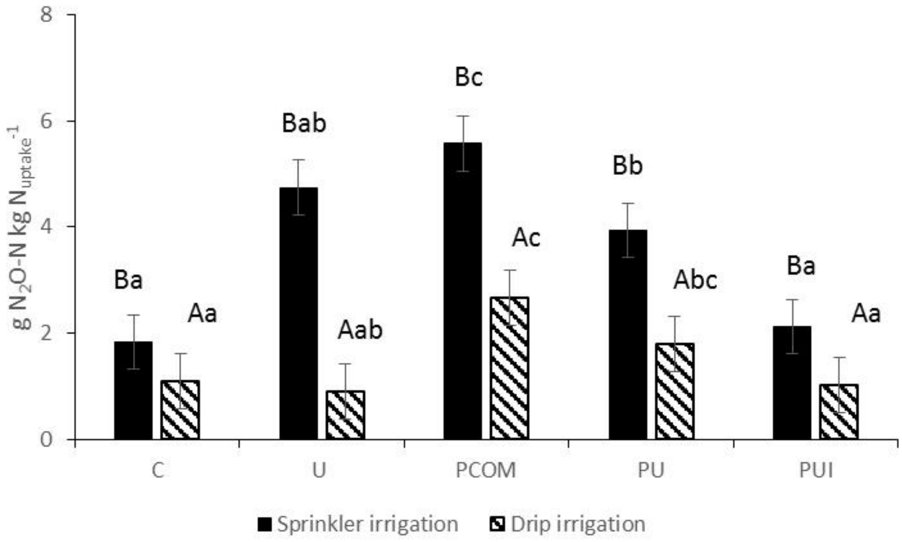

Figure 3: Yield-scaled N_2O emissions of the different fertilizer (control, C, urea, U, compost, PCOM, pig urine, PU, pig urine with the nitrification inhibitor DMPP, PUI) and irrigation treatments. Different uppercase letters within columns indicate significant differences in the irrigation factor, while different lowercase letters within columns indicate significant differences in the fertilizer factor by applying the Tukey's honest significance test at $P<0.05$. Vertical bars indicate the standard errors of the mean.

4. Conclusion and outlook

The partial replacement of urea by organic N sources is an environmentally advisable strategy, but an optimum balance between mitigation of GHG losses and adaptation of an irrigated maize crop in Mediterranean areas requires the use of nitrification inhibitors (e.g. DMPP) with liquid organic sources. Drip irrigation system also showed promising results in terms of abating yield-scaled N_2O emissions and global warming potential, in spite of the risk of decreasing CH_4 uptake.

Acknowledgements

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References

[1] Van Groenigen, J.W.; Velthof, G.L.; Oenema, O.; Van Groenigen, K.J.; Van Kessel, C. (2010): Towards an agronomic assessment of N2O emissions: a case study for arable crops. European Journal of Soil Science, 61(6), 903-913.

[2] Di, H.J.; Cameron, K.C. (2008): Sources of nitrous oxide from 15N-labelled animal urine and urea fertiliser with and without a nitrification inhibitor, dicyandiamide (DCD). Soil Research, 46(1), 76-82.

[3] Zaman, M.; Nguyen, M.L. (2012): How application timings of urease and nitrification inhibitors affect N losses from urine patches in pastoral system. Agriculture, Ecosystems & Environment, 156, 37-48.

[4] Mahmood, T.; Ali, R.; Latif, Z.; Ishaque, W. (2011): Dicyandiamide increases the fertilizer N loss from an alkaline calcareous soil treated with 15N-labelled urea under warm climate and under different crops. Biology and Fertility of Soils, 47(6), 619-631.

[5] Sanchez-Martín, L.; Meijide, A.; Garcia-Torres, L.; Vallejo, A. (2010); Combination of drip irrigation and organic fertilizer for mitigating emissions of nitrogen oxides in semiarid climate. Agriculture, ecosystems & environment, 137(1), 99-107.

[6] Aguilera, E.; Lassaletta, L.; Sanz-Cobena, A.; Garnier, J.; Vallejo, A. (2013): The potential of organic fertilizers and water management to reduce N_2O emissions in Mediterranean climate cropping systems. A review. Agriculture, Ecosystems & Environment, 164, 32-52.

[7] Dunfield, P.; Knowles, R. (1995): Kinetics of inhibition of methane oxidation by nitrate, nitrite, and ammonium in a humisol. Applied and Environmental Microbiology, 61(8), 3129-3135.

[8] Le Mer, J.; Roger, P. (2001): Production, oxidation, emission and consumption of methane by soils: a review. European Journal of Soil Biology, 37(1), 25-50.

[8] Bodelier, P.L.E.; Landbroek, H.J. (2004): Nitrogen as a regulatory factor of methane oxidation in soils and sediments. FEMS Microbial Ecology 47, 265e277.

[10] Abalos, D.; Jeffery, S.; Sanz-Cobena, A.; Guardia, G.; Vallejo, A. (2014): Meta-analysis of the effect of urease and nitrification inhibitors on crop productivity and nitrogen use efficiency. Agriculture, Ecosystems & Environment, 189, 136-144.

TC-P_07 OptiBarn – Optimized animal specific barn climatisation facing temperature rise and increased climate variability

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1. Objectives

OptiBarn is granted in the framework of the FACCE-ERANET+ initiative “Climate Smart Agriculture” and investigates region-specific, sustainable adaptation strategies for dairy housing, focusing on an optimised climatisation of naturally ventilated buildings (NVB) in order to adapt to climate change. Appropriate construction methods and management of the buildings can improve thermal control for increased productivity, improved animal welfare and reduced emissions.

2. Project and methods description

OptiBarn runs from December 2014 to September 2017. Scientists from four countries, six institutes and various disciplines are involved. Companies and stakeholders support the project. Indicators for an optimisation of barn climate in NVB under climate change are developed by:

- Barn-specific assessment of the influence of outdoor climate on the indoor conditions: Numeric simulations, physical modelling in a boundary layer wind tunnel and field studies in selected barns are conducted to assess the effects of boundary conditions on the climate in NVB.
- Region-specific risk analysis on how often extreme weather situations will occur in the future: Data from weather stations and climate simulations are analysed with regard to heat stress.
- Monitoring animal-individual stress responses to the indoor conditions: Field studies and experiments under lab conditions are conducted to propose a sophisticated heat stress scale.
- Development of engineering solutions: New building configurations, ventilation system designs and indoor climate control strategies are investigated.
- Modelling tools to assess environmental and economic effects of adaptation alternatives: The effects of different building and management factors on the thermal control capacity of dairy buildings are quantified and the overall risks are highlighted.

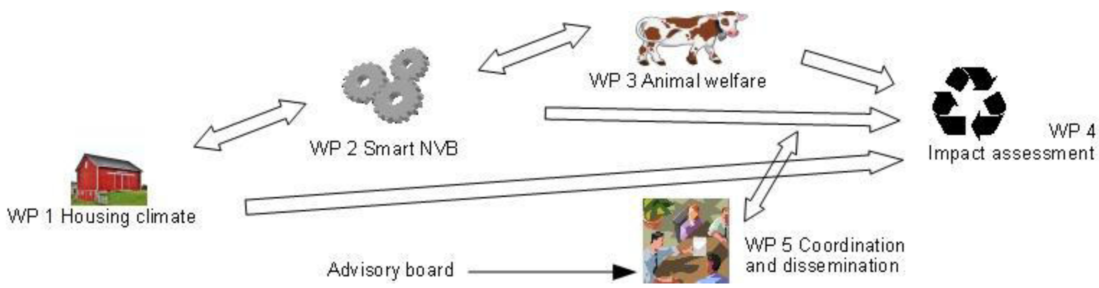

Figure 1: Project layout with five interacting work packages (WP) and external advirory board.

- The paper informs on the early state of the project and on-going tasks in the five work packages (cf. Figure 1). Preliminary results of a first case study in WP 1 related to aims 1 and 5 are shown: 6 consecutive days in August 2014 with 10 minutes temporal resolution and four sampling points (2.7m height) in a barn in Northern Germany. The following analysis techniques are used:Spatial anomaly: Spatial averages $\bar{x}$ over all sampling points are calculated. Anomalies are defined per sampling point and time step as deviations from the spatial averages $A = (x - \bar{x})$.

- Temperature-Humidity-Index: We used the definition according to CIGR glossary 2009 [1]: $= 0.8 * T + (0.01 * H * (T - 14.3)) + 46.4$.
  Here T is the air temperature in degree C and H the relative humidity in percentage.
- Correlation function: We calculated the lagged Pearson correlation $\rho = cor(x(t), y(t + d))$ between time series *x(t)* of the first variable at times t_i to t_f and time series *y(t+d)* of the second variable at times t_i+*d* to t_f+*d*. The delay d was varied from 10 minutes to two days. The correlation for each delay is plotted in dependence of the delay to obtain the correlation function.

3. Results and discussion

3.1 Status of the project

OptiBarn was launched with a kick-off workshop on 1st and 2nd of December 2014. Work packages 1, 2 and 5 started immediately, 3 and 4 recently. The following tasks are completed or on-going:
WP 1: Five barns were selected for case studies in Germany, Spain and Israel. Aggregation of information on barn construction, farm design, management and typical climatic boundary conditions is ongoing. Long term observations of certain key variables (e.g., minimum and maximum daily temperature or wind speed) for the selected reference barns were investigated to identify relevant extreme outdoor weather conditions or episodes and to create an outdoor climate reference dataset for the calibration of the climate models. High resolution climate simulations were collected and are currently complemented to build an ensemble of anticipated future climate conditions. The development of a computational fluid dynamics (CFD) model that considers barn geometry, animals and climate input is ongoing in collaboration with WP2. Wind tunnel experiments are scheduled to improve measurement techniques and experimental plans.
WP 2: First CFD simulations to investigate the potential of using passive earth-air heat exchange ventilation were conducted. Ongoing experiments in a climate chamber consider the convection heat removal in relation to airflow speed and temperature for developing an integrated sensing method for optimal design and control of livestock housing ventilation. Later, with input from WP 3, animal wellbeing shall be detected under complex thermal conditions affected by multi-thermal parameters. A setup to investigate precision zone ventilation configurations for an optimal thermal environment in the Animal Occupied Zone (AOZ) and to remove most pollutant air effectively near to the emission sources for cleaning treatments is currently developed.
WP 3: First experiments were conducted to select sensors (physiological or behavioural) for stress in dairy cows on hot days. A setup to investigate the impact of the dynamics of barn climate and physiological parameter (e.g., age) on the cow individual stress level is currently developed.
WP 4: Collection of farm management data for the case study barns started.
WP 5: The basic information and data exchange structure was established. A project homepage www.optibarn.eu provides general information and will later inform on key results.

3.2 First case study on climate parameters and gas concentrations

First studies indicate that even for symmetric buildings the air flow inside the barn is spatially heterogeneous and non-stationary [2]. Similar temporal and spatial fluctuations are found for temperature, humidity and gas concentration [3]. The time series of the spatial anomalies for wind are shown in Figure 2 as an example.

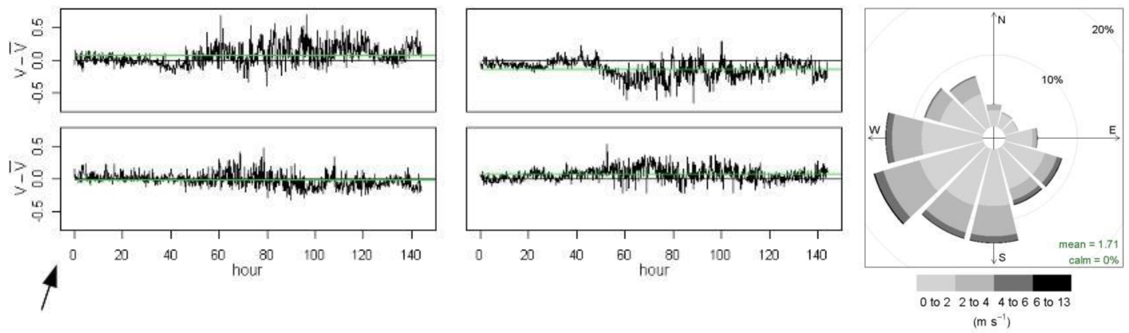

Figure 2: Left: Deviation of air speed in m/s from the spatial average over 6 days at four measurement points. The green line indicates the temporal average of the spatial anomaly. The dominant direction of the incoming wind (S-SW) is indicated by the arrow. Right: A wind rose of the external wind speed and direction for the measurement period.

Figure 3 shows the spatial anomalies for ammonia, methane, relative humidity and temperature exemplarily for the front left point, while Figure 4 summarises the normalised average anomalies for all variables at the four measurement points. The front left part of the barn is characterised by medium air speed, humidity and ammonia values, low temperature and high methane values. The back left part shows high air speed, low temperature, humidity and methane and medium ammonia concentration. The back right part is characterised by low air speed, methane and ammonia concentration, high temperature and medium humidity. In the front right part all variables are significantly higher than the corresponding spatial average.

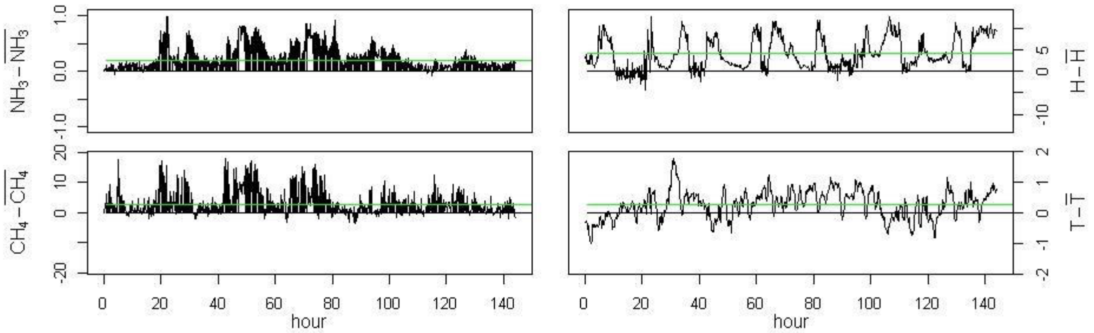

Figure 3: Analogous to Figure 2 time series of ammonia in ppm, methane in ppm, relative humidity in % and temperature in K for the front left measurement point.

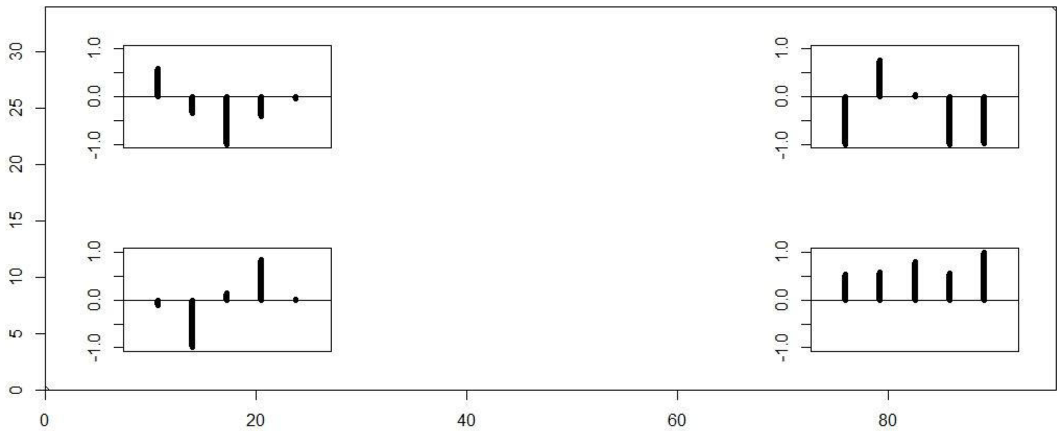

Figure 4: The subplots show a normalised average spatial anomaly for the variables in the following order: wind speed, temperature, relative humidity, methane and ammonia. Normalisation is done dividing by the maximal value for the variable over the four points. The positions of the subplots indicate the measurement points in a plan of the barn (dimensions: 96m x 34m).

Amplitudes and signs of the spatial anomalies vary over time as shown in Figures 2 and 3. The recorded variations in the indoor climate parameters depend on the size and position of openings and on the outdoor climate, particularly on the wind (cf. previous studies [3, 4]). Spatial variations

are likely induced by the turbulence of the air flow. However, the dynamics of single climate variables follow those of the air flow with different shaping and delay. A possible reason is the impact of the animals, which are sources of temperature, humidity, small-scale turbulence and emissions, and affect the barn climate in various ways and at various time scales.
The strong interaction between animals and barn climate suggests that there is a functional relation between air flow, heat stress and emission rates. In order to investigate this implication we consider the correlation between wind speed, gas concentrations and temperature-humidity indices (THI) as a measure of heat stress. As an example, we use the front, right measurement point which is characterised by particularly high values for all variables. Simultaneous measurements of temperature, humidity, wind and gas concentrations show a low instantaneous correlation (i.e., Pearson correlation for lag zero in Figure 5). All values obtained are negative. These results confirm that high temperature and humidity inside the barn usually come along with low wind speed. Moreover, low wind speed results in an increase of gas concentrations since gas exchange is reduced. However, the instantaneous correlation also suggests that first there is a decrease of gas concentrations if temperature and humidity values increase which is counterintuitive. A possible explanation may be the speed-up of chemical decomposition processes.
In order to further assess the relations between the different variables, subsets of the time series (time frames) at the same sampling point were considered to calculate the correlation function (Figure 5). If we take into account theses delays, the revealed functional relations differ from those which the instantaneous correlation suggests. The correlation values are significantly higher than for the instantaneous correlation.

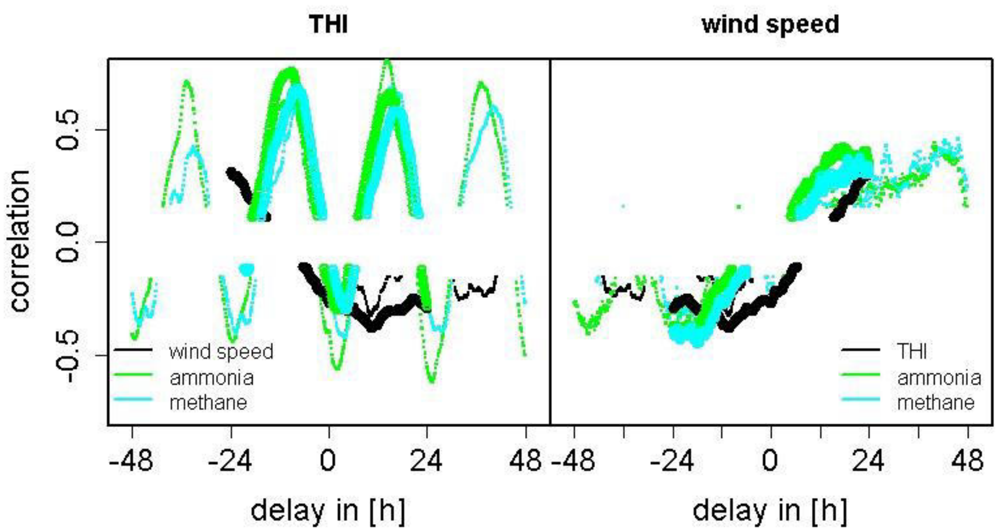

Figure 5: Correlation function for temperature-humidity-index THI with wind and gas concentrations (left) and wind w THI and gas concentrations (right). The THI (cf. section 2) is defined according to CIGR report 2006 [1].

For the correlation between THI and gas concentrations we found a pronounced daily cycle, while for wind the correlation function oscillates with lower frequency. Probably this is related to the fact that temperature, humidity and gas concentrations have a daily cycle while wind does not have a pronounced periodicity at the time scales considered here. The maximal positive correlation between gas concentrations and climate parameters is always obtained with about 12 hours delay. This is likely a result of the interplay of different dominant frequencies (e.g., day-night rhythm or milking times) which determine the dynamics of the barn climate variables.

4. Conclusion and outlook

In this study we found, for the selected boundary conditions, a particular region in the barn (front right) with high values of all considered barn climate variables. The dynamics of temperature, humidity and gas concentrations follow that of the air flow on average with about half a day time delay, probably caused by the dynamics of air exchange and by the responses of the animals.
The spatial and temporal dynamics of barn climate variables highlight the necessity to develop smart ventilation concepts for naturally ventilated barns to reduce emissions and animal stress. An in-depth study on the effects of outdoor conditions and building design on the barn climate and its interaction with the housed animals is essential. In OptiBarn functional relations and typical time delays in the interactions (response times), general spatial and temporal pattern in the barn climate and animal individual stress reactions are investigated. The interdisciplinary approach and

continuous validation under commercial conditions are crucial for a holistic view on the interaction between barn climate, animals and environment in dairy husbandry systems.

Acknowledgements

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References

[1] Panagakis, P., Blanes-Vidal, V., Banhazi, T., Barbosa, J. C., Maltz, E., Da Cruz, V., & Berckmans, D. (2009). Glossary of terms on animal housing: interconnecting engineering, physical and physiological definitions. *Agricultural Engineering International: CIGR Journal*.

[2] Fiedler, A.; Fischer, J.; Hempel, S.; Saha, C.; Loebsin, C.; Berg, W.; Amon, B.; Brunsch, R.; Amon, T.(2014): Flow fields within a dairy barn - Measurements, physical modelling and numerical simulation. In: Proceedings. International Conference of Agricultural Engineering AgEng 2014. Zürich, p. 1-5

[3] Saha, C. K.; Ammon, C.; Berg, W.; Loebsin, C.; Fiedler, M.; Brunsch, R.; von Bobrutzki, K. (2013): The effect of external wind speed and direction on sampling point concentrations, air change rate and emissions from a naturally ventilated dairy building. Biosystems Engineering, 114(3), 267-278

[4] Saha, C.; Fiedler, A.; Amon, T.; Berg, W.; Amon, B.; Brunsch, R.(2014): Assessing effects of different opening combinations on airflow pattern and air exchange rate of a naturally ventilated dairy building - A CFD approach. In: Proceedings. International Conference of Agricultural Engineering AgEng 2014. Zürich, p. 1-4

TC-P_08 Solid manure and its liquid fraction – Quantities and nutrient contents derived from balancing models

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1. Objectives

Mass amounts and contents of solid manure and its liquid fraction are important for nutrient balances and emission inventories and for planning of stables, logistics and storage space capacities. These farm fertilizers are usually inhomogeneous so that accurate determination of its mass and contents is complex. An alternative to measuring or thumb numbers are balancing models based on practical animal feed and performance data.

2. Methodology

Focus of the proposed mass balance computational model is the separation of mass amounts in faeces and urine (Fig. 1). For mammals, the excretions faeces and urine must be determined separately, taking into account the amount of bedding material and their absorption potential for urine as well as rotting losses (OM) and gaseous losses (NH_3). The amounts of seepage from stable manure and usual operational water inflows to the liquid fraction are also considered. For calculating the mass amount of faeces and urine the digestibility coefficients of the energy containing organic raw nutrients (crude protein, crude fat, crude fiber, nitrogen free substances) are required. These figures are derived from feed value tables. Digestibility coefficients are components of the energy calculation of a feed calculation and are determined sufficiently exact in a variety of feeding trials or can be derived by estimating equations. For poultry, faeces and urine are excreted together. The amounts of poultry manure are calculated with the digestibility of organic matter in the feed and the excreted crude ash (ash in the feed minus ash in animal product) taking into account the amount of bedding material as well as rotting and gaseous losses.

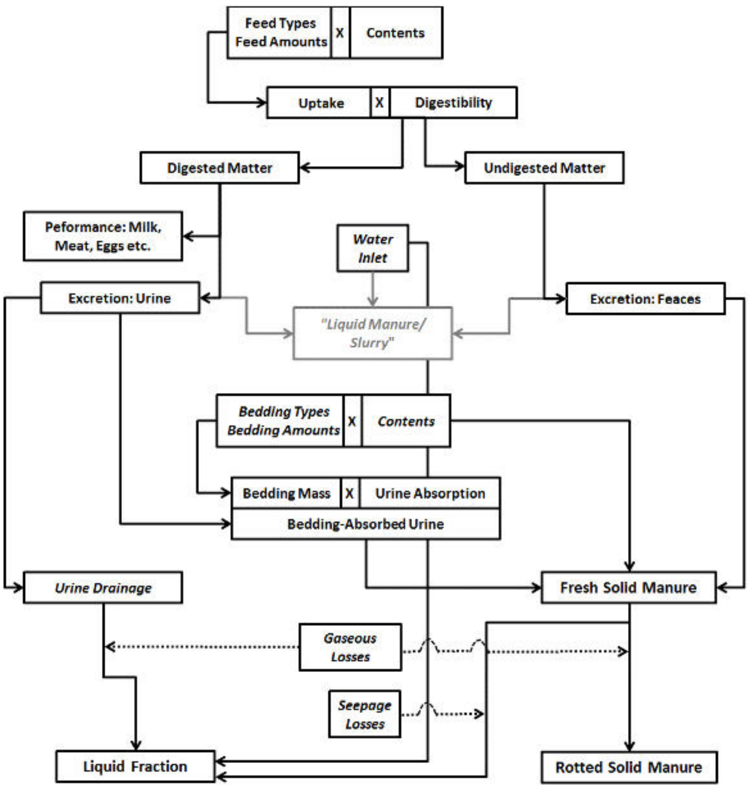

Figure 1: Flow chart of the model for determining the mass amount of solid manure, its liquid fraction and mineral contents

For different animal categories and production methods the calculations are performed for solid manure and its liquid fraction, respectively. Based on animal performance and feeding data the mass amount of rotted solid manure and its liquid fraction are derived. The amount of nutrients per animal place and unit of time and the content values per mass are calculated. The calculated quantities and contents are expected average values. Water additives e.g. milking parlour water, courtyard drain must be added as unaccounted amounts to the liquid fraction.

3. Results and discussion

Key factors in the calculation model are the assumed amounts of feed, their nutrient content and digestibility (Table 1a, Table 1b) that meet the requirements for the assumed animal performance. In the following chapters (3.1, 3.2) examples of the results are given.

Table 1a: Factors influencing the mass amount of solid manure, its liquid fraction and nutrient contents: feed and bedding

Influence variable	Characteristic parameter
feed type, feed amount (e.g. kg DM/(animal · a) contents in feed (e.g. kg N/kg DM) digestibility of the feed (dimensionless); performance; nutrient contents in animal products (e.g. kg ECM/(dairy cow·a); kg N/kg FM)	amount of faeces and urine (e.g.kg FM/a) e.g. nitrogen content (e.g. kg N/kg FM) nutrient excretion (e.g. kg faeces FM/a)
bedding depending on animal category and production system nutrient contents in bedding material (straw) (e.g. kg N/kg FM)	amount of bedding material (e.g. kg FM/d) nutrient contents in solid manure (e.g. kg N/kg FM)
bedding absorption of urine (straw) (kg urine/kg FM); 1,8 kg urine/kg bedding fresh mass (straw)	amount of liquid fraction (e.g. kg FM/a) nutrient contents in solid manure

Table 1b: Factors influencing the mass amount of solid manure, its liquid fraction and nutrient contents: storage

	Influence parameters during storage								
	Loss ratios (seepage + rotting) during storage (dimensionless parameters)								
	Fresh mass	**Crude ash**	**Organic matter**	**N**				**P**	**K**
	Seepage losses								
Fresh solid manure	0.15	0.06	0.02	0.10				0.02	0.24
Deep litter solid manure	-	-	-	-				-	-
	Rotting losses (gaseous)								
				Cattle	Pigs	Poultry	Horses, sheeps, goats		
Fresh solid manure	-	-	0.20	0.30	0.35	0.40	0.45	-	-
Deep litter solid manure	-	-	0.12	0.30	0.35	-	0.45	-	-
Liquid fraction	-	-	-	0.30	0.35	-	0.45	-	-

3.1 Rearing young cattle

Table 2 shows the calculated mass amounts of solid manure, its liquid fraction and their contents for heifer husbandry. For grassland farms, a differentiation according to acreage and fertilization intensity (intensively and extensively) was calculated (results not shown). For extensively production feed from "conservation areas" with lower crude protein and und energy contents was assumed. The mass amounts of the different rearing sections are yielded by multiplying the values of the overall process (Table 2) with the proportional factors from Table 3.

Table 2: Mass amount of solid manure and its liquid fraction for rearing young cattle by grassland farms (first calving at 27 months)

Straw	**Type of manure**	**Mass amounts**						**Contents**		
kg FM/		**FM**	**DM**	**OM**	**N**	**P**	**K**	**N**	**P**	**K**
(animal·d)				**kg/animal**					**g/kg FM**	
	Faeces	8805	1585	1255	40.7	14.1	24.0	4.6	1.6	2.7
	Urine	10085	202	-	87.9	2.0	108.2	8.7	0.2	10.7
3.0	Fresh solid manure	15693	3838	3279	91.9	18.3	101.8	5.9	1.2	6.5
3.0	Rotted solid manure	12333	3083	2558	57.9	17.9	77.4	4.7	1.5	6.3
3.0	Liquid fraction	9651	212	66	41.0	1.5	85.1	4.2	0.2	8.8
6.8 [1)]	Fresh solid manure	24493	6717	5865	157.2	23.5	201.2	6.4	1.0	8.2
	Rotted solid manure	24053	6013	5161	110.1	23.5	201.2	4.6	1.0	8.4

FM: fresh mass; DM: dry mass; OM: organic matter; [1)] Amount of bedding material to absorb the excreted urine completely

Table 3: Age based proportional factors [1)] in relation to the overall process for rearing young cattle (first calving at 27 months; 625 kg LM, 580 kg mass gain)

Life age [month]	Proportional factor
1 to 27	1.0000
1 to 6	0.1042
7 to 12	0.1994
13 to 24	0.5348
25 to 27	0.1616

[1)] By multiplying the values for the overall process with the proportion factor, the mass amounts of each section are calculated; factors based on feed consumption

3.2 Piglet production

Examples of the calculated mass amounts of solid manure, its liquid fraction and their contents for piglet production with 28 piglets per animal place and year are shown in Tables 4 to 6. The results are divided into the total process (together: dry sow, gestation and farrowing; Table 4) and the sub-processes calculation for dry sow plus gestation and farrowing (Table 5, 6), respectively. If liquid feeding is used this results in higher mass amounts of urine and the liquid fraction of solid manure.

Table 4: Mass amount of solid manure and its liquid fraction for piglet production (28 piglets up to 8 kg live mass, dry sow plus gestation and farrowing together, 264 kg live mass gain (piglets + live mass gain sow) per animal place and year (animal place = 1 sow), N-/P-reduced feed

Straw kg FM/ (AP · d)	Type of manure	Mass amounts						Contents		
		FM	DM	OM	N	P	K	N	P	K
		kg/(AP·a)						g/kg FM		
	Faeces	942	235	190	5.2	3.8	1.9	5.6	4.1	2.0
	Urine	1116	22	-	18.7	1.0	8.0	16.8	0.9	7.1
0.9	Fresh solid manure	1861	536	460	16.8	4.8	10.1	9.0	2.6	5.4
0.9	Rotted solid manure	1721	430	359	10.6	4.7	7.7	6.2	2.7	4.5
0.9	Liquid fraction	1169	24	9	7.3	0.6	6.2	6.3	0.5	5.3
1.3	Fresh solid manure	2270	670	580	22.0	5.2	13.8	9.7	2.3	6.1
1.3	Rotted solid manure	2147	537	453	13.8	5.1	10.5	6.4	2.4	4.9
1.3	Liquid fraction	968	22	12	4.6	0.3	5.2	4.8	0.4	5.4
1.7 [1)]	Fresh solid manure	2678	803	700	27.1	5.7	17.5	10.1	2.1	6.5
1.7 [1)]	Rotted solid manure	2877	719	616	19.0	5.7	17.5	6.6	2.0	6.1

AP: animal place = place · (no. of animals/(place · year)); FM: fresh mass; DM: dry mass; OM: organic matter

[1)] Amount of bedding material to absorb the excreted urine completely

Deviations of the calculated mass amounts result (up to 20 %) by changes in the following input parameters:

- amount of feed intake
- composition of feed intake
- nutrient and mineral content of the feed
- admixtures of bedding material and feed residues
- other bedding materials (assumed: wheat straw)
- decomposition rates of dry matter during storage

Table 5: Mass amount of solid manure and its liquid fraction for piglet production (28 piglets up to 8 kg live mass, dry sow plus gestation, 264 kg live mass gain (piglets + live mass gain sow) per animal place and year (animal place = 1 sow), N-/P-reduced feed

Straw kg FM/ (AP · d)	Type of manure	Mass amounts						Contents		
		FM	DM	OM	N	P	K	N	P	K
		kg/(AP·a)						g/kg FM		
	Faeces	593	148	122	3.0	2.4	1.1	5.0	4.0	1.8
	Urine	765	15	-	11.5	0.5	4.7	15.0	0.7	6.1
0.9	Fresh solid manure	1319	386	335	11.3	3.0	7.1	8.6	2.3	5.4
0.9	Rotted solid manure	1236	309	261	7.1	3.0	5.4	5.8	2.4	4.4
0.9	Liquid fraction	784	16	7	3.9	0.3	3.5	5.0	0.3	4.5
1.3	Fresh solid manure	1641	491	430	15.0	3.3	9.8	9.2	2.0	6.0
1.3	Rotted solid manure	1572	393	335	9.5	3.3	7.5	6.0	2.1	4.7
1.3	Liquid fraction	625	14	9	2.0	0.1	2.9	3.2	0.2	4.7
1.7 1)	Fresh solid manure	1783	538	471	16.6	3.5	11.0	9.3	1.9	6.2
1.7 1)	Rotted solid manure	1924	481	415	11.7	3.5	11.0	6.1	1.8	5.7

AP: animal place = place · (no. of animals/(place · year)); FM: fresh matter; DM: dry matter; OM: organic matter

1) Amount of bedding material to absorb the excreted urine completely

Table 6: Mass amount of solid manure and its liquid fraction for piglet production (28 piglets up to 8 kg live mass, farrowing, 264 kg live mass gain (piglets + live mass gain sow) per animal place and year (animal place = 1 sow), N-/P-reduced feed

Straw kg FM/ (AP · d)	Type of manure	Mass amounts						Contents		
		FM	DM	OM	N	P	K	N	P	K
		kg/(AP·a)						g/kg FM		
	Faeces	349	87	68	2.3	1.5	0.8	6,5	4.2	2,3
	Urine	351	7	-	7.2	0.5	3.3	20.6	1.3	9.3
0,9	Fresh solid manure	542	151	125	5.2	1.7	2.8	9.5	3.2	5.2
0.9	Rotted solid manure	486	121	98	3.3	1.7	2.1	6.7	3.5	4.4
0.9	Liquid fraction	385	9	3	3.6	0.3	2.8	9.4	0.9	7.2
1,3	Fresh solid manure	628	179	151	6.5	1.8	3.7	10.3	2.9	5.9
1.3	Rotted solid manure	575	144	117	4.1	1.8	2.8	7.1	3.1	4.9
1.3	Liquid fraction	342	8	3	2.9	0.3	2.5	8.5	0.8	7.2
1,7 1)	Fresh solid manure	895	266	229	10.5	2.2	6.5	11.7	2.4	7.2
1.7 1)	Rotted solid manure	953	238	201	7.3	2.2	6.5	7.7	2.3	6.8

AP: animal place = place · (no. of animals/(place · year)); FM: fresh mass; DM: dry mass; OM: organic matter

1) Amount of bedding material to absorb the excreted urine completely

4. Conclusion and outlook

Mass amounts and contents of solid manure and its liquid fraction can be approximately calculated with a mass balance approach based on feed, digestibility parameters and animal performance. In the future, a model of formulating and analyzing rations for different animal categories will be integrated in the computational model.

References

[1] DLG (2014): Bilanzierung der Nährstoffausscheidungen landwirtschaftlicher Nutztiere. Arbeiten der DLG, Band 199.
[2] Horlacher, D.; Rutzmoser, K.; Schultheiß, U.(2014): Festmist und Jaucheanfall. KTBL-Schrift 502.
[3] Schofield, C.P. (1988): Water Absorption by Spring-Barley bedding straw. Biological Wastes 26, 133–144.
[4] KTBL (2015, August 17): KTBL-Onlineanwendung Wirtschaftsdünger-Rechner. Retrieved from http://daten.ktbl.de/wdrechner/?tx_ktblsso_checktoken[token]=.
[5] BMEL (2015, August 17): Amendment of the German Fertilisation Ordinance from 2014/12/18, see Annex 9 (to §12). Retrieved from http://www.etracker.com/lnkcnt.php?et=dQsrB9&url=http%3A%2F%2Fwww.bmel.de%2FSharedDocs%2FDownloads%2FService%2FRechtsgrundlagen%2FEntwuerfe%2FEntwurfDuengeverordnung.pdf%3F__blob%3DpublicationFile&lnkname=EntwurfDueV.

TC-P_09 Use of a feed additive based on biochar for mitigation of ammonia emissions from weaned piglets and broilers

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1. Objectives

Biochar is a product based on pyrolysis of biomass [1]. It was suggested that its use might contribute to the mitigation of ammonia (NH_3) emissions from livestock production [2]. The aims of the present study were to investigate whether the supplementation of a commercial additive based on biochar ("Carbon Futter", manufactured by EM Schweiz) to the feed contributes to the mitigation of NH_3 emissions from weaned piglets and broilers. Several approaches were applied to measure the emissions from slurry and the manure, respectively, produced by animals fed with the biochar supplemented feeds compared to animals receiving non-supplemented rations. Moreover, parameters related to animal performance and health were studied.

2. Methodology

2.1 Feeding experiment with weaned piglets

A total of 80 weaned piglets with a body weight of 8 kg were used in the experiment. They were housed in 16 pens with partially slatted floor. Eight replicates of five piglets each were assigned to both dietary variants without (DP_C) and with 3% of the additive based on biochar (DP_{BC}), respectively. They received the experimental diets *ad libitum* over 28 days up to a live weight of approx. 18 kg. During this period, live weight gain, feed intake and feed conversion were measured.

2.2 Feeding experiment with broilers

For the experiment with broilers, a total of 1600 birds (Ross PM3) divided into four groups of 400 birds were used. Two groups received a diet without (DB_C) and two groups a diet with 2% of the additive based on biochar over 36 days (DB_{BC}). Straw pellets were used as litter material (1.2 kg/m^2). A covered veranda was accessible after 22 days. The investigations encompassed live weight gain, feed intake and feed conversion, slaughter performances, mortality, litter quality and lesions of foot pads and hocks.

2.3 Collection of excreta and emission measurements

During 6 days, the slurry was collected from one pen of each variant DP_C and DP_{BC}. A part of the collected slurry was stored over 33 days. An aliquot of the slurry from group DP_C was supplemented with 3% of a commercial biochar and stored separately. This variant is denoted DP_{BC}+. The broiler manure was collected at the end of the production cycle.

For the emission measurements, one liter of the non-stored slurry was filled into trays (surface 0.12 m^2) for both of the variants DP_{BC} and DP_C and placed onto the soil of the experimental plot covered with grassland. The trays were used in order to mimic an emitting surface which scoops its maximum emission potential. The stored slurries DP_{BC}, DP_C, and DP_{BC}+ and the broiler manures DB_{BC} and DB_C were spread onto the sward covering a surface of 1m^2. The canopy height was 5 to 8 cm. The applied amount corresponded to 30 m^3 ha^{-1} and to 7 m^3 ha^{-1} for slurry and manure, respectively. The aim of these two treatments was to mimic the normal procedure of slurry and manure application in the field.

The measurements were carried out with wind-tunnels which were placed over the emitting surfaces. The determination of NH_3 air concentrations were carried out using automated impinger sampling devices. They were sampling air at approx. 0.7 L/min, which was drawn through a threaded midget impinger (volume = 22 mL) containing 15 mL of a sodium acetate buffer solution of pH 4. Ammonia is converted to ammonium in the acid solution. The NH_4^+ content was measured using commercial test kits (Hach-Lange, LCK 304) and a photometer (DR 2800 VIS, Hach-Lange GmbH, Rheineck). The measurements lasted over 3 days (slurry) and 6 days (manure). Each variant was carried out in 3 replicates.

3. Results and discussion

3.1 Performance

Weaned piglets: The piglets fed with (DP_{BC}) and without supplementation (DP_{BC}) of the additive did not exhibit statistically significant differences in daily weight gain, feed intake and feed conversion. Although, the feed intake of the group DP_{BC} was numerically higher (Table 1).

Table 1: Daily weight gain, feed intake and feed conversion of weaned piglets fed a diet supplemented with 3% of a commercial additive based on biochar (DP_{BC}) or a control diet (DP_C). SE: standard error

	Average DP_{BC}	**Average** DP_C	**SE**
Daily weight gain (g d^{-1})	360	334	37.1
Feed consumption (g d^{-1})	563	529	51.5
Feed conversion	1.58	1.62	0.05

The mortality of the piglets was relatively high due to problems related to diarrhea at the beginning of the experiment. The losses were not associated to the feeds. Detailed results on the performances can be obtained from [3].

Broilers: The daily weight gains differed only slightly during the entire production period (Table 2). Thus, the live weights after 36 days were 2226 g and 2208 g for the variants DB_{BC} and DB_C, respectively. Feed consumption was slightly higher for the group DB_{BC} compared to DB_C. The feed conversion index exhibited very low differences over all phases of the growing period.

Table 2: Daily weight gain, feed intake and feed conversion of broilers fed a diet supplemented with 2% of a commercial additive based on biochar (DB_{BC}) or a control diet (DB_C).

	Average DB_{BC} [a]				**Average** DB_C [a]			
Phase of the fattening period (day)	10	21	28	36	10	21	28	36
Daily weight gain (g d^{-1})	25.4	43.9	54.5	60.7	26.1	45.6	54.5	60.2
Feed consumption (g d^{-1})	35.4	60.1	73.6	95.1	36.9	61.6	73.9	96.5
Feed conversion index	1.39	1.37	1.35	1.57	1.38	1.35	1.36	1.60

[a] Standard errors are not given due to the low number of replicates

The slaughter performance of the two groups (weight at slaughter, dressing percentage and carcasses quality) was almost equal. The mortality rate was 1.5% for the variant DB_{BC} and 1.8% for DB_C. The causes for mortalities did not differ between the two groups. The quality of the litter was assessed on day 28 and 35. It was considered as slightly humid for the group DB_{BC} on both occasions while for DB_C the litter was considered as dry on day 28 and slightly humid on day 35. Approx. 20% and approx. 30% of the litter surface were crusted for DB_C and DB_{BC}, respectively.
The results of the evaluation of the foot pad and hock lesions are given in Table 3. Lesions of the foot pads occurred more frequently than of the hocks. In general, the birds of the group DB_{BC} exhibited more lesions for both foot pads and hocks than the group DB_C. This might be due to the differing quality of the litter (higher moisture content for DB_{BC}) as mentioned above.

Table 3: Percentages of degree 1 lesions at the foot pads and the hocks of broilers fed a diet supplemented with 2% of a commercial additive based on biochar (DB_{BC}) or a control diet (DB_C) at day 29 and 35 of the experiment

	Average DB_{BC}		**Average** DB_C	
	foot pads	hocks	foot pads	hocks
No lesions at day 29	15%	93%	25%	75%
Lesions of degree 1 at day 29	85%	7%	75%	25%
No lesions at day 35	0%	50%	0%	65%
Lesions of degree 1 at day 35	100%	50%	100%	35%

Within the experiments, an effect of the use of biochar as a feed additive did not significantly influence most parameters related to the performance of the animals. This contrasts to findings from other studies e.g. [5,6].

3.2 Emissions

The cumulated NH_3 emissions of slurry exposed in trays measured during 3 days did not differ between the groups of weaned piglets DP_{BC} and DP_C. It reached approximately 10 g N/m^2 which corresponds to a loss of approx. 15% of applied TAN (Figure 1). This is within the range reported by [4] for field applied pig slurry. The trends of the emission observed in the current experiment differ from the curves observed in field measurements on application of slurry which exhibit a strong decline after the first day. This discrepancy might be due to the slurry exposed in trays and not spread onto the soil.

Similarly, the slurry stored over 33 days and applied onto the sward exhibited comparable emissions for all treatments (Figure 2). Neither the biochar used as a feed additive (DP_{BC}) nor the addition of biochar to the slurry obtained from the control (DP_{BC}+) seemed to influence the release of ammonia. The emission level was significantly lower as compared to the slurry exposed in trays.

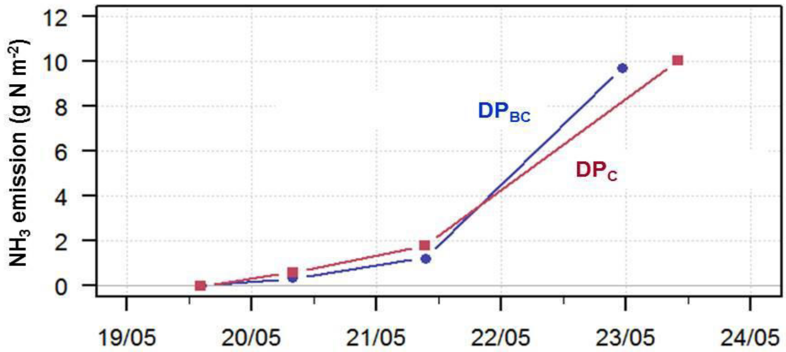

Figure 1: Cumulated NH_3 emissions of slurry exposed in trays obtained from weaned piglets fed a diet supplemented with 3% of a commercial additive based on biochar (DP_{BC}) or a control diet (DP_C) in g N/m^2.

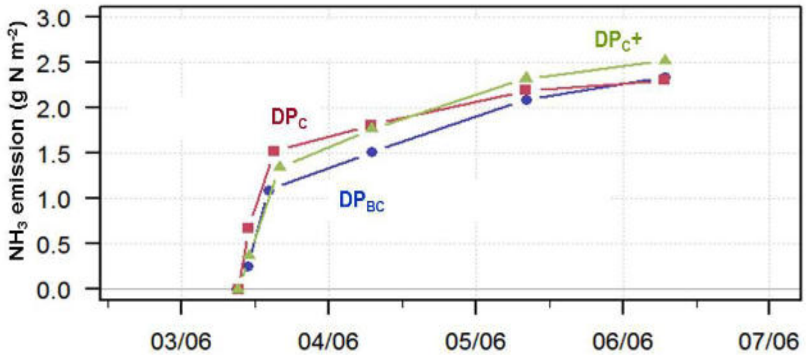

Figure 2: Cumulated NH_3 emissions of slurry spread onto grassland. The slurry was obtained from weaned piglets fed a diet supplemented with 3% of a commercial additive based on biochar (DP_{BC}), a control diet without (DP_C) or DP_C with addition of a commercial biochar (DP_C+) and stored for 33 days before spreading. Emissions are given as g N m^{-2}.

Similar to the slurries, the use of biochar as a feed additive did not influence the emission level of the broiler manure. Table 4 gives an overview on the cumulative emissions and the emission flows at day 3 of the measurements.

The three experiments on NH_3 emissions suggest that the use of biochar was ineffective as a measure to mitigate ammonia emissions. This is in contrast to similar studies [2,7] which found a potential for emission mitigation. It has to be mentioned that the experimental approach of the present study for assessing the emissions was rather simplifying: (i) wind-tunnels are not appropriate to produce quantitative results since they strongly influence the environment above the emitting surface [8]; (ii) the experimental approach only encompassed the emission stage of application. However, under real world conditions the use of biochar and a feed supplement would act over the entire emission cascade which includes housing, manure storage and application. Therefore, the results of the present study cannot be considered as conclusive.

Table 4: NH_3 emissions at day 3 of slurry from weaned piglets and of manure from broilers fed a diet supplemented with commercial additive based on biochar (DP_{BC}, DB_{BC} and a control diet without (DP_C, DB_{BC}) with or without storage over for 33 days before application. DP_{BC}+ denotes addition of a commercial biochar to slurry DP_C. Cumulated emissions are given as g N m^{-2} and the flows as µg N m^{-2} s^{-1}

	Cumulative losses at day 3	**Flow at day 3**
	g N m^{-2}	µg N m^{-2} s^{-1}
Slurry from weaned piglets without storage, DP_{BC}	9.7	78
Slurry from weaned piglets without storage, DP_C	10.0	57
Slurry from weaned piglets with storage, DP_{BC}	2.4	3.8
Slurry from weaned piglets with storage, DP_C	2.3	1.7
Slurry from weaned piglets with storage, DP_{BC}+	2.5	2.9
Broiler manure without storage, DB_{BC}	0.8	0.8
Broiler manure without storage, DB_C	0.8	0.7

4. Conclusion and outlook

To our knowledge, this is the first study which investigated the impact of biochar included in a commercial feed supplement or mixed to slurry before storage on ammonia emissions. Overall, an emission mitigation effect could not be found. Moreover, biochar did not seem to influence the performance and health parameters of weaned piglets and broilers. This contrasts to results published earlier and to experiences reported from the practice. However, additional studies under more realistic conditions are required for a conclusive evaluation of the potential of biochar regarding ammonia emission mitigation in livestock production.

It has to be noted that biochar might exert other positive effects in livestock production systems [1] which were not investigated in the present study. In order to benefit from such effects it seems likely that biochar should be introduced at a prior stage of the emission cascade, i.e. via feed or in the housing system. This would increase the potential to activate multiple useful effects. To confirm this hypothesis further investigations are required.

Acknowledgements

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References

[1] Atkinson C.J., Fitzgerald J.D., Hipps N.A., 2010. Potential mechanisms for achieving agricultural benefits from biochar application to temperate soils: a review. Plant and Soil 337, 1-18.

[2] Taghizadeh-Toosi A., Clough T., Sherlock R., Condron L., 2011. A wood based low-temperature biochar captures NH_3-N generated from ruminant urine-N, retaining its bioavailability. Plant and Soil, 1-12.

[3] Fischlin I. 2014. Use of a feed additive based on biochar for weaned piglets and broilers as a measure to reduce ammonia emissions. BA Thesis in German with abstract in English. Bern University of Applied Sciences School of Agricultural, Forest and Food Sciences, Zollikofen Switzerland (http://www.agrammon.ch/documents-to-download/further-information/; downloaded 2015/05/11).

[4] Häni C., Sintermann J., Kupper T., Neftel A. 2015. Ammonia emission after slurry application to grassland. Proceedings of the 16th RAMIRAN International Conference 8-10 September 2015, Hamburg, Germany.

[5] Van D.T.T., Mui N.T., Ledin I., 2006. Effect of method of processing foliage of Acacia mangium and inclusion of charcoal in the diet on performance of growing goats. Animal Feed Science and Technology 130, 242-256.

[6] Abächerli F., 2014. Pflanzenkohle und Kompost in der Einstreu. Verora GmbH Ressourcenschonende Naturerzeugnisse. [downloaded on 2014/07/01, http://www.ithaka-institut.org/ithaka/media/doc/1397520795797.pdf].

[7] Schimmelpfennig S., Mueller C., Gruenhage L., Koch C., Kammann C., 2014. Biochar, hydrochar and uncarbonized feedstock application to permanent grassland-Effects on greenhouse gas emissions and plant growth. Agric. Ecosyst. Environ. 191, 39-52.

[8] Genermont S., Cellier P., 1997. A mechanistic model for estimating ammonia volatilization from slurry applied to bare soil. Agricultural and Forest Meteorology 88, 145-167.

TC-P_10 Identification of odorous compounds in air following land spreading of animal slurry

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1. Objectives

Odorant emissions following land spreading of animal slurry cause nuisance for local surroundings and can hamper further expansion of animal production locally. Studies of odorous emissions following land spreading of animal slurry are currently scarce, especially on full-scale applications [1]. Here, results from 2 field studies are reported and discussed with objectives of assessing:

- The influence of dynamic flow rate on the odorant concentrations and emissions.
- The difference in emission patterns for different types of slurry and application technology.
- The duration of gaseous emissions.
- The dominant odorants after slurry applications.

2. Methodology

In the first study, emissions from full-scale land spreading of: 1) cattle 2) pig 3) mink and 4) anaerobic digested (mixed) slurry were examined, with application rates of 53, 32, 38 and 53 ton/ha respectively. The second study was carried out in spring 2015 where the effect of acidification of pig slurry on odorous emissions was evaluated (application rate 30 ton/ha).

In both experiments, dynamic chambers were used to supply a constant flow across the soil surface and to simulate the wind effects on gaseous emissions. The dynamic chambers were designed on the basis of CFD modelling, ensuring that the wind profile within the chamber was relatively uniform. Measured VOC emissions included carboxylic acids, aldehydes, alcohols, ketones, phenols, indoles and organic sulfur compounds. A time resolved, high sensitivity Proton-Transfer-Reaction Mass Spectrometry (PTR-MS) was employed for capturing the fast change of VOC, H_2S and NH_3 emissions after the land spreading of manure slurry. The PTR-MS was run under standard conditions during all the measurements, with the drift tube voltage of 600 V, the drift tube temperature of 60 °C and the drift tube pressure of 2.1~2.2 mbar. Thus the E/N number was controlled at around 135 Td during all the measurements. Selected ions were monitored with dwell time between 200 and 2000 ms during each measurement cycle. Masses and dwell time selection for monitoring was based on previous experiences with odorant compounds from slurry samples and animal buildings. Since only the PTR-MS was used for the experiments, the qualification and mass assignment was also based on experiences and relevant literatures for odour measurements of slurry samples and animal buildings [2]. For the quantification, the concentrations (in ppbv) measured by the PTR-MS were first corrected for the field background and then further corrected according to the corresponding reaction rate constant for each available compound.

In order to investigate the influence of flow rate on odorants concentrations and therefore total emissions, two pre-tests were conducted under two different conditions. In the first pre-test, the chamber flow rate was adjusted within a relatively short period (within 2 hours, cross comparison) after the slurry was applied, with the assumption that the slurry emitting conditions were unchanged during the short period. In the second pre-test two different flow rates were set in two identical dynamic chambers for a period of 20 hours after the 1st pre-test was finished, with the purpose of testing the longer-term emission effects.

3. Results and discussion

3.1 Influence of dynamic flow rate on odorant concentrations

Figure 1 shows an example of the acetic acid concentration dependence test on flow rate under the 2nd condition. The test revealed only little dependence of the acetic acid concentration on the two different flow rates, with the estimated difference in area (below the curves; pointing to x-axis) of around 13%, regardless of the flow rate. Other odorants agreed well with the trend observed for acetic acid, meaning the higher flow rate will end up with higher emission rates. In fact, the test under the 1st condition showed the similar tendency, with ammonia showed the highest dependence of the concentration regarding air flow rate. However, the decrease of ammonia concentration was still within 30%, even the flow rate was as high as 3 times.

In summary, the odorant concentrations were not affected significantly by the air flow rate, indicating that for given emitting surface conditions, a high air flow rate may cause higher emissions.

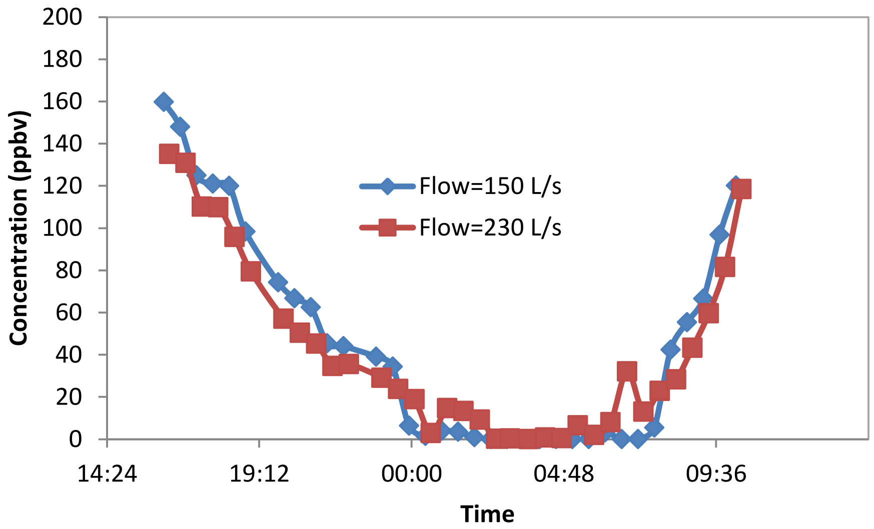

Figure 1: Influence of air flow rate in dynamic chambers on acetic acid concentration.

3.2 Estimation of total emissions

The emission rate (mg h^{-1}) was estimated based on the concentration and corresponding flow rate for each slurry application. Table 1 shows the total emission (mg) within the first day after slurry was applied for selected compounds. It should be noted that only limited data (3 hours) from the second study were included in Table 1 due to an instrument failure. The raw cattle slurry application in the first study emitted less VOCs and H_2S, when compared to raw pig slurry application. Emissions of sulfur-containing compounds were not in the high range for any slurry types except for mink slurry application, although this conclusion was kept temporary for the pig slurry application, with which the first 3~5 minutes was lost after the pig slurry was applied, due to the technical difficulties (lack of people to move chamber efficiently, to make connections and to start the air flow). Due to the very strong wind when the pig slurry was applied, H_2S might be lost significantly during the first 2 minutes, with the high volatility of H_2S [3].

Table 1: Total emission (mg) for selected compounds during the first day after slurry was applied. For the comparison of pig slurry with acidification to raw pig slurry, only the first 3 hours data were available due to instrument failure.

Slurry type\compound	Ammonia	Acetic acid	Butanoic acid	H_2S	Methanethiol	4MP	Duration
cattle slurry, raw	3544	34,2	3,8	0,06	0,08	10,3	21 hours
cattle slurry, accidified	0	242	12,8	0,15	0,25	15,1	22 hours
pig slurry, raw	983	3787	1940	0,34	0,03	445	24 hours
Digested slurry	1037	88,7	16,7	1,66	0,02	10,6	24 hours
Mink slurry, raw	1562	3798	765	15,3	1,56	51,3	24 hours
pig slurry, raw	875	2108	758	0,10	0,14	163	3 hours
pig slurry, acidified	421,3	1021	745	0,53	0,25	255	3 hours

4-MP – 4-methyl-phenol.

The raw cattle slurry application generated comparably higher ammonia emission than other types of slurry, while the mink slurry application gave the second highest ammonia emission. Both raw pig slurry and mink slurry applications emitted very high emissions of carboxylic acids, while pig slurry application also gave very high emission of 4-methylphenol (a very smelly compound) but mink slurry application gave comparably higher emissions of H_2S and methanethiol.
When acidified slurry application was compared to the raw slurry application, the acidification seemed to reduce ammonia emission significantly especially for cattle slurry. While the acidified pig slurry emitted lower amounts of acetic acid compared to the raw pig slurry, the observed trend for cattle slurry was opposite. Thus the influence of acidification on emissions of volatile acids needs to be further investigated. For both slurry types, acidification seemed to increase the emission of 4-methylphenol significantly and H_2S (Table 1).

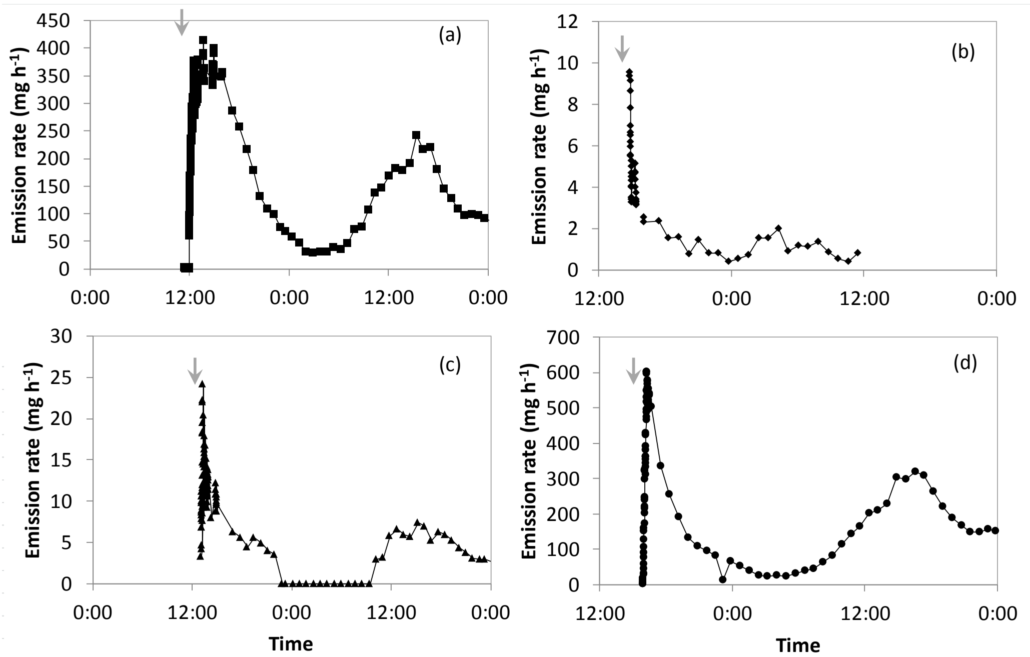

Figure 2: Emission rate (of the area dynamic chamber covered) of Acetic acid for selected types of slurry applications. (a) pig slurry, raw; (b) cattle slurry, raw; (c) digested slurry; (d) mink slurry. The arrows indicate the starting time of the slurry application.

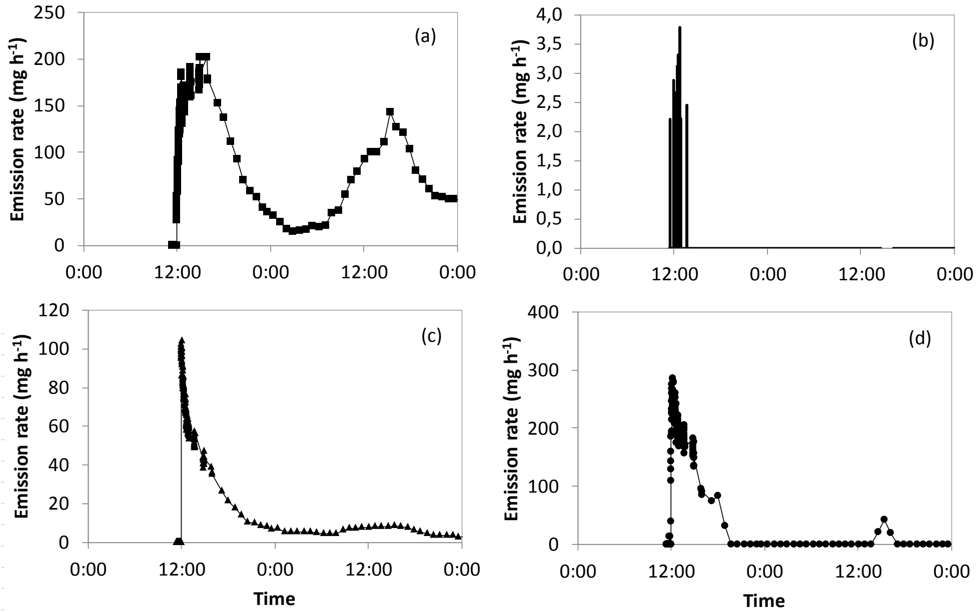

Figure 3: Emission rates (of the area dynamic chamber covered) of selected compounds for pig slurry (raw) application. (a) Butanoic acid; (b) Hydrogen sulfide; (c) 4-methylphenol; (d) Ammonia.

3.3 Emission pattern comparison

Figure 2 shows an example of emission rate of acetic acid for selected slurry applications. High emission peaks of acetic acid were observed for both pig slurry and mink slurry applications, compared to cattle slurry and digested slurry applications. A peak was typically observed during the afternoon of the day after slurry was applied. This may due to the higher air temperature during the afternoon period.

Figure 3 shows examples of emission rates of selected compounds for raw pig slurry application. Different compounds showed different emission trends. Emission of H_2S showed small peaks immediately after the slurry was applied (in this case the first 5 minutes was unfortunately lost), while emission peaks from other compounds were observed to have slower response. Emission of butanoic acid gave a significantly peak during the afternoon of the day after the slurry was applied, while 4-methylphenol only generated a small emission peak. This may due to the higher vapour pressure of butanoic acid, compared to 4-methylphenol at the same temperature.

3.4 Odour contribution

Different slurry types and application methods may contribute to odour significantly different, according to measured real-time odorants concentrations and time scale considered. However, some of the important odorants could be generalized for most of the slurry types, due to either the low odour threshold value (OTV; ppbv) of the odorants [4] or the relative high concentration of the odorant. The most common odorants observed included: 4-methylphenol (OTV: 0.02 ppbv); Butanoic acid (OTV: 0.2 ppbv); Skatole (OTV: 0.005 ppbv); Trimethylamine (OTV: 0.03 ppbv); Pentanoic acid (OTV: 0.4 ppbv) and 3-methylbutanoic aicd (OTV: 0.09 ppbv); Acetic acid (OTV: 6.2 ppbv); Propanoic acid (OTV: 5.7 ppbv); 4-ethylphenol (OTV: 0.4 ppbv); H_2S (OTV: 0.8 ppbv) and methanethiol (OTV: 0.03 ppbv).

4. Conclusion and outlook

Two studies using dynamic chambers on emissions of odorous compounds after slurry application were conducted. Pre-tests revealed that the odorant concentrations were not significantly dependent on air flow rate applied, meaning high flow rate causing high odorant emissions. Different slurry types were associated with significant differences in odorant emissions, depending on compound and conditions in the field. Cattle slurry may give higher ammonia emission (depending on slurry compositions; [5]), but contributes generally lower carboxylic acids and phenols, compared to pig slurry. Pig slurry and mink slurry was found to contribute highest amount of carboxylic acids and phenols. Sulfur compounds were found only at low concentrations, after a few minutes of slurry application. Odorants usually gave another emission peak on the afternoon of the day following application, especially for carboxylic acids, probably due to the temperature change in the soil (not measured). Different odorants contributed to the odour depending on slurry type, application methods, field conditions, and time scale. Major odorants may include: 4-methylphenol, carboxylic acids, skatole, trimethylamine and sulfur compounds.

Acknowledgements

We would like to acknowledge, the technical staff at AU Foulum especially Peter S. Nielsen and Simon Kristensen.

References

[1] Feilberg, A.; Bildsoe, P.; Nyord, T. (2015): Application of PTR-MS for measuring odorant emissions from soil application of manure slurry, Sensors (Switzerland) 15, 1148-1167

[2] Liu, D.; Løkke, M.M.; Leegaard Riis, A.; Mortensen, K.; Feilberg, A. (2014): Evaluation of clay aggregate biotrickling filters for treatment of gaseous emissions from intensive pig production, Journal of Environmental Management 136, 1-8

[3] Feilberg, A.; Liu, D.; Nyord, T. (2014): Temporal variation in odorant composition following land application of manure, Chemical Engineering Transactions 40, 55-60.

[4] Hansen, M.J.; Feilberg, A. (2015). Odor threshold values of key odorants in air from pig houses. Sumitted to a peer-reviewed journal.

[5] Hansen, M.N., Sommer, S.G., Hutchings, N.J. and Sørensen, P. (2008): Emissionsfaktorer til beregning af ammoniakfordampning ved lagring og udbringning af husdyrgødning [Emission factors for calculation of ammonia volatilization by storage and application of animal manure], DJF Rapport Husdyrbrug 84, Ministry of Food, Agriculture and Fisheries, Danish Institute of Agricultural Sciences, Tjele.

TC-P_11 A French inventory of solid manure (cattle, pig, poultry) stored in temporary field heaps

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1. Objectives

The objectives of the present work were:

- to estimate the quantity of solid manure likely to be stored in temporary field heaps for cattle, pig and poultry farms and,
- to specify the farmer practices when storing the solid manure in temporary field heaps. .

2. Methodology

2.1 Estimation of solid manure likely to be stored in temporary field heap.

Solid manure from cattle production. The French cattle breeding is characterized by a diversity of farms specializing in dairy and/or meat production, composed of one or more buildings housing different categories of cattle (cows, calves, young cattle, replacement heifers, ...) under specific housing systems (slurry or litter systems). Based on the 2010 Agriculture Census [1] the estimation of solid manure is done by distributing the number of head of the different cattle species (Table 1) according to a set of type of housing (tie or free stall with litter floor). The quantity of manure produced by cattle under each different type of housing is a technical value [2] expressed in tone per livestock unit (LU) and per year without considering the housed period. The animal Livestock Unit value for each cattle species is given table 1. In France, according to the region, the housed period ranges from 180 to 240 days for dairy cattle and from 60 to 180 days for other cattle. The total manure produced in France by cattle was thus obtained with the use of an appropriate set of statistical and technical data which allowed to multiply the quantity of manure (t/LU year) with the number of heads of the different categories of cattle according to the housed period.
Solid manure from poultry production. In France each type of poultry (Laying Hens, Broiler, Turkey Guinea Fowl) is kept under a wide variety of production systems that range from standard to quality production (Outdoor, Red Label, Organic, AOC) which differ mainly by the duration and type of livestock feed. These different types of production involve different types (solid manure, droppings or slurry) and quantities of manure per animal. The total solid manure (solid manure and droppings) produced by the different poultry species was estimated by multiplying the statistical number of heads [3] with the quantity of manure per head established by a farm survey [4]. The main data and references used in this study are listed Table 1.
Solid manure from pig production. The French pig production is well standardized and mainly conducted on slatted floor with slurry production. According to the 2010 Agriculture Census the pig production on litter concerns less than 10% of the places of sows, weaners or fattening pigs. In this study the estimation of solid manure produced by the pig farms was done by multiplying the statistical number of places on litter for the sows, the weaners and the fattening pigs [1]. with the technical quantity of solid manure given for the 3 categories of pig (Table1).

2.2 Survey of temporary field storage practices.

A questionnaire covering management aspects relevant to temporary field storage (shape and dimension of the field heap, time and period of field storage,…) was sent to some of the professional organization (Chambers of Agriculture, Technical Institutes) to provide technical data for storage in the field.

3. Results and discussion

3.1 Estimation of the quantity of solid manure likely to be stored in temporary field heaps for cattle, pig and poultry farms

The total solid manure produced by the cattle, pigs and poultry in French farms is estimated at around 66 million tons per year. Under the current regulations in France, about 80% of this manure (55 million tons) is likely to be stored in temporary field heaps (Table 2). Around 94% of it is produced by cattle, 5% by poultry and 1.5% by pigs. The amount of cattle manure likely to be stored in field heaps (52 million tons, Table 2) comprises: (i) about 36 million tons of solid manure produced by loose housing on deep litter, mainly from dairy cattle (c.a. 27%) and other cattle (c.a. 60%), (ii) about 8.5 million tons of solid manure produced in tie-stall by suckler cattle (c.a. 37%), dairy cows (c.a.35.6%) and other cattle (26.5%) and (iii) about 7 million tons of solid manure from free stall bedding area on litter mainly produced by dairy cattle (c.a. 59%). The remaining manure from cattle is solid manure from free stall cubicle house (about 12 million tons) mainly present in farms specializing in milk production. This last solid manure is not allowed to be stored in temporary field heaps.

The amount of solid manure and droppings produced by poultry are estimated at about 2 million tons of solid manure and about 0.8 million tons of droppings. Solid manure is produced at around 48% by broilers, at 31 % by turkey and at 21% by other poultry (guinea fowl, duck, geese,…). About 66% of poultry manure comes from standard farms and the remaining from quality farm (Outdoor, Red Label, Organic, AOC). Droppings are produced at 82% by laying hens for consumption eggs, the rest of droppings being from laying hen for hatching eggs. The quantities of manure from pig farms are estimated at around 0.8 million tons and are produced at 63% by fattening pigs, at 25% by sows and at 12% by weaning pigs.

Table 1: Quantity of fresh solid manure produced according to the species and the type of housing [2], [4]

Animal specie	Type of housing	Quantity of manure
Cattle	Tie-stall	
Dairy cow (1.1 LU[1])	Litter floor	15 t/Lu year
Suckler cow (0.85 LU)	Free stall	
Beef <1 year (0.3 LU)	Whole deep litter	13.5 t/Lu year
Beef 1-2 years (0.6 LU)	Litter lying area	6.5 t/Lu year
Beef>2 years (0.8 LU)	Litter cubicle	16 t/Lu year
Poultry		
Laying Hens[2]	Cages or litter floor	17 kg/animal/year or 1.15
Broiler[3]	Litter floor	kg/animal
Turkeys[3]	Litter floor	1.2 to 1.9 kg/animal
Guinea Fowl[3]	Litter floor	9.4 to 15 kg/animal
		1.5-1.9 kg/animal
Pig		
Sows (1 animal cycle/year)	Scrapped litter or Accumulated litter	1763.5 kg/animal cycle
Weaners (6.5 animal cycles/year)		36 kg/animal cycle
Fattening pigs (3.3 animal cycles/year)		242.5 kg/animal cycle

[1] Livestock Unit, [2] in cages and alternative systems, [3] standard or quality (organic, label,…) production

Table 2: Total amount of solid manure likely to be stored in temporary field heap in France from cattle, pig and poultry farms and distribution by type of animal (in million tons)

Cattle	**National Amount**	**Dairy cattle**	**Suckler cattle**	**Beef (< 1 year)**	**Beef (1-2 years)**	**Beef (> 2 years)**
Tie-stall	8.5	3.0	3.2	0.7	0.7	0.8
Whole area on litter	36.1	6.7	10.1	5.4	6.9	7.2
Bedding area on litter	6.9	4.1	0.8	0.5	0.8	0.8
Poultry	National Amount	Broiler	Turkey	Other poultry	Laying hen (eggs for consumption)	Laying hen (hatching eggs)
Solid manure	1.9	0.9	0.6	0.4	-	-
Dropping	0.8	-	-	-	0.6	0.2
Pig	National Amount	Weaning Pig		Fattening Pig	Sow	
Manure	0.8	*0.1*		*0.5*	*0.2*	

Under the current French regulations most of solid manure can be stored directly in field. However, in some cases, on-farm storage can be necessary to respect the 2 months of storage in the farm set by the French regulation. Thus, dairy manure should be stored on farm before field storage if the building cleaning frequency is less than two months for health reasons (mastitis).

In poultry, due to a production cycle of less than two months, manure from broiler farms under standard production (0.6 million tons) requires an on-farm storage before the field storage.

In pig farming, an on-farm storage may be necessary if manure cleaning frequency (accumulated or scrapping) is less than 2 months.
However, the lack of accurate data on cleaning frequencies doesn't allow to estimate manure amount likely to be stored in field without on-farm storage for pigs and poultry. It is thus considered in this study that all pig manure can be stored in the field. All poultry manure is considered storable in field directly or following an on-farm storage; in the absence of statistical data on manure drying devices, the resulting total values were retained.

According to the Figure 1 cattle manure likely to be stored in field is heterogeneously distributed over the country. However, solid manure from suckler cattle housed in tie-stall is mainly produced in regions where tie-stall is more dominant than other stall type (Auvergne, Franche Comté, …). In the other regions the solid manure from loose housing on deep litter is the main manure likely to be stored in field. This deep litter manure is on average 67% of the regional amount but this proportion is even higher for some regions where it reaches about 90%, (Centre, Picardie, Champagne Ardenne). Solid manure from bedding area on litter is mainly produced by dairy cows (Bretagne, Pays de la Loire, Basse Normandie) due to a greater number of places in this mode of stalling. Some regions such as Bretagne, Bourgogne, or Auvergne have a larger amount of manure produced by dairy cows or lactating cows while other regions, Loire and Champagne Ardennes, have to manage manure from all the different animal categories.
In poultry, the largest quantities of manure likely to be stored in field are located in areas of high poultry production (Bretagne, Pays de la Loire, Rhône Alpes and Aquitaine). The region of Bretagne and Pays de La Loire predominantly manage broiler manure and to a lesser extent, laying hen droppings while the region of Rhone Alpes mainly manages laying hen droppings.
The regions producing the most pig manure are the region of Bretagne and Pays de la Loire. This information is important because it implies that the characteristics of storable manure to the field should vary from one region to another.
All these results are very useful to map and characterize the threat of temporary field heaps storage to water quality according to the Nitrate Directive regulations.

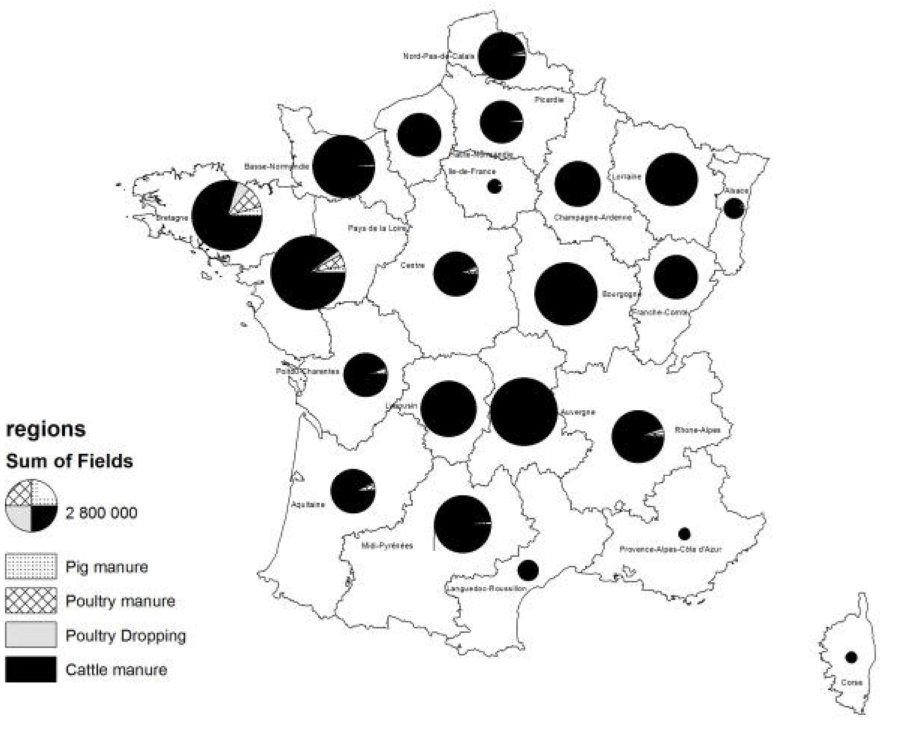

Figure 1: Geographical distribution of solid manure likely to be stored in temporary field heap in France

3.2 Farmer practices when storing the solid manure on field

There is very little statistical or technical data on temporary field manure storage practices. The questionnaire comebacks from the profession were very limited (3 for cattle and 2 for poultry) and do not allow to draw conclusions about temporary field manure storage practices. Instead, the questionnaire returns reflect the expected diversity of storage practices for cattle manure. The responses indicate various practices. The manure field heap is shaped according to the trailer used for transporting manure from farm to field. Heap height is between 1.5 and 2 meters with a circular or rectangular base, a width of 2 to 3 meters and of variable length (3 to more than 40 meters) as a function of the amount of manure to be stored. Storage times reported vary from 1 to 4 months or 8 to 10 months depending on the region. For poultry, storage times in the field are probably short but are not specified in the questionnaire. The field storage is generally deformed and obtained by a simple manure discharge without heap shaping. No response was given about the period of storage.

4. Conclusion and outlook

The estimation of the French national amount of solid manure likely to be stored in temporary field heaps (55 millions of tons) show that the main source is cattle farms (94% of the quantity) followed by poultry and pig farms. A distribution of the manure according to the animal species and the type of building allows specifying the different solid manures likely to be stored in temporary field heaps in the different geographical regions. These results are very useful to map and characterize the threat of the temporary field heaps storage to water quality according to the Nitrate Directive regulations. The study was also an opportunity to update the amount of solid manure to be managed in France by the main livestock production ie cattle, pig and poultry farms. The value of 63 million tons obtained in this study is lower than the values currently used (90 million tons, [5]) to address national regulations on environmental issues or to guide sustainable development actions like the valorization of manure by anaerobic digestion. For example, this new data would be useful for emissions inventory if we consider the impact of the type of solid manure on gaseous emissions [6]. Finally, despite the impact on environment, there is still a lack of data on the practices of the field heap storage.

Acknowledgements

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References

[1] Agreste (2014) : Cheptels selon la taille du troupeau, Rubrique 04_RA_001A : Recensement agricole 2010. Données en ligne - Disar. https://stats.agriculture.gouv.fr/disar/

[2] Ifip (2014) : Composim, calculateur de la quantité et composition des effluents d'élevage. Notice d'emploi et Guide Méthodologique. 35p.

[3] Agreste (2010) : Enquête production avicole 2008. Production de volailles et d'œufs détaillée par catégories de produits. http://agreste.agriculture.gouv.fr/IMG/pdf_aviculture2010T12.pdf

[4] Corpen (2006) : Estimation des rejets d'azote, phosphore, potassium, calcium, cuivre, zinc par les élevages avicoles. Brochure du Corpen, 55p

[5] Biomasse Normandie (2002) : Evaluation des quantités actuelles et futures des déchets épandus sur les sols agricoles et provenant de certaines activités lot 3 : effluents d'élevage rapport final. 90p

[6] Webb, J.; Sommer, S.; Kupper, T.; Groenestein, K.; Hutchings, N.; Eurich-Menden, B.; Rodhe, L.; Misselbrook, T.; Amon, B. (2012): Emissions of Ammonia, Nitrous Oxide and Methane During the Management of Solid Manures. In: Lichtfouse, E. (Ed.), Agroecology and Strategies for Climate Change. Springer Netherlands, pp. 67-107.

TC-P_12 Environmental benefits of amending cow slurry with the nitrification inhibitor dicyandiamide

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1. Objectives

Housing cattle generates large quantities of slurry. Besides being a valuable source of nutrients, slurry can also be a source of pollution after field application (nitrous oxide and methane emission). The objective of this research was i) to investigate if the nitrification inhibitor dicyandiamide (DCD) degrades in cattle slurry during anaerobic storage (soil research has long shown that DCD degrades quickly in soil but the stability of DCD in slurry is unknown) and ii) to test whether anaerobic cattle slurry amended with DCD long term during storage can be considered as a new effective and practical mitigation measure to reduce greenhouse gas (GHG) emissions from landspreading of slurry.

2. Methodology

2.1 DCD stability

Experimental units consisted of cups filled with 40 mL of cattle slurry (5% dry matter) and organised as follows: two treatments (slurry no DCD, slurry + DCD) × five sampling times (1, 6, 13, 22 and 41 days) × four replicates incubated under anaerobic conditions at 15°C (randomized block design). In the DCD solution treatment, 1 mL of a 18.2 g L^{-1} DCD solution was added to slurry (1 mL of deionised water was added to the slurry control). DCD was extracted from all treatment × time combinations with water. DCD concentration was measured by HPLC analysis and UV-Vis detection (215 nm) [1]. The relationship between time and DCD concentration was examined by regression analysis.

2.2 GHG emissions

The field experiment was conducted in autumn (October-November) on a permanent grassland site (Johnstown Castle, Ireland). The soil (0-10 cm) was a moderately drained loam. Three treatments were investigated: slurry no DCD (33 m^3 ha^{-1}, 96 kg total N (TN) ha^{-1}, 45 kg NH_4-N ha^{-1}), slurry + DCD (33 m^3 ha^{-1}, 18 kg DCD ha^{-1}) and an untreated control. Each treatment was replicated three times in a randomised block design. Treatments were applied with a syringe as a simulated bandspread application, with two lines of slurry 20 cm apart inside the pre-installed collar of gas chambers. Slurry from the slurry + DCD treatment was spiked with DCD and stored anaerobically (15°C) for six months before lanspreading. Gas emissions (daily and cumulative fluxes) of nitrous oxide (N_2O) and methane (CH_4) were measured using the static gas chamber method (0.4 m × 0.4 m size). Gas samples were collected on fifteen occasions over 35 days and analysed by gas chromatography (GC). Differences in cumulative net fluxes (value from untreated control subtracted) between slurry no DCD and slurry + DCD were examined by a two-way ANOVA. The size of the data set made it difficult to check that the model assumptions were met and therefore a distribution-free permutation test was used for the overall ANOVA test for treatment differences.

3. Results and discussion

3.1 DCD stability (incubation study)

DCD concentration in the slurry + DCD treatment remained relatively stable over the 41 days of the incubation study, close to the calculated initial concentration of 444 mg L^{-1}. Its variation (between 435 and 482 mg L^{-1}) with time was not significant ($P = 0.3$) (Fig. 1). Similarly, DCD in the slurry + DCD treatment that was incubated for six months (the exact same slurry was used in both incubation and field studies) and later applied to grassland plots did not show any sign of degradation as the DCD concentration (551 mg L^{-1}) remained close to the calculated initial concentration (542 mg L^{-1}). The stability of DCD in anaerobically stored (for up to six months) cattle slurry has never been investigated before to our knowledge. This finding indicates that it is feasible to directly add DCD to slurry tanks at the commencement of storage

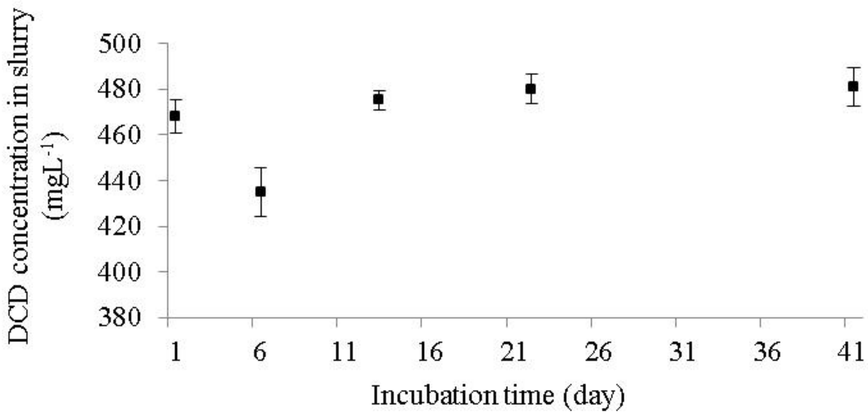
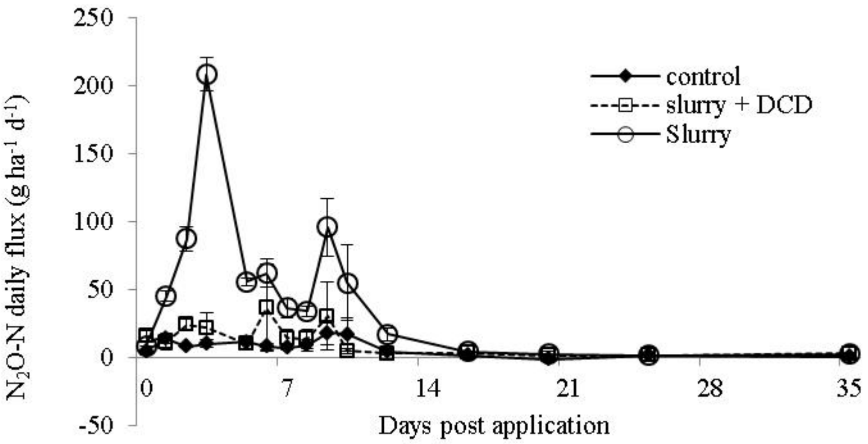


Figure 1: Variation of DCD concentration with time (± one deviation of standard error (SE), n = 4) in the slurry + DCD treatment. (All incubations were carried out at 15°C).

3.2 GHG emissions (field study)

Mean daily N_2O-N fluxes ranged in the untreated control from -1.4 to 18.0 g N ha^{-1} d^{-1}, in the slurry + DCD treatment from 1.3 to 36.4 g N ha^{-1} d^{-1} and in the slurry no DCD treatment from 1.2 to 207.9 g N ha^{-1} d^{-1} (Fig. 2a). Slurry application in the slurry no DCD treatment resulted in a sharp increase in the N_2O-N flux from the day following application until about three weeks post-application. In contrast, application of the slurry + DCD treatment did not cause any large N_2O-N peak.

a.



b.

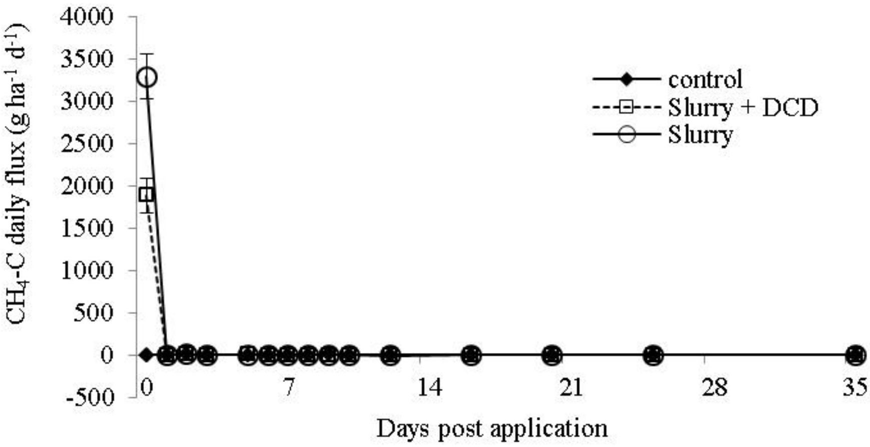


Figure 2: Mean daily greenhouse gas fluxes of a. N_2O-N and b. CH4-C (± SE, n = 3) from a pasture soil under three treatments: untreated control, slurry no DCD (33 m^3 ha^{-1}, 96 kg TN ha^{-1}, 45 kg NH_4-N ha^{-1}), slurry + DCD (15 kg DCD ha^{-1}).

The cumulative N_2O-N net losses calculated after subtracting the control value (145 g N ha^{-1}) were significantly higher (P = 0.008) in the slurry no DCD treatment (793 g N ha^{-1}) than in the slurry + DCD treatment (95 g N ha^{-1}). This result shows that i) DCD was still active and effective at lowering emissions and ii) the composition of slurry was not altered in a manner that would have caused an increased N_2O-N flux. In fact DCD reduced N_2O losses by 88%, similarly to other studies [2].

On the day of application, the daily CH_4-C fluxes were higher in the slurry no DCD treatment (3285 in g C ha^{-1} d^{-1}) than in the slurry + DCD treatment (1888 g ha^{-1} d^{-1}). Within 24 h, daily CH_4-C fluxes in these two treatments dropped to the levels observed in the control and they remained low until the end of the experiment. The cumulative CH_4-C net emissions calculated after subtracting the control value (-14 g C ha^{-1}) were higher in the slurry no DCD treatment (1634 g C ha^{-1}) than in the slurry + DCD treatment (942 g C ha^{-1}). The cumulative CH_4-C net emission in the slurry + DCD treatment were 42% lower than in the slurry treatment but this difference was not significant at the 0.05 significance level (P = 0.1). The origin of the short-lived spike of CH_4 is likely to originate from the degassing of slurry. This finding is in line with other research that previously found DCD decreased CH_4 emissions from anaerobic soils under rice production [3], but reasons for this remain unclear. Further work is required with more replicates and a less conservative statistical test to investigate the effect of DCD on CH_4 emissions.

4. Conclusion and outlook

DCD incubated in anaerobic cattle slurry for up to six months did not degrade and effectively reduced N_2O and CH_4 emissions from land-spread slurry by 88% and 43% respectively in grassland during autumn, when there is a higher risk of high N_2O emission. The results suggest that DCD can be mixed directly in slurry tanks any time before landspreading is planned, but further work will be required at a larger scale to mimic real farm storage conditions. This finding opens up opportunities for the addition of DCD to slurry and possibly other organic farmyard wastes during storage as an effective GHG migration measure following landspreading.

Acknowledgements

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References

[1] Turowski, M.; Deshmukh, B. (2004): Direct chromatographic method for determination of hydrogen cyanamide and dicyandiamide in aqueous solutions, Analytical Letters 9, 1981–1989

[2] Calahan, E.; Ernfors, M.; Müller, C.; Devaney, D.; Laughlin, R.J.; Watson, C.J.; Hennessy, D.; Grant, J.; Khalil, M.I.; McGeough, K.L.; Richards, K.G. (2014): The effect of the nitrification inhibitor dicyandiamide (DCD) on nitrous oxide and methane emissions after cattle slurry application to Irish grassland, Agriculture, Ecosystems and Environment 199, 339–349

[3] Malla, G.; Bhatia, A.; Pathak, H.; Prasad, S.; Jain, N.; Singh, J. (2005): Mitigating nitrous oxide and methane emissions from soil under rice–wheat system with nitrification and urease inhibitors, Chemosphere 58, 141–147

TC-P_13 Can earthworms reduce greenhouse gas emissions from composting of urban waste?

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1. Objectives

The present study has two objectives: (i) to quantify greenhouse gas emissions and N losses from conventional composting and vermicomposting and link this with properties of starting material; (ii) to determine the effect of earthworm abundance on greenhouse gas emission from vermicomposting of urban waste.

2. Methodology

Two independent experiments were carried out to address the two objectives. The first experiment was conducted to quantify N losses and greenhouse gas emission during composting and vermicomposting. On the other hand, the second experiment was initiated mainly to determine the effect of earthworm density on greenhouse gas emissions during vermicomposting of different urban waste.

Experiment I: Vegetable waste was collected from the food processing industry, and mixed thoroughly with wheat straw at two ratios namely; 5:1 and 10:1 (vegetable:straw) to produce starting material with different carbon to nitrogen ratios. The chemical properties of the starting materials are presented in Table 1. Conventional composts were then prepared from the two mixtures (mixture A and B) (Table 1) in 60 litres polyethylene reactors. 5 kg (fresh weight basis) of each mixture was put in the reactors, and the moisture content was adjusted to 50-60 % by spraying water. The reactors were insulated with 5cm foam to minimize heat loss during composting. Vermicompost was prepared from the same materials (Table 1). Adult earthworms (Eisenia fetida) were added in each container at stocking rate of 2 kg earthworm m-2 (equivalent to 350 g earthworms per container). The moisture content in the vermicompost bin was kept at moisture content of 80-85 %.

Experiment II: In order to determine the effect of earthworm abundance on N losses and greenhouse gas emissions, vermicompost was prepared from different quality of urban and agricultural waste (i.e vegetable waste, pre-decomposed cattle manure and wheat straw) (Table 1). Vermicompost was then prepared as described earlier in experiment I, and adult Eisenia fetida were applied at stocking rates of 1 kg m-2 and 3 kg m-2, which were equivalent to 175 and 350 g earthworms per container, respectively.

Table 1: The characterstics of starting materials used in the experiment.

Treatment Code	Mixing ratio [1] (vegetable:straw)	Total C	Total N	C/N	N-NH_4	N-NO_3
		--------- g kg^{-1} DM---------			-------- mg kg^{-1} DM----------	
		Experiment I				
Mix A	5:1	443.8	14.5	30.6	6220.5	119.7
Mix B	10:1	425.9	17.9	23.8	8160.5	91.5
	Experment (vegetable:cattle manure: straw)					
Mix A	5:0:1	443.8	14.5	30.6	6220.5	119.7
Mix B	10:0:1	425.9	17.9	23.8	8160.5	91.5
Mix C	4 :1: 0.25	391.4	19.5	20.1	1038.8	269.0
Mix D	3:1:0	382.1	26.9	14.2	1440.3	478.0

[1] fresh weight basis; DM = Dry matter

Gas sampling

Gas samples were taken from the sealed composting reactors ??????? every two days for the first week and twice a week until the end of the experiment. Gas samples were collected from five time points (0, 20, 40, 60 and 80 minutes), and analyzed using Gas chromatography (Bruker 450-GC 2011). The emission rate in mg kg^{-1} drymatter day^{-1} was calculated as:

$$Gas\ flux = \left(\frac{\Delta C}{\Delta t}\right) * \left(\frac{V}{A}\right) * \left(\frac{M}{Vs}\right) * \left(\frac{P}{Po}\right) * \left(\frac{273}{T}\right) * 60\ min * 24\ hr\ * \left(\frac{A}{W}\right) \qquad \text{Eq.1}$$

Where ΔC is the change in concentration of gas in time interval Δt, V and A are the headspace volume (liter) and reactor surface area (m^2), M is the molecular weight of the gas interest (16, 44 and 44 for CO_2, CH_4 and N_2O, respectively), Vs is the volume occupied by 1 mole of a gas at standered temprature and pressure (22.4 liter), P is the atmospheric pressure (mbar), Po is the standard pressure (1013 bar), T is the tempraure inside the chanber during the deployment time in Kelvin, W is the initial dry matter of the composting material.

Cummulative fluxes in mg kg^{-1} dry matter were then calculated using trapezoid technique as:

$$Cummulative\ flux\ = \frac{(fa+fb)*(db-da)}{2} \qquad \text{Eq.2}$$

Where *fa* and *fb* are fluxes in mg kg^{-1} dry matter day^{-1} at day *a* and *b*; *db* and *da* are the dates of measurements.

How was CO_2 equivalent calculated – what GWPs were used for CH_4 and N_2O? Probably best to include this briefly here in the methods section.

3. Results and discussion

3.1. Greenhouse gas emissions from composting and vermicomposting

The cumulative greenhouse gas emissions from different composting methods and starting materials are presented in Table 2. The cumulative C-CO_2 emission was significantly ($P<0.01$) effected by different composting methods, mixing ratios and their interaction. Use of earthworms for composting increased cumulative C-CO_2 by 40 % in mixture A and 9 % in mixture B compared to conventional composting. High C-CO_2 emission from vermicompost indicated higher decomposition and carbon turnover in earthworm working substrates than conventional compost. The effect of earthworms on C-CO_2 emission was more noticeable in mixture A (40%), and it may sugessted that earthworms are able to feed on relatively recalcitrant carbon which have been difficult to decompose by microorganisms. The C-CO_2 emission in the present study was consistence with the increase in temperature in both composting methods. During conventional composting and vermicomposting, peak C-CO_2 emission was observed within 24 hrs, and it decreased gradually with composting time. The decrease in C-CO_2 emission with composting time indicates utilization of readily degradable carbon compounds by microorganisms [1].

The peak C-CH_4 emission was observed within few days (< 2 days), and it decreased rapidly thereafter. The decrease in C-CH_4 emission with time was rapid during conventional composting compared to vermicomposting. The sharp decrease in C-CH_4 production was observed after 4 days during conventional composting while it was observed after 10 days during vermicomposting. Similar patterns of C-CH_4 emission have been reported during conventional composting of animal manure [1, 2]; however, C-CH_4 production pattern during vermicomposting has not been studied previously which could support or contrast with our findings. Our study demonstrated that cumulative C-CH_4 emission was not significantly different ($P>0.05$) between composting methods; however, the effect of starting material on C-CH_4 emission was highly significant ($P<0.01$). Cumulative C-CH_4 production in mixture A was lower than Mixture B, and could have resulted from better air circulation in mixture A (i.e highly straw amended mixture). Moreover, relatively high concentration of easily available carbon in mixture B could be responsible for high C-CH_4 production. This study also showed that use of earthworms for composting decreased cumulative C-CH_4 by 22% and 26% in mixture A and mixture B, respectively.

The cumulative N-N_2O emission was significantly different ($P<0.05$) between composting methods and mixing ratio of starting materials (Table 2). The highest N-N_2O emission was observed during conventional composting and when bulking agent was added in small quantity (mixture B). Use of earthworms decreased N-N_2O emissions by 25% and 36% in mixture A and B, respectively. The possible explanation for low N-N_2O emission from vermicompost is due to burrowing activities of earthworms that offset N_2O production.

Table 2: Cumulative greenhouse gases emission after 40 days of household waste composting and vermicomposting.

Treatments code	C-CO_2	N-N_2O	C-CH_4	CO_2 equivalent (kg tonne^{-1} DM)			
	kg tonne^{-1} DM			N_2O	CH_4	Total (CO_2 not included)	Total (CO_2- included)
T_mix A	111.8 (4.25)	0.008	1.0 (0.05)	2.4	25.8 (1.25)	28.3 (1.92)	140.1 (6.05)
T_mix B	123.7 (2.60)	0.066	1.3 (0.08)	19.7	31.5 (2.00)	51.2 (2.16)	174.9 (9.42)
V_mix A	157.5 (0.83)	0.006	0.8 (0.07)	1.8	20.0 (1.75)	21.8 (2.64)	179.4 (1.81)
V_mix B	134.5 (3.31)	0.042	0.9 (0.20)	12.6	23.3 (5.00)	35.8 (1.56)	170.3 (2.13)
ANOVA							
Mix	***	***	**	***	**	*	***
Method	**	*	NS	*	NS	***	***
Method*mix	***	NS	NS	NS	NS	NS	***

Values in parentheses indicated standard error of the mean (n=3); ***, **,* represent significant difference at P<0.001; P<0.01 and P<0.05, respectively; NS = non significant difference; T = conventional composting; V= vermicomposting; mix A = 5:1 (waste:straw ratio); mix B = 10:1 (waste:straw ratio); DM = Dry matter

3.2. Carbon and nitrogen mass balances during composting and vermicomposting

Table 3 indicates C and N mass balances during composting and vermicomposting of urban waste. The C loss accounted about 36 % and 40 % of the initial carbon content during conventional composting and vermicomposting, respectively. Carbon loss as C-CO_2 accounted 25-30 % of initial total C during conventional composting while it ranged between 31-33 % during vermicomposting. The C loss as C-CH_4 was very small in all substrates regardless of composting method. Only 0.19-0.22 % and 0.17-0.20 % of initial total C was lost from conventional composting and vermicomposting, respectively. Our study showed that vermicomposting reduced N loss significantly (P<0.05) compared to conventional composting. The use of earthworms for composting reduced N loss by 10 % in mixture A and 21 % in mixture B.

Table 3: Carbon and nitrogen mass balances during composting and vermicomposting of household waste

Treatments code	C balance				N balance		
	Total C Retained	C loss (C-CO_2)	C-loss (C-CH_4)	Carbon unaccounted	Total N retained	N loss (N-N_2O)	N-unaccounted
	---------------------% of initial carbon----------------				-------% of initial nitrogen------		
T_mix A	63.3	25.0	0.2	11.5	81.1	0.1	18.8
T_mix B	60.8	30.3	0.2	8.7	76.8	0.4	22.8
V_mix A	59.5	33.6	0.2	6.7	83.6	0.04	16.3
V_mix B	58.4	31.3	0.2	10.1	81.5	0.3	18.2
ANOVA							
Method	**	**	NS	NS	*	*	*
Mix	*	*	NS	NS	**	**	NS
Method*mix	NS	**	NS	NS	NS	NS	NS

Values in parentheses indicated standard error of the mean (n=3); **,* represent significant difference at P<0.01 and P<0.05, respectively; NS = non significant difference; T = conventional composting; V= vermicomposting; mix A = 5:1 (waste:straw ratio); mix B = 10:1 (waste:straw ratio); N = Nitrogen; C = Carbon; DM = Dry matter matter

3.3. Effect of earthworm abundance on greenhouse gas emissions

The analysis of variance showed that increasing earthworm density increased cumulative C-CO_2 emission from vermicompost significantly (P<0.05) (Table 4). This indicates high decompostion rates of substrates with increasing earthworm density. Increasing earthworm density increased cummulative C-CO_2 emission by 4%, 14%, 2% and 5% in mixture A, B , C and D, respectively. Cummulative C-CH_4 emission varied significantly (P<0.001) between starting materials; however, a significant (P>0.05) effect of earthworm density on cumulative C-CH_4 emission was not observed. The present study also showed that substrate quality influenced cumulative N-N_2O

emission ($P<0.001$), but no significant effect of earthworm density (Table 4). The highest N-N_2O emission was observed during vermicomposting of pre-decomposed cattle manure (mixture C and D), and this could be due to high N-NO_3 concentration in cattle manure (Table 1), which is the most important factor for production of N_2O through denitrification process. Increasing earthworm density (3 kg m^2) increased N_2O emission by 60% and 33% in mixture B and D, respectively compared to low density (1 Kg m^2). Interestingly, when more straw was added (i.e mixture A and C), the effect of earthworm density on N_2O emission was negligible.

Table 4: Cumulative greenhouse gases emission and total greenhousegas emission after 45 days of vermicomposting of household waste.

Treatments code	C-CO_2	N-N_2O	C-CH_4	CO_2 equivalent (Kg tonne^{-1} DM)			
	kg tonne^{-1} DM			N_2O	CH_4	Total (CO_2 not included)	Total (CO_2- included)
VH- Mix A	155.0	0.007	0.9	2.1	23.0	25.1	180.1
VH- Mix B	129.5	0.03	1.8	10.3	43.8	53.9	183.3
VL- Mix A	149.4	0.007	1.3	2.1	32.0	34.1	183.5
VL- Mix B	113.3	0.02	2.0	4.8	49.5	54.3	167.5
VH- Mix C	116.7	0.05	0.08	14.6	2.0	16.6	133.3
VH- Mix D	95.4	0.2	0.09	56.6	2.3	58.9	154.3
VL- Mix C	113.9	0.05	0.10	15.5	2.5	18.0	131.9
VL- Mix D	90.6	0.1	0.14	41.7	3.5	45.2	135.8
ANOVA							
Mixture	**	**	***	***	***	***	***
Density	*	NS	NS	NS	*	NS	*
Density* mixture	NS	NS	NS	NS	NS	NS	NS

Values in parentheses indicated standard error of the mean (n=3); ***, **,* represent significant difference at $P<0.001$; $P<0.01$ and $P<0.05$, respectively; NS = non significant difference; VH = earthworm density at 3 kg m^{-2}; VL = earthworm density at 1 kg m^{-2}; mix A = 5:1 (waste:straw ratio); mix B = 10:1 (waste:straw ratio); mix C = 4:1:1/4 (waste:manure:straw ratio); mix D =3:1:0 (waste:manure:straw mixture); DM = Dry matter matter

4. Conclusions

The present study demonstrated that use of earthworms for composting is an effective method to reduce total N loss and greenhouse gas emissions. Use of earthworms for composting has a significant potential to reduce N_2O and CH_4 emissions, and we observed that earthworms reduced N_2O emission more effectively in low C:N ratio substrates. This study also showed that earthworm density does not influence CH_4 and N_2O emissions from vermicompost; however, it was possible to observe that higher earthworm density accelerated the decomposition rate (C-CO_2 emission). Furthermore, our study showed that vermicomposting is an efficient method to accelerate the decomposition of waste with high C:N ratio. In general, N loss and GHG emissions from vermicompost decreased when straw is added as bulking agent.

Acknowledgements

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References

[1] Chowdhury, A.; de Neergaard, A.; Jensen, LS. (2014): Composting of solids separated from anaerobically digested animal manure: Effect of different bulking agents and mixing ratios on emissions of greenhouse gases and ammonia, Biosystems Engineering 124, 63-77

[2] Vu, QD.; de Neergaard, A.; Tran, TD.; Hoang, HT.; Vu, TK.; Jensen, LS, (2014): The effect of the biochar and different crop residues on greenhouse gas emission during passive aeration composting of manure and digestate typical of small scale livestock farms in Vietnam, Environmental Technology 30, 1-12

TC-P_14 Evaluation of reduction effect by different kind of soil injection in bare soil

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1. Objectives

Shallow closed slot injection of slurry in bare soil is an effective method to reduce ammonia emission from land spreading of animal slurry (Moseley et al., 1998; Nyord et al., 2010). In Denmark, injection of slurry in bare soil is traditionally done by a robust spring tine mounted with an attached slurry hose, placing the slurry in the slot created by the tine. However, on areas with huge amounts of crop residues on the soil surface, e.g. maize stubble and residues from catch crops, the risk of residues building up in front of the tines is high. Samson Agro (Vestermarksvej 25, DK-8800, Viborg, Denmark) has developed a new injector system based on disc coulter tines, which cuts the crop residues and covers the applied slurry effectively without mixing the slurry with the soil.

In corporation with Samson Agro a series of trials were conducted in autumn 2014. The aim of the trials was to examine the optimal setting of the discs compared to the slurry hoses in order to get the maximum reduction of ammonia emission.

2. Methodology

The trial was made first time in the beginning of November 2014 and repeated two weeks later. Field trials were carried out at Aarhus University, Research Centre Foulum on loamy sand (9 % clay, 11.2 % silt, 40.5 % fine sand 34.8% coarse sand and 4.4 % organic matter). Soil water content was 23.9 and 24.2% (vol/vol) respectively at the experiment 1 and 2, which was close to field capacity. The soil surface was covered by cereal stubbles. Slurry was applied with a commercial slurry trailer from Samson Agro equipped with a new developed disc injector with the discs mounted in two rows. Slurry applied was cattle slurry with the following characteristics: pH = 7.3, total nitrogen (N) = 3.6 g/l, ammonia N = 2.5 g/l, content of dry matter = 5.6 %. The following four treatments were compared:

1. Surface application, no incorporation (reference)
2. Slurry injected right after the discs at the first row where slurry was placed in the slit
3. Slurry placed randomly on the soil surface in front of the discs at the front row
4. Slurry placed exactly between the discs on the soil surface in front of the discs at the front row

Experimental setup of the slurry injector is shown in Figure 1.

Figure 1: Disc slurry injector. To the left: slurry applied in front of the discs which could either bee between discs or right in front of the discs. To the right; slurry placed right after the discs.

In both experiments dynamic chambers are used to supply a constant air flow flushing soil surface and to simulate the wind effects on gaseous emissions. The dynamic chambers were designed on the basis of CFD modelling ensuring that the wind profile with in the chamber was relatively uniform with an air exchange of 14 times per minute. The chambers were 0.5*0.5*1.75 m height,

width and length respectively. In front of the chambers a 0.5 m long funnel with an opening of 0.2*0.2 m ensured relatively uniform wind movement through the chamber, for more details see (Nyord et al., 2012). Ammonia emission was measured 72 hours after application (two replications per treatment) by impingers (gas washing bottles), with 50 ml 0.02 M boric acid solution which were changed after 1, 4, 8, 24, 48 and 72 hours following slurry application.
30 ton of slurry was applied per hectare. Discs on the injector were operating in 10 cm below soil surface. Air temperature in the measuring periods varied between 2.2 and 8.9 ºC. Weather conditions were relatively similar in the two experiments.
Differences in ammonia emission depending of the application method were tested using ANOVA. Pairwise multiple comparisons with trails hose application according to the Holm–Sidak method to isolate treatments that differed from the others, using SigmaPlot 12.5 (Systat Software Inc., 2011).

3. Results and discussion

Ammonia emissions following surface application are measured at relatively low level in this trial, especially when it is considered that the emission is determined by the wind tunnel method, which has been reported to overestimate the emissions (Misselbrook et al., 2005). Here it should be taken into consideration that the air temperature generally was low in the measuring periods. As seen in Figure 2, ammonia loss pattern between treatments were similar between the two trials and could be explained by the almost identical soil and climate circumstances in the experiments. It indicates that the measuring method is likely to have worked successfully at both trials and that the effect of the incorporation is consistent.

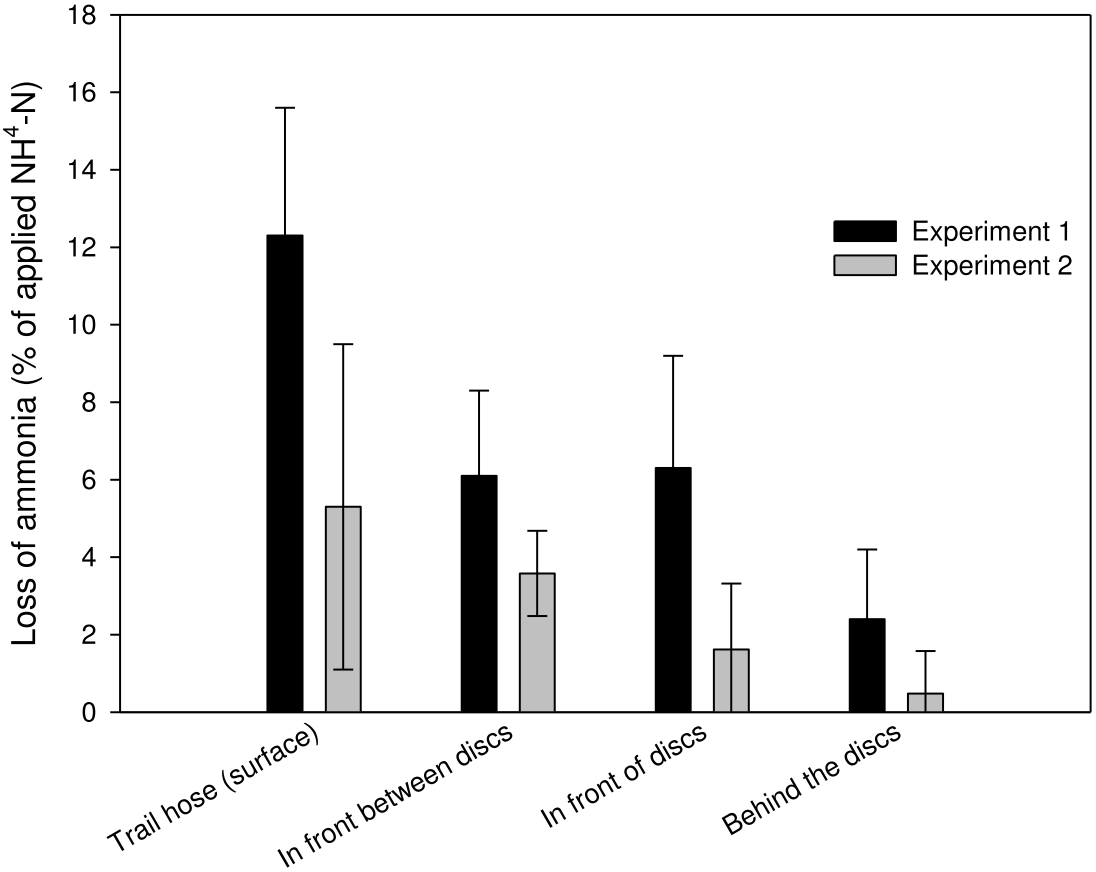

Figure 2: Average ammonia emission given as % of applies ammoniacal N. Black bars represent the loss at the first experiment and grey the second experiment. Error bars indicate standard deviation (n=2).

Figure 3 shows, that the loss is approximately halved by injection, when slurry is placed in front of the discs, in comparison with trail hose application. This is in good correspondence with the 62 %

reduction compared to surface applied slurry, found in (Wulf et al., 2002). Here, cattle slurry was incorporated right after surface application at a working depth of approximately 10 cm, which is similar to the treatments where slurry was placed in front of the disc injector.

Slurry placed behind the discs during injection reduces ammonia losses even greater to about 85 % of the loss following trail hose application. There is no statistically significant difference at the 95% level, between the three injection methods. However, the trend is clear: slurry placed behind the discs reduces ammonia loss more than placement in front of the discs. The explanation is probably that the slurry placed in front of the discs will be placed throughout the entire volume of the worked soil, leaving a proportion of the slurry on the soil surface or in the very top soil. From this ammonia can evaporate especially in a dynamic chamber setup with no precipitation in the measuring period possible, due to the covering of the emission area. Precipitation could lead to deeper infiltration of the ammonia N into the soil and thereby preventing evaporation. Opposite when slurry is placed behind the discs in the slot created by the discs on the first row and then covered by soil from the discs mounted on row two. This kind of injection is more similar to the system described in (Balsari et al., 2005; Weslien et al., 1998). In these papers, an emission reduction of 84 and 94% respectively, compared to surface application, was reported, which corresponds well with the 84% found in this trail.

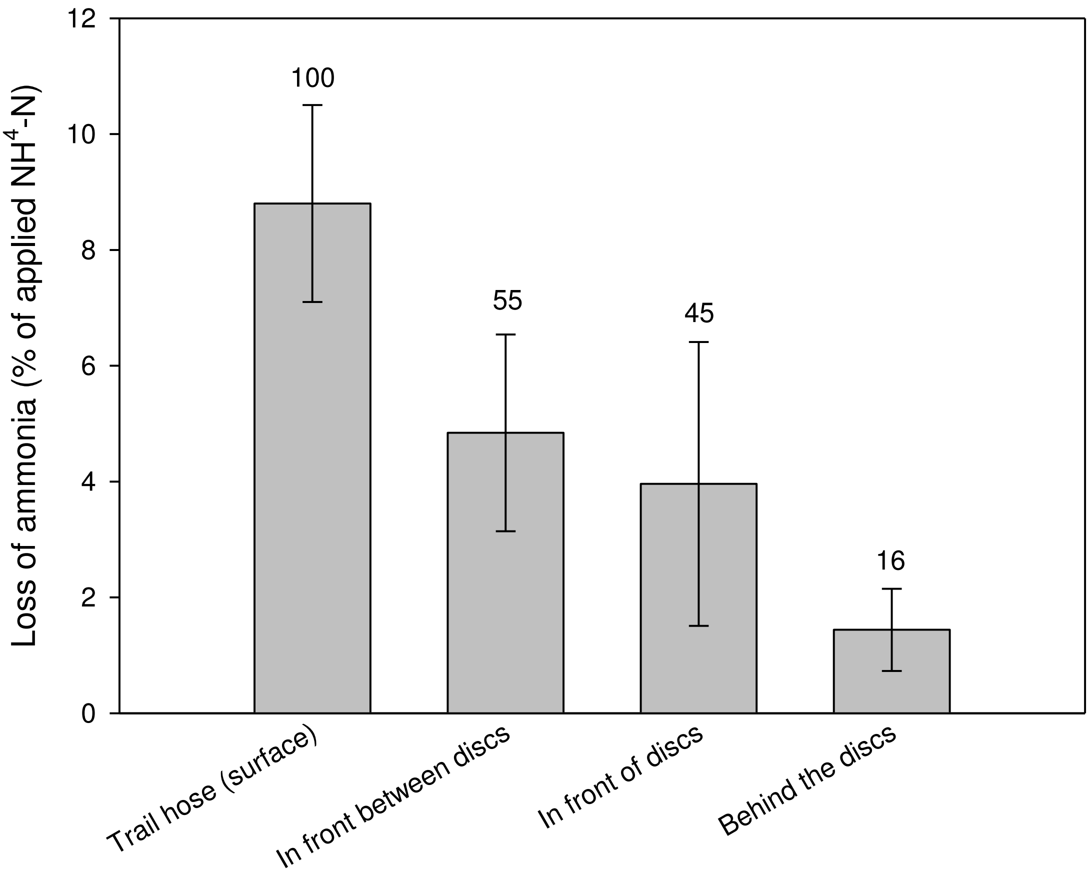

Figure 3: Average ammonia emission given as % of applied ammonia N. Numbers indicates average reduction in emission relative to Trail hose application. Letters indicates significant differences (95 % level). Error bars indicate standard deviation (n=4).

4. Conclusion

Soil injection of cattle slurry was found to be very effective to reduce ammonia emission compared to surface application. The most effective injection method seems to be placement of the slurry right behind the discs. However, no significant difference was found I comparison with the two treatments were slurry was placed in front of the discs.

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References

[1] Balsari,P., G.Airoldi, and F.Gioelli. 2005. Improved recycling of livestock slurries on maize by means of a modular tanker and spreader. Bioresource Technology 96:229-234.
[2] Misselbrook,T.H., F.A.Nicholson, B.J.Chambers, and R.A.Johnson. 2005. Measuring ammonia emissions from land applied manure: an intercomparison of commonly used samplers and techniques. Environmental Pollution 135:389-397.
[3] Moseley,P.J., T.H.Misselbrook, B.F.Pain, R.Earl, and R.J.Godwin. 1998. The Effect of Injector Tine Design on Odour and Ammonia Emissions following Injection of Bio-solids into Arable Cropping. Journal of Agricultural Engineering Research 71:385-394.
[4] Nyord,T., M.N.Hansen, and T.S.Birkmose. 2012. Ammonia volatilisation and crop yield following land application of solidGÇôliquid separated, anaerobically digested, and soil injected animal slurry to winter wheat. Agriculture, Ecosystems & Environment 160:75-81.
[5] Nyord,T., E.F.Kristensen, L.J.Munkholm, and M.H.Jorgensen. 2010. Design of a slurry injector for use in a growing cereal crop. Soil & Tillage Research 107:26-35.
[6] Weslien,P., L.Klemedtsson, L.Svensson, B.Galle, A.Kasimir-Klemedtsson, and A.Gustafsson. 1998. Nitrogen losses following application of pig slurry to arable land. Soil Use and Management 14:200-208.
[7] Wulf,S., M.Maeting, and J.Clemens. 2002. Application technique and slurry co-fermentation effects on ammonia, nitrous oxide, and methane emissions after spreading: I. Ammonia volatilization. J Environ Qual 31:1789-1794.

TC-P_15 Animal feeding strategies to abate nitrous oxid and ammonia emission from surface applied slurry to a grassland soil

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1. Objectives

It is estimated that nitrogen (N) losses from fertilized crops range between 50-70% of applied N, depending on management practices, climate and soil conditions. Ammonia (NH_3) emissions following land application of animal manures give rise to a significant proportion of the total NH_3 emissions from agricultural sources [1]. Additionally, the increased amount of N and soluble organic carbon (SOC), associated with the application of slurries, may favour the emission of nitrous oxide (N_2O) and also affect the balance of other greenhouse gases (GHG) such as carbon dioxide (CO_2) (e.g. [2]). A large effort has been made to develop cost-effective strategies to reduce N losses avoiding, as much as possible, pollution swapping. In recent years, animal feeding strategies have been developed aiming to increase the N use efficiency of agrosystems. Most of this research has focused on NH_3 abatement but little is known about the combined effect of animal feeding on NH_3 and GHG emissions following slurry application [3]. The main objective of this study was to evaluate the effect of five different feeding strategies on the emissions of NH_3, N_2O and CO_2 from a grassland soil fertilized with pig slurries.

2. Methodology

A greenhouse experiment was carried out in the ETSI Agrónomos of the Technical University of Madrid between 29th November and 8th January. Five types of pig slurry (Table 1) were surface-applied in a complete randomized block design with three replicates to a grassland (*Lollium perenne*) soil (*Calcic Haploxerepts*), previously collected from the field, air dried and sieved (2 mm). This soil has a clay loam texture (28% clay, 17% silt, and 55% sand) in the upper horizon (0-28 cm). The soil was packed after dried into small PVC containers (26 cm inner diameter). A soil with no fertilizer applied was used as a control. Headspace samples for GHG analysis were taken following the procedure of Abalos et al. [4] from a closed static chamber (7.96 l). Concentrations of GHG were determined by gas chromatography. Ammonia emissions were measured by a dynamic chamber (same dimensions as for GHG) connected to a chemiluminescence analyser.

Statistical analysis was performed using Statgraphics Plus 5.1 (Manugistics 2000). The normal distribution of the data was checked using the Kolmogorov–Smirnov test. Differences between treatments were analyzed using analysis of variance (ANOVA,$p<0.05$). The least significant difference (LSD) test was used for multiple comparisons between means. For non-normally distributed data, the Kruskal–Wallis test was used on non-transformed data to evaluate differences at$p<0.05$. Schaich–Hamerle analysis was then used as a post hoc test.

Table 1: N applied, N content and pH of applied slurries. “PSControl” refers to a slurry coming from pigs fed with barley (40%), wheat (45%) and soybean (10%); “Ga” refers to slurries produced by pigs fed with a mixture of “Garrofa” a by-product of the Carob tree (*Ceratonia siliqua*) (7.5 and 15% for Ga7.5 and Ga15, respectively); wheat (45%); barley (28 and 16 % for Ga7.5 and Ga15, respectively) and soybean (13 and 16 % for Ga7.5 and Ga15, respectively). “Pn” refers to slurries produced by pigs fed with a mixture of orange pulp (7.5 and 15% for Pn7.5 and Pn15, respectively); wheat (45%); barley (30and 20%, respectively) and soybean (12 and 14 % for Pn7.5 and Pn15, respectively).

Treatment	N applied (kg N ha^{-1})	Total N (g kg^{-1})	Ammonium N (g kg^{-1})	pH
PSControl	100	9.91	5.32	8.89
Ga 7.5	100	7.97	3.75	8.38
Ga 15	100	9.72	4.35	8.20
Pn 7.5	100	8.71	4.53	7.93
Pn 15	100	7.84	3.41	8.08

Soil samples were taken ten times during the experiment to determine the concentrations of mineral N (ammonium, NH_4^+ and nitrate, NO_3^-) as well as SOC. The total biomass of plants was also measured and total N and C determined.

Finally, additional analysis was carried out at the School of Environment, Natural Resources and Geography (Bangor University) on the liquid fraction of the slurry to quantify the content on certain compounds that have been shown to be potential nitrification inhibitors (e.g. hippuric and benzoic acid).

3. Results and discussion

Application of slurries increased ($P<0.05$) N_2O emission in all cases compared to the control (from 98% to 96% for PSControl and Pn7.5, respectively). The type of pig slurry had an effect on these emissions. Incorporation of by-products in the animals’ diet decreased ($P<0.05$) N_2O emissions from applied slurry by 37 and 56% for Ga and Pn, respectively (Figure 1). This was probably related to the higher NH_4^+-N of PSControl (Table 1) which would have been probably nitrified. Nitrification was thought to be the main pathway in the production of N_2O since soil conditions were favorable to this microbial process (average WFPS of 63%). Additionally, the concentration of NO_3^- was negatively correlated with that of NH_4^+ (R^2 0.96) and positively with N_2O emissions (R^2 0.89).

Application of slurries enhanced soil respiration (i.e. CO_2 fluxes) in all cases (64% on average), although lower for PSControl (35%) (data not shown), possibly due to a reduction of plant biomass as a result of plant damage following slurry application.

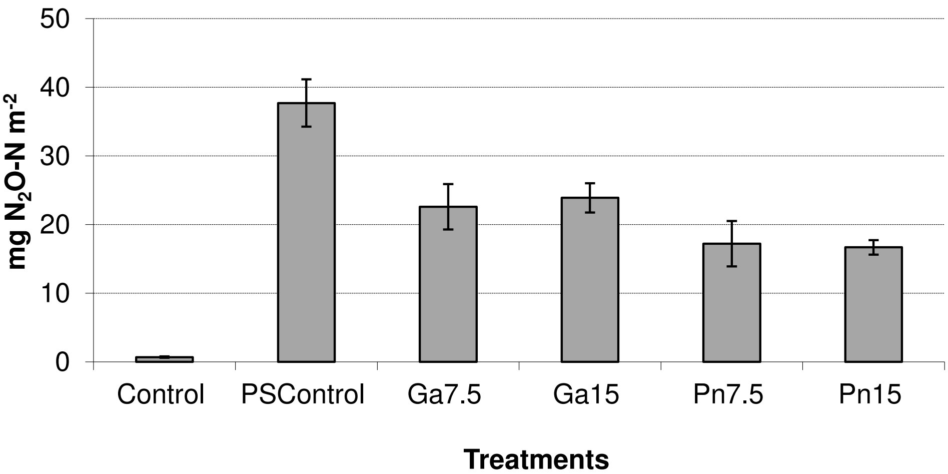

Figure 1: Cumulative N_2O emissions (mg N_2O-N m^{-2}) over the 42 day experiment. “PSControl” refers to a slurry coming from pigs fed with barley (40%), wheat (45%) and soybean (10%); “Ga” refers to slurries produced by pigs fed with a mixture of “Garrofa” a by-product of the Carob tree (*Ceratonia siliqua*) (7.5 and 15% for Ga7.5 and Ga15, respectively); wheat (45%); barley (28 and 16 % for Ga7.5 and Ga15, respectively) and soybean (13 and 16 % for Ga7.5 and Ga15, respectively). “Pn” refers to slurries produced by pigs fed with a mixture of orange pulp (7.5 and 15% for Pn7.5 and Pn15, respectively); wheat (45%); barley (30and 20%, respectively) and soy-bean (12 and 14 % for Pn7.5 and Pn15, respectively).

Control is soil without any N fertilizer added. Ammonia emissions were also enhanced by slurry application. Similarly than with N_2O, PSControl produced the highest emissions (36 and 23% higher than Ga and Pn, respectively) (Figure 2). Again the higher NH_4^+ concentration measured in soil treated with PSControl may have led to increased volatilization rates.

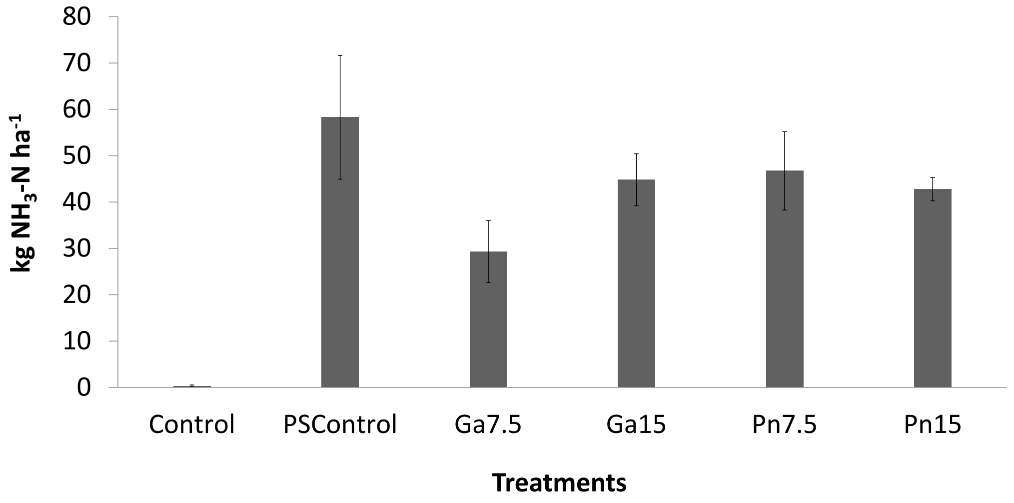

Figure 2: Cumulative NH_3 emissions (kg NH_3-N ha^{-1}) over the 42 day experiment**.** “PSControl” refers to a slurry coming from pigs fed with barley (40%), wheat (45%) and soybean (10%); “Ga” refers to slurries produced by pigs fed with a mixture of “Garrofa” a by-product of the Carob tree (*Ceratonia siliqua*) (7.5 and 15% for Ga7.5 and Ga15, respectively); wheat (45%); barley (28 and 16 % for Ga7.5 and Ga15, respectively) and soybean (13 and 16 % for Ga7.5 and Ga15, respectively). “Pn” refers to slurries produced by pigs fed with a mixture of orange pulp (7.5 and 15% for Pn7.5 and Pn15, respectively); wheat (45%); barley (30and 20%, respectively) and soybean (12 and 14 % for Pn7.5 and Pn15, respectively). Control is soil without any N fertilizer added.

Several authors have observed an effect of certain N compounds present in the urine (e.g. hippuric acid) over both NH_3 and N_2O losses from urine treated soils. Whitehead et al. ([5]) provided indirect evidence that urine composition may affect N_2O emissions. They observed that the presence of hippuric acid in urine caused the higher NH_3 volatilization among the tested treatments. On the other hand, Kool et al. ([6]) measured decreased N_2O emissions in soils treated with urine containing relatively high amounts of hippuric acid. In our study, PN slurries were those with the higher contents of hippuric acid (data not shown), and resulted in a non-significant ($P>0.05$) effect on NH_3 emissions but a significant ($P<0.05$) effect in reducing N_2O losses. Conclusion

Inclusion of by-products from the Mediterranean agro-food industry in the animals’ diet could effectively decrease both N_2O and NH_3 emissions from N application. Partial substitution of soybean and barley by “garrofa” and orange pulp in the diet reduced NH_3 and N_2O from slurry application under controlled conditions. These preliminary results show the potential of alternative feeding strategies to reduce some of the environmental problems associated with agriculture, and for decreasing the external dependency of N imports for feeding animals in Spain. Further research under field conditions is needed to confirm these results.

Acknowledgements

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References

[1] Misselbrook, T.H., Smith, K.A., Johnson, R.A., Pain, B. F., 2002. Slurry application techniques to reduce ammonia emissions: Results of some UK field-scale experiments. Biosystems Engineering 81, 313-321.

[2] Amon B., Kryvoruchko V., Amon T., Zechmeister-Boltenstern S., 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agriculture, Ecosystems and Environment 112, 153-162.

[3] Arriaga H., Salcedo G., Calsamiglia S., Merino P., 2010. Effect of diet manipulation in dairy cow N balance and nitrogen oxides emissions from grasslands in northern Spain. Agriculture, Ecosystems and Environment 135, 132-139.

[4] Abalos D., Sanz-Cobena A., Garcia-Torres L., van Groenigen J.W., Vallejo A., 2013. Role of maize stover incorporation on nitrogen oxide emissions in a non-irrigated Mediterranean barley field. Plant and Soil 364, 357–371.

[5] Whitehead, D.C., Lockyer, D.R., Raistrick, N., 1989. Volatilization of ammonia from urea applied to soil: influence of hippuric acid and other constituents of livestock urine. Soil Biology & Biochemistry 21, 803–808.

[6] Kool, D.M., Hoffland, E., Hummelink, E.W.J., van Groenigen, J. W., 2006. Increased hippuric acid content of urine can reduce soil N_2O fluxes, Soil Biology & Biochemistry 38, 1021-1027

TC-P_16 Ammonia emissions following crop-based and manure-based digestate applied to maize, with or without a nitrification inhibitor

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1. Objectives

To quantify ammonia emissions and crop available nitrogen supply from crop-based digestate and manure-based digestate, applied to maize, with or without the nitrification inhibitor Dicyandiamide (DCD).

2. Methodology

Replicated field scale experiments were conducted at two sites, Gleadthorpe (Nottingham) and Fakenham (Norfolk), on sandy loam textured soils. The organic materials were applied at two timings: in April 2014 *(pre-drilling)* and May 2014 (*post-drilling* i.e. topdressed onto the growing crop). The liquid organic materials were applied by bandspreading (using the ADAS purpose-designed small plot applicator) and fibre digestate was surface broadcast. Treatments at both sites, were:

1. Untreated control (i.e. zero N-inputs)
2. Cattle slurry
3. Manure-based digestate
4. Crop-based digestate: *separated liquor (Fakenham) whole digestate (Gleadthorpe)*
5. Separated fibre from crop-based digestate *(pre-drilling application only)*
6. Crop-based digestate, with nitrification inhibitor dicyandiamide (DCD). DCD was applied at a rate of 10 kg/ha, and was mixed in with the digestate before application.

At both sites, ammonia (NH_3-N) emissions were measured, using small scale wind-tunnels (Lockyer, 1984), for 7 days following the application of organic materials. Maize yields were measured on each plot, using a specialist plot harvester. Additional plots received applications of manufactured fertiliser nitrogen (N) (0-200 kg/ha N in 40 kg/ha increments). The fertiliser N replacement value of the different organic material applications were calculated by comparing the maize yields and N offtakes on the organic material treatments with those on the manufactured fertiliser N plots. Differences between treatments were tested using Analysis of Variance (ANOVA) followed by Duncan's multiple range post hoc test, in Genstat (16th Edition).

3. Results and discussion

3.1 Properties and nutrients supplied by organic materials

Representative samples of the organic materials were collected at application and analysed for dry matter, pH, total N and readily available N (i.e. ammonium-N – NH_4-N) (Table 1). Applications of cattle slurry and manure-based digestate supplied similar amounts of total N (90-112 kg N/ha) and NH_4-N (52-59 kg N/ha), however the pH of manure-based digestate was 0.4 units (at Fakenham) and 0.8 units (at Gleadthorpe) higher than cattle slurry. Crop-based digestate (either as whole digestate at Gleadthorpe, or separated liquor at Fakenham) supplied more total N (142-180 kg N/ha) and NH_4-N (77-163 kg N/ha) compared to cattle slurry or manure-based digestate. The pH of the crop-based digestate, was 0.3 units (at Fakenham) and 0.8 units (at Gleadthorpe) higher than cattle slurry. Notably, the amount of NH_4-N supplied by crop-based fibre digestate differed between sites, this was due to the contrasting fractionation methods used to separate the fibre fraction. Specifically the fibre fraction from crop-based digestate applied at Gleadthorpe which was separated using a belt and centrifugal supplied, *c.*120 kg NH_4-N /ha, (exceeding the amount supplied by either cattle slurry or manure-based digestate). While the fibre fraction from crop-based digestate applied at Fakenham, which was separated using conventional farm slurry separation equipment, supplied the lowest amount of NH_4-N, <10 kg/ha.

Table 1: Summary of organic material analyses applied at Fakenham and Gleadthorpe experimental sites.

	Units	**Cattle slurry**	**Manure-based digestate**	**Crop-based digestate**	**Fibre crop based digestate**
Application rate	t or m^3/ha	30	30	30	40
			Fakenham		
Dry matter	%	7	5	8	16
pH		7.8	8.2	8.1	8.3
Total N	kg/ha	90	98	142	230
Ammonium-N	kg/ha	52	59	77	6
			Gleadthorpe		
Dry matter	%	8	8	8	22
pH		7.5	8.3	8.3	8.5
Total N	kg/ha	91	112	180	270
Ammonium-N	kg/ha	56	59	163	121

3.2 Fakenham - Ammonia emissions and crop yield

Ammonia-N (NH_3-N) losses following the application of separated digestate fibre at <1% of N-applied were lower (P <0.05) than from all other treatments reflecting the low ammonium-N content of the digestate fibre (c.2% of total N applied) (Figure 1). Ammonia-N losses following the application of cattle slurry were lower (*c.*15% of total N-applied) than other treatments (P <0.05). The greater NH_3-N losses following the application of crop-based digestate and manure-based digestate may be a reflection of the higher pH of these materials. DCD had no effect (P >0.05) on ammonia emissions from crop-based digestate.

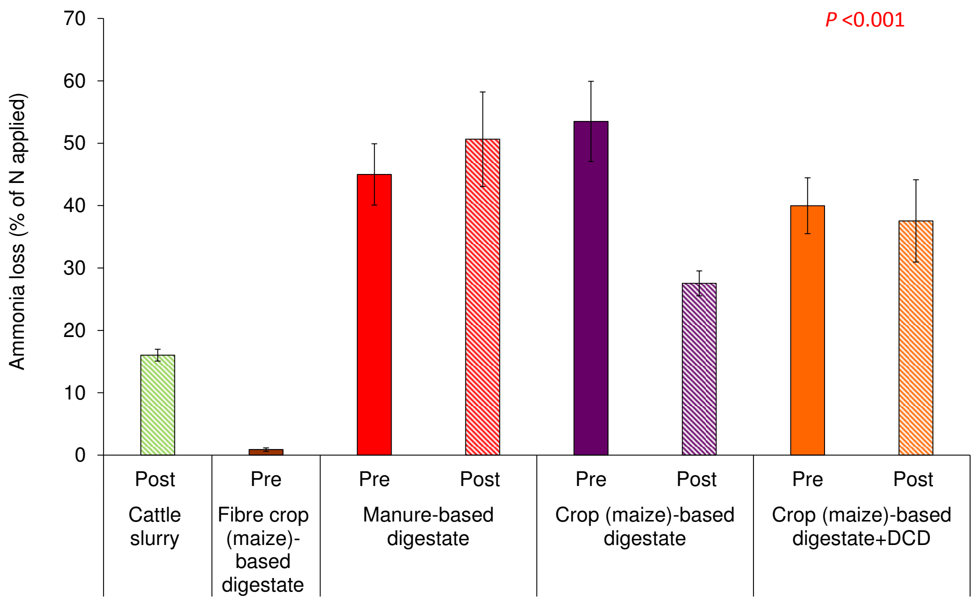

Figure 1: Ammonia volatilisation losses (% of total N-applied) following the application of organic manures at Fakenha either pre-maize drilling (*viz. pre*) or post-maize drilling (*viz. post*).

3.3 Gleadthorpe - Ammonia emissions and crop yield

Ammonia losses following the application of crop-based digestate and crop-based digestate +DCD were significantly greater (*P* <0.05) at *c.*40 and *c.*35 % of total N applied, respectively, than losses following the application of cattle slurry (22 % of total N applied) (Figure 2).There were no differences in NH_3-N volatilisation losses from manure-based digestate (27 % of total N applied) and cattle slurry (22 % of total N applied) (*P* >0.05). DCD had no effect on ammonia emissions from crop-based digestate. The higher ammonia emissions from the crop-based digestate applications may be a reflection of the higher pH of the digestate compared to the cattle slurry and manure based digestate. Application timing had no effect (*P*>0.05) on ammonia losses from any of the liquid organic material applications.

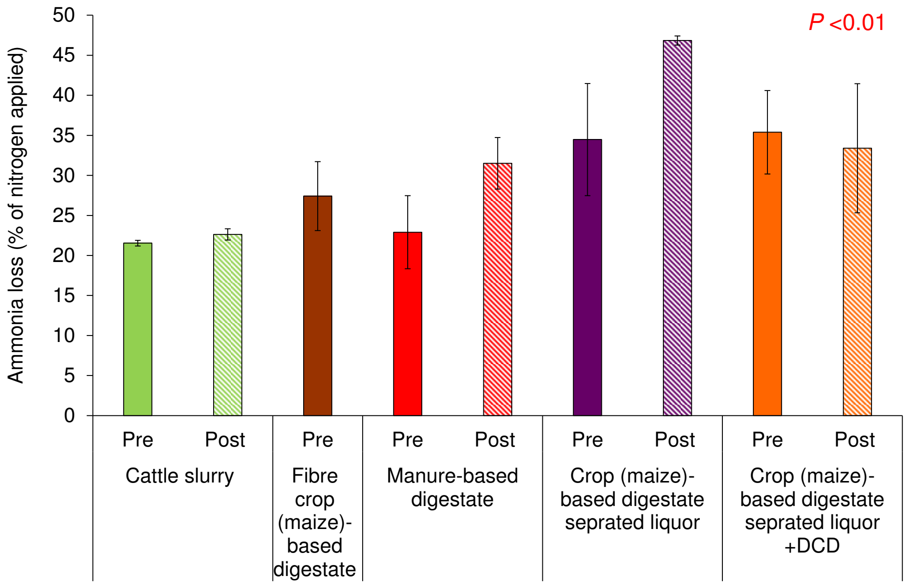

Figure 2: Ammonia volatilisation losses (% of total N-applied) following the application of organic manures at Gleadt horpe either pre-maize drilling (*viz. pre*) or post-maize drilling (*viz. post*). There was a significant difference in ammonia losses among liquid organic materials (*P* <0.01).

At Gleadthorpe, maize was harvested on 3 October 2014 and dry matter yields calculated (t/ha) (Figure 3). Overall mean dry matter yields were *c.*1 t/ha greater from organic materials applied pre-drilling compared to post-drilling.

Maize yield responded to N-fertiliser application and a linear+exponential curve was fitted (Figure 2). The economic optimum N-rate was not reached this was despite applying up to 200 kg N/ha. There was no significant difference in the nitrogen use efficiency (NUE) (%) between different manure types (*P* > 0.05). However, reflecting dry matter yields, the overall mean NUE of pre-drilling application of organic manures at 50% was greater (*P* <0.01) than post-drilling (overall mean NUE = 30%) application.

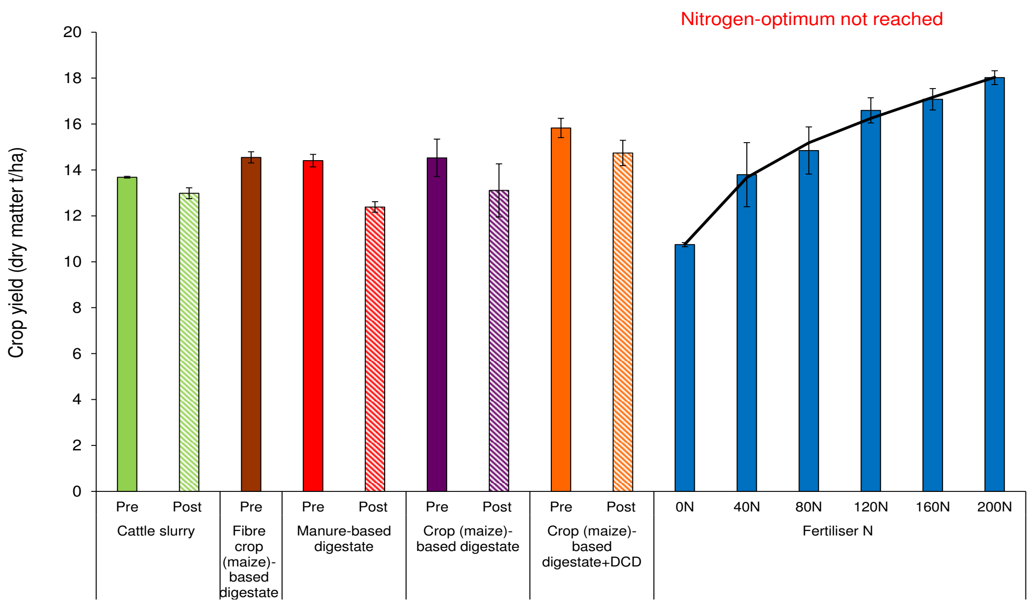

Figure 3: Maize yield at harvest (t/ha dry matter) following the application of organic manures at Gleadthorpe, either pre-maize drilling (*viz. pre*), post-maize drilling (*viz. post*) or following the application of nitrogen (N) fertiliser (ammonium nitrate), i.*e. N-response treatments*. The black line, represents the fitted yield (line ar+exponential curve), note: N-optimum not reached.

4. Conclusion and outlook

In summary the results indicate:

- At both sites, Ammonia volatilisation losses were greater following the application of crop-based digestate than following cattle slurry applications which may reflect the higher pH of the digestate.
- At Gleadthorpe, N-use efficiency of the organic materials was higher when applied pre-drilling, reflecting N-supply coinciding with crop requirement.

These results emphasise the need to develop strategies to reduce ammonia emissions from digestate applications. This information is essential to maximise the fertiliser N-value of digestate and to promote the sustainable use of organic materials in agricultural systems.

Acknowledgements

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References

[1] Lockyer, D.R. (1984). A system for the measurement in the field of losses of ammonia through volatilization. *Journal of the Science of Food and Agriculture* **35**, 837-848.

TC-P_17 Limiting nutrient losses and improving product quality during storage of cattle manure by composting and ensiling

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Objectives

Storage of cattle farmyard manure (CFM) on the field might cause nutrient leaching [1]. Therefore, Flemish farmers are not allowed to store manure (i) on the field between 15/11 and 15/01, and (ii) longer than 1 month during the rest of the year. It is evaluated whether composting and ensiling can reduce nutrient losses, and optimize the quality of CFM as fertilizer and soil improver.

Methodology

Storage experiments were conducted during April-June 2014. Four CFM treatments consisting of the same volume (24 m³) were compared: (1) untreated storage on the field, (2) composting on the field (2 times turned and covered with a semipermeable fabrics cover), (3) composting on a concrete floor (2 times turned and covered) and (4) ensiling on a concrete floor. During the experiment, mineral N enrichment in the soil for the field treatments, gaseous emissions and product quality were monitored.

Results

Composting CFM resulted in a more homogeneous product with a lower volume compared to the untreated CFM. However, the differences in product quality between untreated storage and composting on the field were small. We observed a lower amount of NH_4^+-N in the 0-30 cm soil layer under the compost pile compared to the untreated pile, possibly due to more leaching and mineralization under the untreated pile. Further research is being conducted to confirm those results.
The composted pile on the concrete floor was wetter than the one on the field, likely because of absorption of run-off water. This resulted in a better composting process and a trend toward higher gaseous emissions. However, little differences in product quality between composting on the field and on the concrete floor were observed.
Ensiling CFM on a concrete floor resulted in a lower temperature compared to composting, due to a limited organic matter decomposition. The losses of organic matter, dry matter and nitrogen were smaller during ensiling the CFM. The silage end-product had a higher NH_4^+-N and moisture content and was more heterogenic than the compost. High CH_4-emissions were noticed when opening the silage.

Conclusion

Composting CFM resulted in less mineral N leaching from the pile to the soil underneath compared to untreated storage. The composted CFM is more easy to transport and spread than the untreated and ensiled CFM. Ensiling the CFM resulted in a product in which organic matter and nutrients were better conserved during composting. Differences in product stability are currently assessed by an incubation test in which N-mineralization is measured in soil amended with the different products.

References

[1] Sommer, S.G. (2001): Effect of composting on nutrient loss and nitrogen availability of cattle deep litter, European Journal of Agronomy, 14, 123-133

Thematic Area TD – The bioresource challenge
(Oral presentations)

Cell factory, processed photograph, Institute of Bioprocess and Biosystems Engineering, TUHH

TD-K Bioresource utilization in context with regional development

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1. Objectives

Bio-resources are becoming increasingly contested as food demand as well as pressure to provide energy and materials from them are growing. A rational utilisation of bio-resources is however a pre-condition for a successful bio-economy in Europe which in turn is a crucial element for achieving sustainable development.
Rational use of bio-resources must take their environmental, societal and technological functions as well as the chances and restrictions posed by their diversity and properties into account. As a matter of fact all these aspects show considerable deviation from fossil resources that constitute current dominant resources. This implies that technical approaches as well as logistics and business models of industry will have to change profoundly if bio-resources shall play a major role in supporting European economic and social development.
Based on the analysis of services of bio-resources in a bio-based economy, their particular properties and the characteristics of current state technologies the paper develops systemic approaches to utilise bio-resources within the framework of sustainable development, with particular focus on the regional context of their utilization.

2. Methodology

The paper builds on the results of a discourse among European experts within the framework of the European Sustainable Energy Innovation Alliance (eseia). This discourse was initiated by the political discussion about the role of bio-energy within the encompassing framework of the European Strategic Energy Technology Plan (SET Plan) [1], taking the requirements of a European Bio-economy into consideration [2]. The discourse is summarized in a “Discourse Book” [3]. The argument within this discourse book starts from particular properties of bio-resources that require special focus on the spatial context of bio-resource utilization, analyzes the functions of bio-resources in ecological, economic and societal dimensions as well as the current state of technologies and finally lead to recommendations regarding the integration of bio-resource utilization in (regional) planning processes. The current paper is a short overview of the argument in the discourse book.

3. Results and discussion

3.1 Contested bio-resources

Bio-resources are an important way to convert natural income in the form of solar radiation into material goods. There seems to be no other group of resources that can fulfil such a wide variety of demands as bio-resources, providing food and feed to energy to material products. There is however no other group of resources that entail so direct and large scale impact on society and the environment for its provision and that so clearly highlight the general conundrum of sustainable development, namely living within “limited infinity”. Bio-resources require as a basic production factor area, which is limited as our planet has a limited surface. Besides that it requires other ingredients, namely fertile environmental compartments, be they soil or water bodies.
Seen from the vantage point of converting solar energy into energy embodied in materials, the generation of bio-resources by the process of photosynthesis is not a particularly efficient process. Maximum theoretical conversion rates of solar radiation into bio-resources are between 4.6 for C-3 plants (e.g. wheat) and 6 % for C-4 plants (e.g. corn). Conversion rates in real agriculture are however much lower at about 50 % of these theoretical values. This compares with conversion rates of about 60 to 75 % from solar radiation to heat for thermal solar collectors. It is this low efficiency of converting solar radiation into useful energy from bio-resources (regardless if used for nutrition or technical purposes) that makes “living in limited infinity” particularly obvious: bio-resource generation rate is limited by the limitation of the production factors, most notably arable land and forest area. This rate may however be sustained over (practically) infinite time if human society learns to manage these production factors cleverly.

Land as a basic production factor for most bio-resources is strictly limited on our planet. This means that any competition for bio-resources will inherently end up as a competition for land. On top of that different actors vying for most bio-resources have different preferences regarding the form of resource the use: the food sector needs crops of various kinds, pulp & paper as well as construction industry need wood from forests whereas the chemical sector has the broadest portfolio of resources. This leads to direct competition between sectors, such as the competition for crops between the food sector and bio-fuel industry, but also to a competition between different forms of land use. Forests providing resources for energy provision or pulp &paper industry may compete with fields that provide crops for food or bio-fuel.
One of the major differences to other (in particular fossil) resources is the wide variety of bio-resources. This does not only concern the wide variety of plants and animals that man has domesticated for his purposes. It also applies to by-products and wastes from industrial and societal processes using primary bio-resources.
The intense competition for bio-resources (and hence for fertile land) applies in particular to the relative small number of primary agricultural crops and wood from forestry. There is however another kind of bio-resources that is outside the current pattern of competition: secondary bio-resources that are by-products or wastes from agricultural, industrial or societal processes. Their use does in general not add to the direct competition for land although it may influence the fertility of land. In general their use is either in the form of a cascade (prolonging the value chain of a primary resource within society) or parallel to a valuable crop (using parts of plants that are usually not entering the markets such as straw and corn cobs) or additional (using land that is underutilised or otherwise not cultivated). As a rule of the thumb overall global flow of secondary bio-resources is in the order of magnitude of the current flow of agricultural products. These resources therefore offer a significant chance to alleviate the future resource burden.

3.2 Logistical features of bio-resources

The most obvious dissimilarities between bio-resources and fossil resources are their different logistical parameters. If bio-resources have to play a more prominent role in energy provision and industry this means a radically changed economic and logistical structure of resource provision. Bio-resources are characterised by high humidity and/or low transport density and generally lower energy content. In many practical cases therefore transport volume will become limiting for the collection logistic of these resources. The differences between fossil resources and bio-resources are poignant: there is a factor of 24 between the energy density of straw and light heating oil.
The logistical challenge becomes even more visible if different means of transportations are factored in. According to their energy efficiency (and strongly influenced by their particular ratio of empty weight to load capacity) transport systems require different energy to transport a load a certain distance. If we set the limit of the energy used to transport a resource to its utilisation site arbitrarily to 1 % of the contained energy and take energy density and the available transport volume into account, we obtain the following results [4]:

- In case of manure, straw and corn silage 1 % of the energy contained in a tractor trailer load will power a tractor (as the most common short distance means of transportation on farms) 5.7, 12 or 18 km respectively;
- 1 % of the energy contained in a wood chips and split logs load will power a truck for 40 and 100 km respectively;
- For wood pellets and corn a train will go for 475 and 525 km respectively using 1 % of the transported energy content;
- An ocean going ship loaded with crude oil however will travel 7.800 km with 1 % of the energy contained in its cargo.

These numbers highlight the spatial context of resource use: whereas it is fully rational to establish a global fossil economy as transport from source to utilisation plays almost no role, the use of bio-resources must become regional and possibly even local. This is particularly true for secondary bio-resources as they are characterised by especially disadvantageous logistic parameters.

3.3 Functions of bio-resources

Any attempt to balance the utilisation of bio-resources sustainably must be based on an analysis of the services that bio-resources have to fulfil for global sustainable development. This requires defining the social, economic and environmental services that bio-resources may provide and to analyse which of them can or cannot be performed by other resources or what the restrictions on other resources are to provide them in a sustainable way. Table 1 provides an overview on the services provided by bio-resources to society, economy and the environment. Priority should be given to utilisation of bio-resources to provide services that may not be provided by other resources.

Table 1: Services of bio-resources

Type of service	Service	Possible other resources
Social	Nutrition	None
	Jobs and development for rural regions	None
	Social stability for rural regions	None
Economic	Stability for energy distribution grids	Smart grids, hydro power, pumped hydro power, hydrogen, compressed air energy storage, (fossil resources)
	Transport fuel	Electricity (using battery storage), hydrogen, synthetic fuels, (fossil resources)
	High temperature industrial heat	Hydrogen, (fossil fuels)
	Feedstock for synthetic materials and plastics	(fossil resources)
	Feedstock for conventional bio-based products	None
Environmental	Reduction of greenhouse gas emissions	Wind and hydro power, solar thermal systems, photovoltaic, oceanic power, geothermal energy
	Preserving soil fertility	None
	Preserving water and nutrient cycles	None
	Preserving bio-diversity	None

3.4 Guidelines for using bio-resources in the regional context

The logistical properties of bio-resources as well as the services they have to provide define their strong dependence on spatial context. The following notions can be seen as heuristics guiding a regional discourse about utilising bio-resources based on the properties, technological aspects and service provision priorities presented earlier in this article.

3.4.1 “Refinerize” conventional sectors

Bio-resources are the only possible basis for sectors that conventionally utilise wood and crops such as pulp & paper, timber or oils and fats industry but also the food sector. These sectors provide products that cannot easily be replaced by other goods or services and render decent profits along their value chain. Moreover such industries have established well organised logistical systems and provide jobs and income as well as skills and qualification for employees. Putting priority to serving these sectors is sensible and in the case of the food sector even obligatory.
The sectors themselves, however, have to evolve into flexible bio-refinery systems based on their main resource but accommodating other (secondary) bio-resources provided by their spatial and economic context. This strategy serves two objectives:

- Employing existing logistic systems as well as skills and technological infrastructure to realising the rule of fully utilising sustainably available bio-resources and
- Offering a broader portfolio of goods and services from available bio-resources thus adding to the necessary flexibility of markets, accommodating possible shifts in preferences of consumers in a sustainable bio-based economy.

3.4.2 Give priority to material goods

Material goods provide higher added value on the resource than energy services as they offer opportunities for longer and more complex value chains. As a rule of the thumb value added to resources increases with longer value chains as well as with more complex products. Their provision therefore will bolster the economic viability of a bio-economy. The range of such products however is wide, from simple solid biomass fuel such as wood pellets to gaseous and liquid biofuels to bio-plastics to complex pharmaceutical products.

3.4.3 Retain as much material as possible close to productive land

Preserving the productivity of the primary resource land is to a large extent depending on closing material cycles and returning nutrients to the land. This requires that any part of bio-resources that is not converted into material goods (be they nutritional or for other purposes) shall be recycled to fields, grass land and forests. As most of the waste flows generated by the utilisation of bio-resources have particularly unfavourable logistic parameters their transport distances must be kept short. This demands that technologies conditioning bio-resources for further use or transport as well as technologies conditioning bio-waste shall preferably be realized de-centrally, close to the origin of bio-resources.

3.4.4 Use intersections of distribution grids as a means to fully utilise bio-resources

Technologies like thermal biomass utilisation and anaerobic digestion have a narrow product portfolio: Thermal biomass utilisation may provide heat and power (in case of CHP) and biogas can be up-graded to bio-methane or used in a CHP to generate heat and power. All these products may be distributed via distribution grids. The optimum location for these technologies therefore is where these grids intersect [5]. Particularly advantageous in this respect are tri-valent technologies. These are technologies that produce bio-methane either from anaerobic digestion in combination with gas cleaning or from biomass gasification with catalytic conversion to synthetic natural gas, besides electricity and heat. These technologies can serve all energy distribution grids, from heat to electricity to gas. These technologies should be operated according to the electricity grid requirements, switching to gas cleaning and injection into the gas grid whenever no electricity is necessary to stabilise the grid and serve heat customers via heat storage systems fed during the CHP mode.

The necessary proximity of heat users to thermal bio-energy providers gives raise to another type of bio-refinery: de-central bio-refineries converting secondary or under-utilised bio-resources to intermediate “platform” materials, improving quality and transport properties and conditioning these bio-resources for processing in central large scale bio-refineries.

4. Conclusion and outlook

Bio-resources are inherently contextual resources, their rational utilization requires taking their local and regional environmental, social and economic context into account. They are also highly contested resources as the food, energy and chemical sector compete for them. This competition will push the energy as well as the industrial sector more and more to utilize lower quality bio-resources that are either by-products or wastes from other uses. As these resources have even less advantageous logistical properties than crops and high quality forest products, this will increase the spatial connotation of bio-resource utilization systems.

Increased reliance on agricultural and forest by-products and harvest residues open however more scientific challenges. Their use is only sustainable when their withdrawal does not impair the fertility of land. If the withdrawal is too high, soil quality may suffer, putting further use of the basic resource land into jeopardy. The definition of sustainable withdrawal rates as well as the right way to re-integrate residues from technical uses of bio-resources to maintain soil quality will become a major challenge for rational and sustainable bio-resource utilization in future.

Acknowledgement

The authors want to thank the members of the Working Group Bio-Resources of eseia for their valuable input to the discourse book on which the current paper is based.

References

[1] EC (2007): A European Strategic Energy Technology Plan (SET-PLAN)- Towards a Low Carbon Future, COM(2007) 723 final , EC, Brussels,
[2] EC (2012): Innovating for Sustainable Growth: A Bio-economy for Europe COM(2012) 60 final, Brussels
[3] eseia: Innovation challenges towards the rational use of bio-resources in Europe – a discourse book, eseia, Graz, 2014, http://www.eseia.eu/files/attachments/10457/453058_eseia_Discourse_Book_May_2014.pdf, last accessed May 2015
[4] Narodoslawsky M. (2014,): Utilising bio-resources – a rational strategy for a sustainable bio-economy, ita-manuscript, Vienna, http://epub.oeaw.ac.at/ita/ita-manuscript/ita_14_02.pdf
[5] Stoeglehner, G., Niemetz, N., Kettl, K.-H. (2011): Spatial dimensions of sustainable energy systems: new visions for integrated spatial and energy planning. Energy, Sustainability and Society, 1, 1-9

TD-O_01 Co-digestion of manure and organic residues in the Netherlands

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1. Objectives

The aim of this study is to assess the environmental, economical, health, societal, and legislative aspects of co-digestion of livestock manure with organic residues in the Netherlands [1]. The results of this study are used by the ministries of Economic Affairs and of Infrastructure and the Environment in an evaluation of the policy on co-digestion of manure in 2015.

2. Methodology

The assessment was based on a literature review, analysis of statistical and administrative data, evaluation of permits of co-digesters and discussion sessions with stakeholders. The study focused on:

- the added value of co-digestion in terms of sustainable energy production, the use of organic wastes, the reduction of greenhouse gas emissions and manure surpluses;
- the risks of co-digestion to the health and safety of humans, animals and the environment;
- measures that could be taken to reduce these risks; and
- regulations for co-digestion and the ability to enforce these regulations, and measures to improve their enforceability.

3. Results and discussion

3.1 Co-digestion

Digestion is a method for sustainable energy production. Biomass can be digested to produce biogas containing methane (CH_4). The biogas can be refined to ‘green gas’ (having the quality of refined natural gas) or can be used in its unrefined state to generate electric power and heat. In the context of this article, co-digestion is defined as ‘any form of digestion of animal manure mixed with plant and/or animal-derived co-digestion wastes, with a minimum manure share of 50% (w/w)’. The digestate produced by the co-digestion process can be used as an organic fertilizer. Energy rich organic wastes that are used for this process are called co-digestion materials. The composition and fertilizer value of digestate is dependent of the composition of the manure and the added co-digestion materials.

Many factors and actors influence co-digestion (Figure 1). Bio-physical, technological, and management factors affect the co-digestion process and the production and quality of biogas and digestate. The markets for energy rich organic wastes and energy, legislation and societal aspects determine the economic viability and the acceptance by society of co-digestion.

3.2 Economy and policy goals

Digestion of manure and co-digestion materials contributes to three policy goals: (i) sustainable energy production, (ii) reduction of greenhouse gas emissions to the atmosphere and (iii) utilisation of the energy contained in organic wastes. Co-digestion also provides employment and generates income for the companies involved in the co-digestion chain.

Interest in the digestion of manure and co-digestion materials grew after governmental subsidy schemes for renewable energy were introduced from 2003 in the Netherlands. Various subsidy schemes were used which stimulate amongst others energy production by co-digestion. Over the 2010–2013 period, average annual subidies amounted to 55 to 60 million euros. The government also stimulated co-digestion by subsidising the construction of plants, providing information and funding research. After 2010, the growth of the number of co-digestion plants stagnated due to rising prices of co-digestion materials, falling electricity prices and changes to subsidy schemes for renewable energy. There are currently about 100 co-digesters in the Netherlands and this number has been stable during the last years (Figure 2).

Figure 3 shows the increase in use of manure and organic residues for co-digestion. About 3 percent of the total amount of cattle and pig slurry manure is digested annually. The main co-digestion materials used are silage maize, silage grass, grain residues, glycerin from biodiesel production, spent grain for brewery and bioethanol production, and wastes from the food and feed industry. In the absence of co-digestion, co-digestion materials can find uses as raw material for livestock feed, compost (followed by use as organic fertiliser), or would have been incinerated.

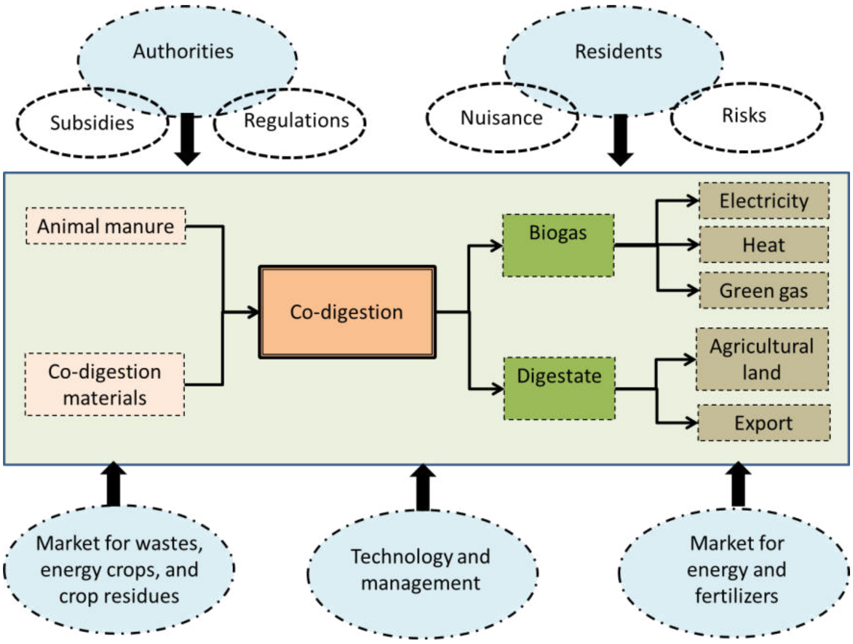


Figure 1: Schematic presentation of co-digestion of manure and co-digestion materials to biogas and digestate and the factors and actors that affect co-digestion.

3.3 Environment and spatial planning

The digestion of manure and co-digestion materials has both favourable and unfavourable effects on the environment. The favourable effects derive from the three policy goals defined for co-digestion, that is, sustainable energy production, reduction of the greenhouse gas emissions to the atmosphere and utilisation of wastes (see previous section). The unfavourable effects derive from the potential inclusion of risk bearing components for humans, animals, crops and the environment in the co-digestion materials and possible leakages of biogas from stationary and operating co-digestion plants and handling, transport and storage of manure and co-digestion materials.

About half of the 155 types of energy rich organic wastes that have been authorised for co-digestion have passed an environmental test on contaminants and pathogens [2]. This means that the concentrations of environmentally harmful contaminants in these wastes are below the standards specified in the Fertilisers Act. Those intending to use the other half have to be able to show upon inspection that they meet the environmental requirements, that is, that the concentrations of heavy metals and organic micropollutants in the co-digestion materials meet the standards specified in the Manure and Fertilisers Act.

When authorised co-digestion materials are used and the rules and regulations are complied with, the risks for human, animal, crop and the environment due to the use of the digestate as a organic fertiliser are neglectible. By contrast, the risks may be large if non-authorised, contaminated wastes are mixed with the co-digestion materials. Waste from animal origines (animal byproducts) have to be authorized for use as co-digestion material in accordance of European legislation on animal byproducts. The Netherlands Food and Consumer Product Safety Authority (NVWA) is the authority who controls uses of animal by products and other co-digestion material within the frame work of the Fertilizer act. Nearly 30% of the co-digestion plants inspected by the NVWA in 2013 and 2014 were found to use co-digestion materials which did not meet legal provisians in force. Several batches of organic wastes found at the location of digestion plants were declared wastes, because of excessive concentrations of heavy metals (especially nickel and chromium). The

National Police Services Agency concluded from their inspections that co-digestion is attractive for illegal waste dumping activities. In response to this outcome, the responsible State Secretary of the Ministery of Economic Affairs indicated in 2014 that companies appointed co-digestion materials have to be certified. This is the first step towards ensuring that no contaminated organic wastes are used. The next step can be certification of appointed co-digestion materials.

There have been several media reports of complaints by citizines living near co-digestion plants about odour nuisance and discomfort by traffic due to transport of co-digestion materials, manure and digestate. The capacity of digestion plants has grown steadily since the start of subsidised renewable energy production by digestion, which has led to questions about suitable locations and operational management, and the safety of such plants. Inconvenience to people living near the plants and environmental impacts could be reduced by further professionalisation of the co-fermentation sector. External and internal communications must be improved; the biogas sector association is already working on this.

Adding co-digestion materials to manure means that extra phosphate (about 2.7 million tons P_2O_5 in 2013) is supplied in animal manure. Digestate of co-digestion ranks under animal manure. As a result, the total manure production in terms of phosphate rose from 167 to 170 million kg P_2O_5 in 2013. The volume of digestate exported in 2013 has been estimated at about 1 million kg phosphate. On a national scale, the effects of co-digestion on ammonia emissions, nitrate leaching and soil fertility are limited compared to those of untreated animal manures.

Figure 2: Location of co-digestion plants in the Netherlands.

3.4 Health and safety

The effects of co-digestion on the health and safety of humans and animals include (i) occupational health risks to employees of co-digestion plants, (ii) the risks to the external safety of citizins living near these plants and (iii) the inconvenience to citizins living near the plants due to odour nuisance. The permits for co-digestion plants rules the planning, the construction and operation of the plant. These rules foresee in an acceptable risk.

The risks of co-digestion to the health and safety of humans and animals mostly relate to the transport, handling, storage and composition of the biogas, manure, digestate and co-digestion materials. Biogas is flammable and contains hydrogen sulphide (H_2S), ammonia (NH_3), carbon dioxide (CO_2) and CH_4, high concentrations of which are poisoning or asphyxiating to humans and animals. These gases may also be formed in stored manure and digestate.

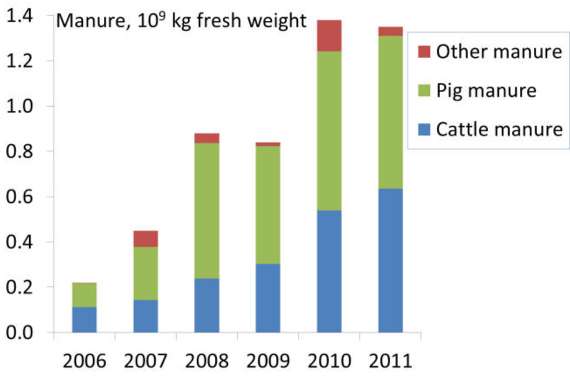
The risks of co-digestion are greater for the employees actually working at co-digestion plants than for citizins living near them. The safety of co-digestion plants can be improved by increasing the risk awareness of employees and by training on safe working conditions.

When authorised co-digestion materials are used, the risks of pathogenic micro-organisms being spread as a result of co-digestion processes or from the digestate are comparable or less to those of non-digested manure which are considered to be manageble low risks.

3.5 Implementation, enforcement and compliance

The implementation of the policy on co-digestion involves agencies of the Ministries of Economic Affairs and of Infrastructure and the Environment, as well as provincial authorities, municipalities and regional environmental services. If incidents occur at digestion plants, agencies of the Ministries of Security and Justice and of Health, Welfare and Sport also become involved. The tasks are clearly defined, but competence of enforcing agencies do overlap to some extent. More collaboration and exchange of information could result in synergy.

The implementation of and enforcement on permits and legislative regulations for co-digestion are difficult in practice. Not only because of frequent changes in subsidy schemes and the market for co-digestion materials which affects the operation of the plan. Also the large number of parties involved in the co-digestion chain hinders enforcement. Next, numerous laws and regulations apply on planning, construction and operational use of co-digestion plants. Lastly the fragmented organisation of the inspection and limited powers of inspectors hinders enforcement.



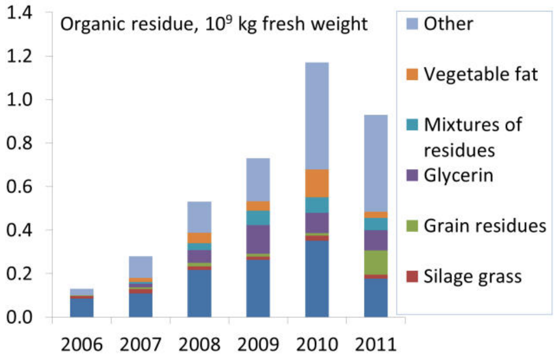


Figure 3: Input to co-digestion installations as manure (left figure) and organic residues (right figure) in The Netherlands. (Source: Statistics Netherlands, The Hague).

4. Conclusion and outlook

Co-digestion of manures with co-digestion materials contributes to renewable energy generation, reduction in greenhouse gas emissions and recycling of organic wastes, but the contributions to the totals are relatively small in the Netherlands. Co-digestion is without subsidies not economic feasible currently. When authorised co-digestion materials are used and the rules and regulations are complied with, the risks of environmental pollution due to the use of the digestate as a fertiliser are limited. Without sufficient inspection and enforcement, there is a risk of non-authorized and possibly hazardous wastes being mixed in during co-digestion of manure. There is a need for more efficient enforcement and controls, better communication and education to decrease risks related to co-digestion for humans, animals, crops and environment and to increase the acceptance by the society.

Acknowledgements

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References

[1] Scientific Committee of the Manure Act CDM (2014) Benefits and risks of co-digestion: synthesis. WOt-report 32, Wageningen University and Research Centre, Wageningen. In Dutch.
[2] Scientific Committee of the Manure Act CDM (2013) Protocol assessment of wastes Fertiliser Act. Version 3.1. WOt-report 335, Wageningen University and Research Centre, Wageningen. In Dutch.

TD-O_02 Perspective and conversion routes for bioproduction of bulk chemicals and fuels from organic wastes and green electricity

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1. Objectives

To deal with the global challenges of limited fossil resources, climate change and environmental pollution bio-economy has been identified globally as a strategic development goal. In this regard, many industrial countries and regions have set very ambiguous targets: e.g. within the EU 25 to 30% of all chemicals and other industrial products should be bio-based by 2030. To this end, new concepts of bioproduction are desperately needed. This work presents a new perspective for a sustainable bio-economy [1].

2. Methodology

The major limitations of present bioproduction systems are analyzed, especially regarding substrate availability, product yield and processing costs. Against these background and facts a new strategy toward a sustainable and feasible bio-economy is proposed which takes advantages of the latest developments in biology, renewable energy technologies and electro-biotechnology. The components of the strategy are discussed in more details, especially regarding the resources and technologies.
Results (Note: The major text and materials are taken from Zeng and Kaltschmitt [1] with slight modification).

The following facts are of particular importance in developing a sustainable bioproduction system for bio-economy:

- CO_2 as the final product of each use cascade of organic material could basically be recycled into products following the example of nature and could thus be used as a carbon source,
- electrical energy from renewable sources (like wind or solar) is available already at low costs; due to the fluctuating characteristic of the wind speed and solar radiation this green electricity is available to a certain extend as a surplus when high shares of such energy sources are used within the electricity supply system and no large scale storage systems are technically available at low costs, and
- organic waste material from agri-, horti- and aquaculture, from households and from industry can often be used most efficiently by anaerobic fermentation providing bio-methane which could be used easily and highly efficient within the existing energy system. Adding the fact that methane from fossil sources (i.e. natural gas) is a widely used feedstock, a so-called “Methane Bio-engineering” is emerging [2].

We therefore propose the concept E&G^{2}C with the central idea to first convert organic wastes into a widely usable product – biogas (CO_2 +CH_4) – which is then used as a clean and uniform substrate for the synthesis of bulk-chemicals and/or fuels, especially by using green electricity from wind and solar (Fig.1).
The different components of this concept are briefly outlined below.

- Biogas production from organic (waste) material has made significant progress in recent years. In Germany, for example, nearly 8000 biogas plants are under operation, producing more than 8 x10^9 m³ of biomethane as well as roughly the same amount of "green" carbon dioxide per year [3]. This biochemical process for the efficient conversion of organic waste fractions into biogas is probably the most sustainable one to transform biomass into a widely useable intermediate product due to several reasons.
    - First, it can use basically all kinds of organic (waste) materials (with the exception of lignin) and convert them into CO_2 and CH_4. Because only a part of carbon, oxygen and hydrogen is removed from the feedstock by this biochemical process, the macro-

and micronutrients can easily be recycled back to nature; thus a significant contribution in closing nutrient and humus cycles can be achieved.
- Second, mixed microorganisms are used for bioconversion under non-sterile conditions, leading to much reduced investment and operation costs compared to monoculture and mono-septic conditions usually needed for bioproduction. This technology allows for on-site and decentralized production in rural areas.
- Third, the gaseous product can easily be transported via the existing natural gas grid well developed at least in most parts of Europe. This allows it to link a decentralized biogas production – necessary due the low area specific energy density of e.g. biomass waste streams as well as the desired recirculation of the digested substrate within the natural material cycles. Thus, the existing and already amortized infrastructure can be used and a smooth transfer from the fossil economy to bio-economy is possible.

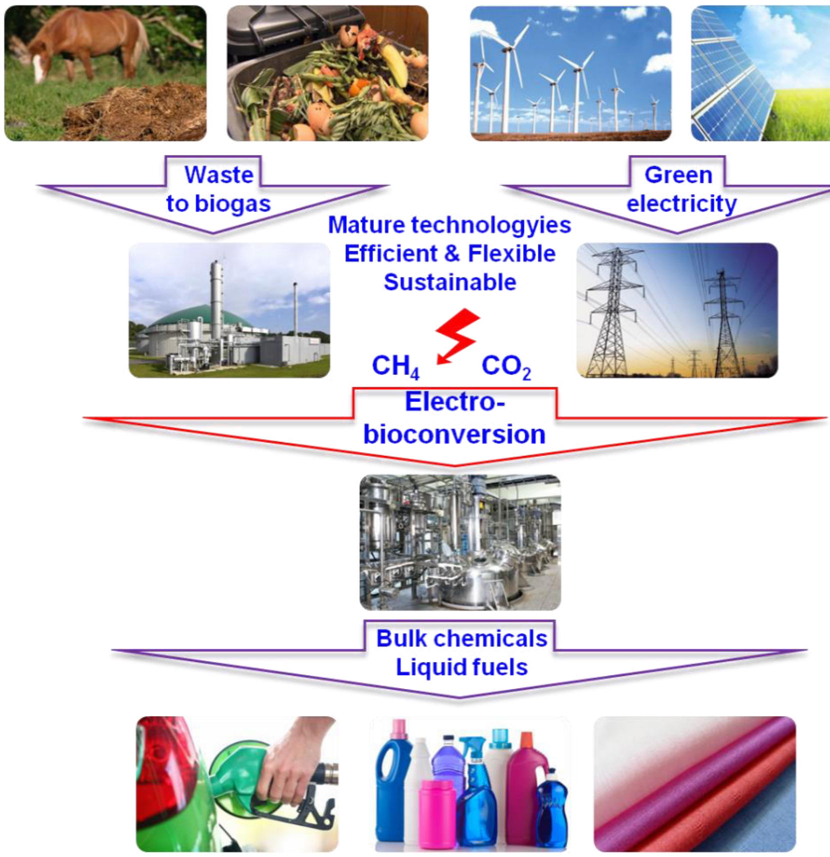

Figure 1: The concept of electricity and biowastes via biogas to bulk-chemicals (**E&G^{2}C**) [1]: organic wastes are converted into biogas which is used together with "green" electricity as a basis for the biosynthesis of bulk-chemicals and fuels; the bio-electrochemical synthesis is the key technology to be developed.

- Electricity generation, especially from wind power and solar radiation, has gained more and more importance in recent years. For example, in Germany close to 30 and 50 TWh (Tera Watt hour) have been produced from photovoltaic systems and wind mills in 2013, respectively [3]. The costs for them have been dramatically reduced during the last decade; in German wind electricity can be provided onshore at reasonable sites with ca. 0.06 €/kWh and PV at roughly 0.10 €/kWh. Due to this price drop and the efforts to reduce GHG emissions it is most likely that the electricity generation from these renewable sources will be expanded considerably in the years to come – and not only in Germany with its "Energiewende". Due to the strong fluctuations of wind and solar radiation as well as the varying demand within the electricity grid it becomes more likely that electricity exceeding the simultaneous electricity demand, especially with most likely increasing shares of electricity from wind and solar energy within the grid. This is especially true because cheap storage facilities on a large scale are hardly available (exception: pumped storage power station). Thus the probability for the occurrence of a so called "surplus energy" increases. It is expected that cheap "green" electricity will be available in the years to come with increasing shares [4].

- Biogas consisting of CH_4 and CO_2 is an ideal starting material for biosynthesis. In fact, synthesis of chemicals using either CO_2 or CH_4 alone has shown to face many obstacles:
  - CO_2 is too oxidized and contains basically no energy for chemical or bio-chemical synthesis. Here hydrogen atoms have to be incorporated into the molecule both for energy and as an elementary component of the products (e.g. through hydrogenation).
  - CH_4 is a strongly reduced compound. The incorporation of oxygen atoms (in form of O_2, H_2O or CO_2) is necessary since most of the bulk-chemicals and liquid fuels of interest are so-called oxygenates.

Thus the use of a mixture of CO_2 and CH_4 as found in biogas as starting material for (bio)synthesis can principally overcome the shortcomings mentioned above and represents probably the most economic and ecological way to use biogas.

- The use of electricity for biosynthesis has received increasing attention recently [5] because it can help to overcome some of the major limitations of present bio-production processes [6]. Product yield from substrate is a key parameter for an efficient bioproduction process. However, about 40 to 50 % of the substrate (e.g. carbohydrates) are used for energy metabolism and biomass formation in typical fermentation processes, limiting the product yield mostly to less than 50 to 60 % (by weight). Here a major factor limiting the product yield is often the provision and regeneration of reducing power (in form of nicotine adenine dinucleotides, also called coenzyme or simply cofactor NADH). In this respect the use of electricity for energy and cofactor supply can greatly improve the product yield. It can also strongly reduce the production of byproducts found in typical fermentation processes. The latter point can help to reduce the costs of energy intensive downstream processing for product recovery and purification.

The E&G^{2}C concept is open for the feed-in of additional CO_2-streams e.g. from the combustion and/or gasification of solid biomass. Additionally also electricity from other renewable sources of energy can be used easily. Besides this, if land is available, also energy crops (like maize silage) can be used as a feedstock for biogas provision. Last but not least, the bulk-chemicals produced can be either directly used as high energy density fuels (e.g. n-butanol and iso-butanol) or further processed with biotechnological methods to more complex molecules to provide more valuable products. Beside this, these bulk- chemicals can also easily be converted to high-value biofuels (e.g. jet fuels) ready to be used within the globally still fast growing transportation sector.

Within a transition period till such a concept has been implemented on a large scale it is also well compatible with the existing energy system on fossil fuel energy as well as the (bio-) chemical industry. From a purely chemical point of view the composition of natural gas and flue gas from coal fired power plants are quite similar to biogas. The existing infrastructure developed in recent years for the production, provision and conversion of fossil fuels could be a valuable add-on; i.e. it could be used also with the concept E&G^{2}C.

In the work of Zeng and Kaltschmitt [1] possible technological routes for the realization of the concept are assessed, especially regarding electrobioconversion of biogas to bulk chemicals [Fig.2]. They will be illustrated in this presentation in more details.
Bioelectric processes at anode have been intensively studied in the past for the generation of electricity using microbial fuel cells (MFC). In the contrary, bioprocesses at the anode have been rarely explored for the purpose of biosynthesis. Nevertheless, the knowledge from studies of MFC is helpful for developing microbial electrosynthesis processes; e.g. materials and design of electrodes and reactor systems developed for MFC could be directly adapted for bio-electrochemical systems. In this integrated concept of electric bioconversion of biogas the anode can be used for the oxidation of water, which results in molecular O_2; the latter can then be used to oxidize CH_4.

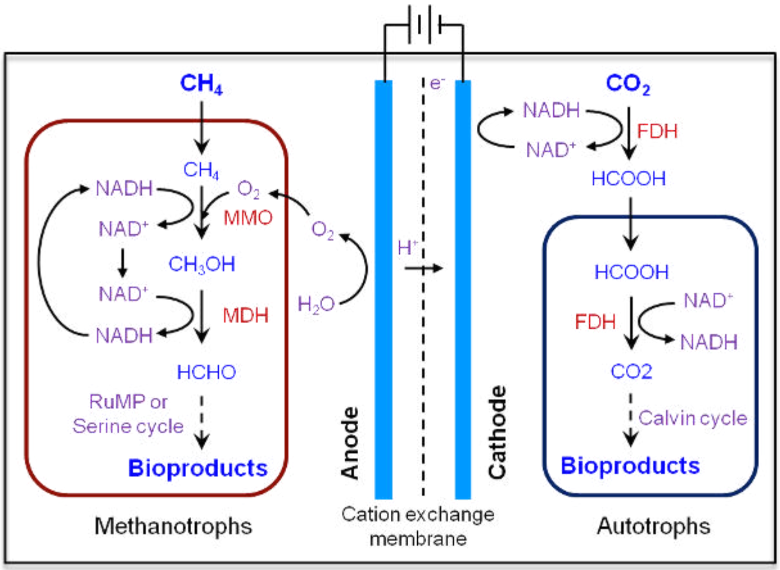

Figure 2: Principle of a bio-electrochemical system and a possible concept for an integrated electromicrobial conversion of biogas into chemicals [1].

Bioelectric processes at cathode are of particular interest for electro-biosynthesis because they can supply electrons which are required in many biosynthesis processes. This is especially true for chemicals or fuels with high energy content. For example, several investigations have demonstrated the successful reduction of CO_2 at cathode. Li et al. [7] showed a direct formation of formate from CO_2 on the cathode which is then converted to isobutanol or 3-methly-1-butanol by an engineered *Rastonia eutropha*.

3. Conclusion

The concept E&G^{2}C has the potential to overcome major limitations of known bioproduction systems. Biogas as a substrate of biosynthesis has many unique advantages including sustainability, efficiency and flexibility. And the use of electricity for biosynthesis with biogas represents an ideal system for efficient bio-electrochemical conversion. The realization of the concept can significantly contribute to the development of more sustainable integrated solutions for handling agro-industrial residues and for providing opportunities for rural-industry symbiosis.

References

[1] Zeng AP., Kaltschmitt M. (2015) Green electricity and bio-wastes via biogas to bulk-chemicals and fuels. Eng. Life Sci. DOI: 10.1002/elsc.201400262. (in print).
[2] Haynes C., Gonzalez R. (2014) Rethinking biological activation of methane and conversion to liquid fuels. Nat Chem Biol. 10(5), 331-339.
[3] Lenz V., Naumann K., Kaltschmitt M., Janczik S. (2014) Erneuerbare Energien – Erkenntnisstand 2013 in Deutschland; BWK 66(4), 61 – 80
[4] Kaltschmitt M., Streicher W., Wiese A. (2013) Erneuerbare Energien – Systemtechnik, Wirtschaftlichkeit, Umweltaspekte; Springer, Berlin, Heidelberg, 5. Auflage.
[5] Rosenbaum M., Franks A. (2014) Microbial catalysis in bioelectrochemical technologies: status quo, challenges and perspectives. Appl Microbiol Biotechnol 98, 509–518.
[6] Zeng AP. (2013) Fundamentals and Application of New Bioproduction Systems. Advances in Biochemical Engineering/Biotechnology, Vol. 137. 238 p. Springer-Verlag, Berlin.
[7] Li H.,Opgenorth P., Wernick DG, Rogers S et al. (2012) Integrated electromicrobial conversion of CO2 to higher alcohols. *Science*, *335*, 1596.

TD-O_03 Bioconversion of renewable feedstocks and agri-food residues into lactic acid

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1. Objectives

Renewable resources can be utilized directly, e.g. as energy carriers, as packaging materials, as fibres, for the production of colouring agents or as lubricants. However, they can also be converted biotechnologically by enzymes and microorganisms, giving us access to a multitude of biocompatible products and possible uses. The carbon sources from agricultural feedstocks and residues can be utilized by a lot of microorganisms, e.g. lactic acid bacteria (LAB) to produce lactic acid (LA). Lactic acid, its salts and esters have a wide range of potential uses and are extensively used in diverse fields.

Besides the quantity and availability of raw materials together with their properties and quality the feedstock costs are very important for the production of bulk chemicals like lactic acid. The goal is to develop a fermentation process based on the substitution of expensive substrates (e.g. sugar and starch based primary crops, which are also critical in view of the competition debate with food and feed) as well as nutrient supplements (e.g. yeast extract, peptones) by cheaper materials from renewable resources due to their main proportion of the whole process costs.

The global lactic acid demand was estimated to be 714.2 kilo tons in 2013, which is expected to reach 1,960.1 kilo tons by 2020, growing at a compound annual growth rate of 15.5 % from 2014 to 2020 [1]. Growth in demand for LA and its salts and esters in industrial applications will be driven mainly by LA-based biodegradable polymers like polylactic acid (PLA) and, to a lesser degree, lactate solvents [2, 3].

2. Methodology

Renewable feedstocks (crops, lignocellulosics, green biomass, residues etc.) are already being used as raw materials for the production of bio-based products. However, these feedstocks cannot be used normally for fermentation directly because the fermentable sugars are bound in the structure especially as cellulose and several types of hemicelluloses. They have to undergo a pretreatment to release these sugar components.

The value of (agri/food) residues as carbon and/or nutrient source depends on their specific contents of cellulose, hemicellulose, lignin, starch, protein and minerals. The contents can vary between different crops and a screening prior to fermentations is recommended. Table 1 gives an overview of different residues and their compositions.

Table 1: Composition of representative lignocellulosic feedstocks, adapted from [4]

Feedstock	Cellulose	Hemicellulose	Lignin
	Carbohydrate composition (% DM)		
Barley straw	36-43	24-33	6,3-9,8
Banana waste	13	15	14
Corn stover	35.1-39.5	20.7-24.6	11.0-19.1
Coffee pulp	33.7-36.9	44.2-47.5	15.6-19.1
Rice husk	28.7-35.6	11.7-29.3	15.4-20.0
Wheat bran	10.5-14.8	35.5-39.2	8.3-12.5
Sugarcane bagasse	25-45	28-32	15-25
Grasses	25-40	25-50	10-30

DM-dry matter

Food waste consists of 30-60% carbohydrates, 5-10% proteins and 10-40% (w/w) lipids, and to a lesser extent of polyphosphates [5]. Carbohydrates, proteins and polyphosphates can be converted by biological and/or chemical methods into sugars, amino acids and phosphate, respectively. The recovered nutrients can then be used as feed for microorganisms in biotechnological processes. It should be emphasized here, that the metabolic versatility of microbes enables the production of a wide range of products from bio-plastic to bio-fuel (Figure 1).

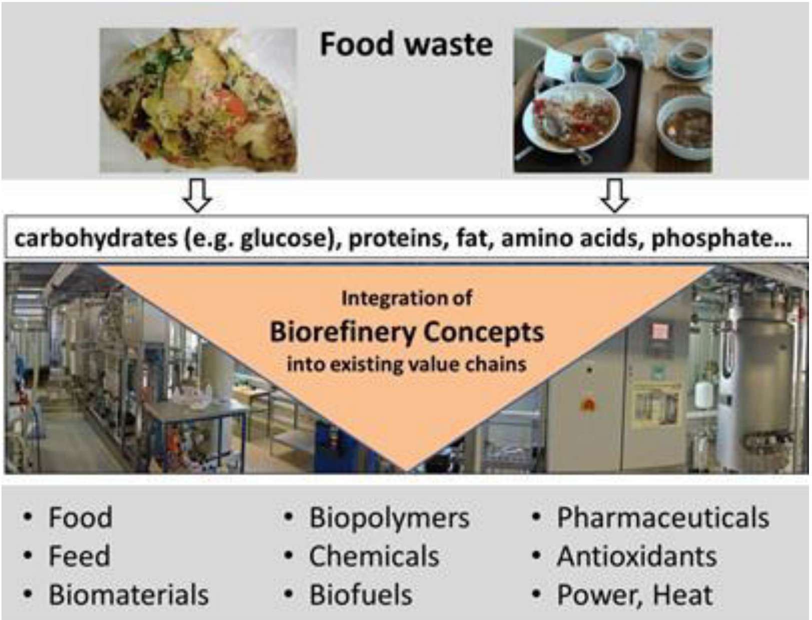

Figure 1: Integration of food waste treatment in biotechnological processes for the production of various bio-based products.

Cellulose, hemicellulose, lignin, proteins and phosphorus containing compounds in agricultural residues are mostly arranged in recalcitrant structures. That structure needs to be broken-up by a pre-treatment in order to make carbon, nitrogen and phosphorus compounds accessible to microbes. The pre-treatment includes the hydrolytic conversion of cellulose, hemicelluloses, proteins and phosphorus compounds into utilizable nutrients (e. g. pentose and hexose sugars, amino acids and phosphate) by chemical (acidic), biological and/or enzymatic methods. Pre-treatment methods are energy-intensive and the selection of an efficient method is crucial for the overall economy of a biotechnological process [6].

3. Results and discussion

Agricultural residues can be used to obtain sugar solutions that may be usefully exploited for the production of lactic acid through the following steps: a) pre-treatment, to break down the lignocellulosic structure, b) enzymatic hydrolysis, to depolymerize lignocellulose to fermentative sugars, c) sugar fermentation to lactic acid by lactic acid bacteria and d) separation and purification of lactic acid [7]. In this context different fermentation regimes were tested for the development of an innovative and environmental benign lactic acid production.
According to the difficulties mentioned in the mobilization of fermentable sugars a range of other, easy accessible substrates are suitable for subsequent fermentation processes (such as residues from fruit and vegetable processing, by-products from starch and sugar factories or from the baking industry). In the following, two examples are given for the use of different (agri/food) residues for lactic acid production via several process regimes and/or microorganisms.

3.1 Example: Green biomass

“Green Biorefinery” concepts facilitate the multiple use of fresh green biomass [8]. Originally, grass press juice was a residue from green fodder pellet production, but after identification of its valuable components, a number of approaches for its industrial utilization were developed. For that purpose fresh green material was pressed to separate the juice and the cake.
Generally, the grass press juices contain a variety of nitrogen compounds and inorganic salts, and can act as a substitute for synthetic nutrients in already existing processes. For instance, the Figure 2 shows the more efficient sugar consumption (in blue color) as well as the product formation (in red color) when grass press juices are used as nutrient sources.

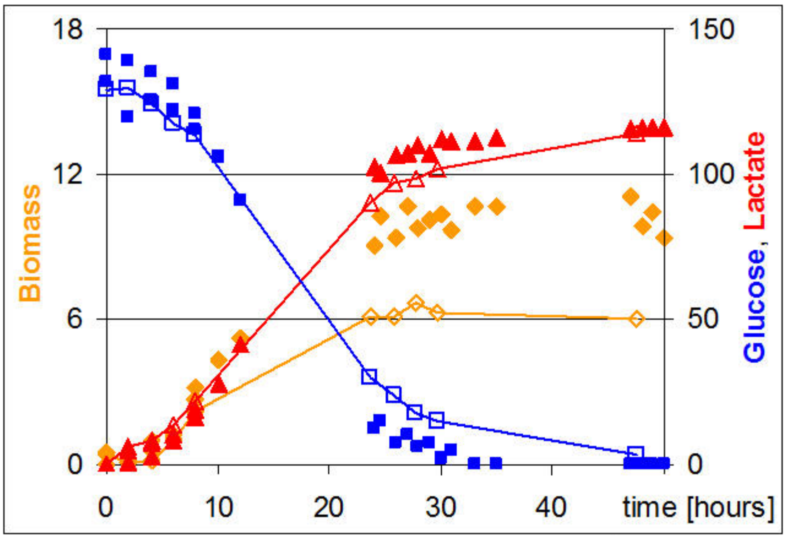

Figure 2: Typical time course of a batch lactic acid fermentation supplemented by conventional nutrients, and green juice as an alternative source (synthetic nutrients: open symbols; alfalfa green juice: solid symbols).

Another set of experiments performed with grass press juice from different harvesting sites in Germany revealed a maximum volumetric lactic acid productivity of 2.5 g L^{-1} h^{-1} and a yield of 0.66. It could further be shown that the composition of the fermentation broth (i.e. nitrogen content) has a strong impact on the product formation and sugar consumption [9]. Generally, the higher the concentration of nitrogen compounds in green press juice, the better the microbial performance.

3.2 Example: Bakery waste

Carbohydrates like starch, which is the main constituent of the bread dry weight, are preferably used as substrates/nutrients for several biotechnological (fermentation) processes. As part of a European project (BREAD4PLA) [10, http://www.bread4pla-life.eu/index.php] we have been carried out different analytical methods for the estimation of parameters which are basically characteristic in terms of the further use as main substrate for lactic acid fermentation processes.
In order to scale-up the lactic acid fermentation production two representative waste bread samples cake (sugar containing) and crust (starch containing) were processed in bioreactors up to bench-scale 75-L-vessel and pilot-scale between 450 and 1000 L, respectively.

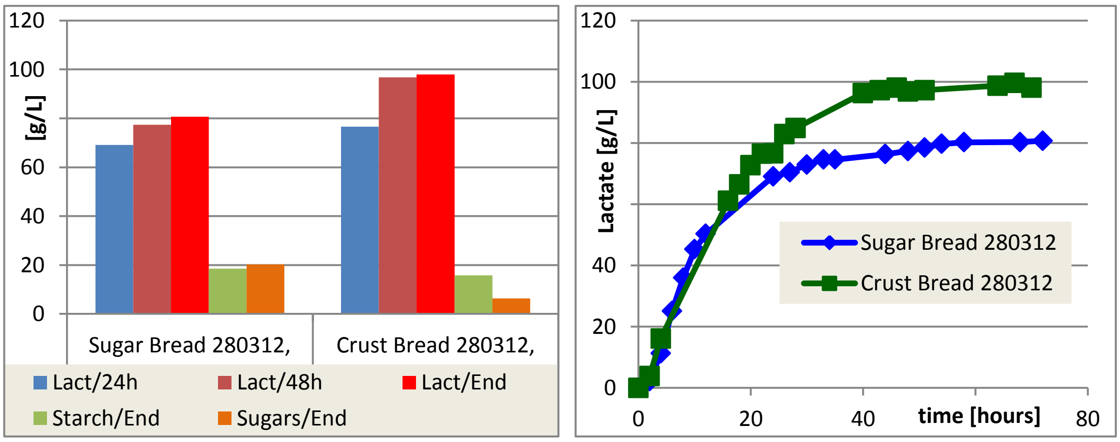

Figure 3: Comparison of the product formation (lactate) and substrate consumption (starch & sugars) for sugar containing bread and crust bread residues in 75-L-Bioreactor.

The figure 3 shows a summary of the results for the bench-scale bioreactor. From these data the crust bread seems to reach a better performance than the sugar bread with respect to the lactate formation and the utilization of the containing carbohydrates (starch and sugars). However, for the final evaluation of the best bakery waste feedstock it will be necessary to calculate the costs of the overall process (including downstream processing) including the enzymes and additional nutrients (e.g. yeast extract) together with the needs for the product quality (impurities) at the end. Lactic acid bacteria need besides the carbon source also a source of nitrogen and other nutrients, and a phosphorus source [10].
The viability of the production of lactic acid from bakery and pastry wastes has been demonstrated from laboratory up to pilot scale including the entire value chain starting from the raw material and resulting with a polymer-grade product (LA). According to the characterization of different waste bread samples a series of experimental trials have been carried out in order to convert the containing carbohydrates directly and/or after enzymatic hydrolysis. As a result of the achievements so far the optimization of pre-treatment, hydrolysis, fermentation, and downstream processing steps in parallel together with the screening of other LA producing bacteria have been performed.

4. Conclusion and outlook

New strains with new properties alone will not lead to more efficient production of LA, but only in interaction with new raw materials, progress in fermentation technology as well as downstream processing development. Because of the relatively low price of LA, one of the major challenges in its large-scale fermentative production is the cost of the raw material. Lactic acid can be produced from a wide spectrum of carbon sources including starchy materials, many food industry by-products (e.g. molasses, whey, fruit and vegetable residuals), agro-industrial residues and by-products (e.g. lignocellulose hydrolysates, corn stalks, bagasse) and various other renewable resources. Together with the need of low-cost carbon, there is an additional demand of suitable supplements, which should not cause additional costs and problems in view of impurities. Therefore, the kind of nutrients as well as the optimization of their concentration is essential. It is likely that one of the future trends in lactic acid production will end up in mixtures of different low-cost raw materials in order to avoid the use of expensive complex supplements [11].
The entire processing chain has been implemented: from the feedstock, the pre-treatment/hydrolysis for releasing C5 and C6 sugars, the fermentation to lactic acid and the downstream processing of fermentation broth to generate marketable lactic acid of high enantiopurity and quality. Exploitation of L(+)- and D(-) lactic acid for the production of biopolymers is one of the recent applications.

Acknowledgements

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References

[1] SpecialChem (2014) http://www.specialchem4bio.com/news/2014/05/23/global-lactic-acid-market-to-grow-at-a-cagr-of-15-5-from-2014-20-grand-view-research#sthash.McKc8RdH.dpuf. Accessed 18 Jun 2014
[2] Abdel-Rahman MA, Tashiro Y, Sonomoto K (2013) Recent advances in LA production by microbial fermentation processes. Biotechnol Adv 31(6):877–902
[3] Taskila S, Ojamo H (2013) The current status and future expectations in industrial production of lactic acid by lactic acid bacteria. In: Kongo M (ed) Lactic acid bacteria - R & D for food, health and livestock purposes. InTech, Rijeka, pp 615–632
[4] Menon, V.; Rao, M. (2012) Trends in bioconversion of lignocellulose: Biofuels, platform chemicals & biorefinery concept. Progress in Energy and Combustion Science 38: 522-550
[5] Pleissner D, Lau KY, Schneider R, Venus J, Lin CSK (2015) Fatty acid feedstock preparation and lactic acid production as integrated processes in mixed restaurant food and bakery wastes treatment. Food Research International 73: 52–61
[6] Pleissner, D.; Venus, J. (2014) Agricultural residues as feedstocks for lactic acid fermentation. - ACS Symposium Series, Vol. 1186 "Green Technologies for the Environment", Chapter 13, pp 247–263
[7] Moldes, A. B.; Torrado, A.; Converti, A.; Dominguez, J. M. (2006) Appl. Biochem. Biotechnol. 135: 219–227
[8] Kamm, B., Kamm, M., Venus, J.(2006) Principles of biorefineries: the role of biotechnology, the example lactic acid fermentation. – In: Edwin C. Hearns (Ed.), Trends in Biotechnology Research, pp. 199-223
[9] Papendiek, F.; Venus, J.(2014) Cultivation and fractionation of leguminous biomass for lactic acid production. Chem. Biochem. Eng. Q., 28 (3) 375–382
[10] Venus, J.(2014) Utilization of Waste Bread for Lactic Acid Fermentation. American Society of Agricultural and Biological Engineers Annual International Meeting 2014, Volume 1, 557-562
[11] Koutinas AA, Vlysidis A, Pleissner D, Kopsahelis N, Lopez Garcia I, Kookos IK, Papanikolaou S, Kwan TH, Lin CSK (2014) Valorization of industrial waste and by-product streams via fermentation for the production of chemicals and biopolymers. Chem Soc Rev 43:2587–2627

TD-O_04 Bioresource utilization as a challenge of coordination

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1. Objective

Bio-resources are a-typical goods from an economic point of view. Economics distinguishes between private and public goods and suggests markets and hierarchies as best fitting coordination mechanisms, respectively (Williamson (1985). However, given their discussed/contested nature, bio-resources can also be considered as common pool resource (Ostrom, 2005) with self-governance as mode of coordination. But self-governance is not an easy task given the numerous societal twists and debates on bio-resource utilization projects. The paper contributes to our knowledge of mitigating contested bio-resource utilization in practice.

2. Methodology

The paper is basically theoretical in focus. Drawing on institutional economic literature and the literature on bio-resource utilization, the paper explores the nature of bio-resources as an economic good and modes of coordination for their adequate application. Drawing on the IAD framework of Elinor Ostrom, the analysis concentrates on bio-resources as a common pool resource (a commons problem). The analysis examines the ways suggested by the IAD framework to cope with property, value and interest conflicts. The argument will be illustrated during the conference presentation of the paper.

3. Results and discussion

3.1 Bioresources as a discussed/contested resource

Bioresources are discussed/contested because of their alternative applications. They are resource for food and for many other purposes, products and services. The production of bioresources also competes for limited fertile land. Not only primary, but also secondary (organic residues) and tertiary bioresources are wanted for products and services and for that reason there is also competition for them, too (eseia, 2014). Adequate coordination is therefore needed if bioresources are to be used effectively and in a societally acceptable manner in a biobased economy. One approach to coordination is to cascade bioresources from high to low value applications: a type of coordination based on the physical and chemical properties of the material combined with the economic value of bioresources. Such an approach, however, does not account for the specificity of bioresources as an economic good. Based on the criteria of access and excludability, an economic good can be classified as private or public. Private goods are excludable, public goods are not and for that reason public goods are provided by the state and are accessible to all. However, bioresources of all generations (first, second, third) actually do not fit in this economic classification scheme because bioresources incorporate collective values, which are neglected in the economic process. The trees in a privately owned production forest are considered to be a private good, because they are privately owned and can be exchanged in bilateral private transactions. However, for the inhabitants of the nearby village, the trees and the forest provide them with a high quality landscape, which can be considered a collective or social value. The production of biogas by manure digestion is another example. In the Netherlands in particular, manure digestion is a highly contested option for processing the huge manure surplus in the Dutch agricultural system. For the farmer, the biogas installation is an economic necessity for the continuation of his business. The installation and products can be privately exchanged, but the biogas option encounters massive societal resistance grounded in a mixture of motives and reasons, which are quite often grounded in contested values (Upreti, 2004).

Simple as these examples are, they illustrate the value conflicts incorporated in the application of bioresources. Our expectation is that these value conflicts will be reinforced as the biobased economy and the circular economy develop. We argue that the core solution of the market-based economy, legal protection of private property rights, is too limited to cope with contest and rivalry for bioresource applications incorporating conflicts between private social, community and societal values, such as sustainability, landscape quality, etc. We argue that there is a need for new types of coordination and regulation beyond the distinction between private and public goods, because bioresources do not fit this classification. In the remainder of this paper we argue that considering bioresources as a common pool resources could help to find ways to mitigate the transactional value conflicts incorporated in bioresources.

3.2 Bioresources as a common pool resource

Ostrom (2005) suggested a useful classification of economic goods, which permits a good positioning of bioresources. See figure 1.

		Subtractability of Use	
		High	Low
Difficulty of excluding potential beneficiaries	High	*Common-pool resources*: groundwater basins, lakes, irrigation systems, fisheries, forests, etc.	*Public goods*: peace and security of a community, national defense, knowledge, fire protection, weather forecasts, etc.
	Low	*Private goods*: food, clothing, automobiles, etc.	*Toll goods*: theaters, private clubs, daycare centers

Figure 1: Four types of economic goods (source Ostrom 2005).

Based on the criteria of excludability and rivalry of use, Ostrom distinguishes between four types of economic goods. This adds to the distinction between private and public goods two lesser known types. Club, or toll goods, can be considered as private goods of a group of people. Entry into this group is by becoming a member or being invited as a member. The fourth type is the common pool resource, the type of economic good that is most interesting for positioning bioresources. Common pool resources are goods for which there is great rivalry for their usage, whereas at the same time it is very difficult to exclude potential users. This type of economic good represents the famous “tragedy of the commons”, stating that if each person acts rationally we shall all end up ruined.[2] Common pool resources typically represent goods for which individual rationality is highly competitive for optimum solutions at the collective level. In a way this is also the case with bioresources. The above examples show that the individually optimum solution, with private citizens owning the wood’s owner and the biogas producer, can be perceived as a private gain on account of collective values like quality of the landscape, quality of the living environment in the countryside, or sustainability. Ostrom’s work on the sustainable management of common pool resources can be an inspiration to find ways to deal with the value conflicts incorporated in bioresources.

3.3 Tailor made local coordination of bioresources

Ostrom, in collaboration with many scholars, has devoted her academic life to an empirical study of how people deal with the commons problems in local settings (Ostrom, 2005). Her starting point was the idea that there are more types of coordination than market and hierarchy (government control), and the idea of polycentricism, which refers to many centers of decision making in a certain area or sector, all formally independent of each other (Ostrom 2010, p. 643). Both points are also highly relevant to bioresources. Many different actors are involved with bioresources. In many cases they have no formal (contractual) relationships, meaning that there are many decision making units. Ostrom took this idea of polycentrism as a positive point and started investigating how people succeed or fail to derive sustainable solutions in the management of the common pool resource. She studied fishery communities to discover their solutions to balancing the private interest of the fisherman and the risk of overfishing (the collective good). She did the same for farmer communities to study their solutions in the management of irrigation. Numerous empirical studies were conducted in this way (Ostrom, 2005).
She processed her own and others’ numerous empirical studies into an economics-oriented institutional framework (Ostrom 2005), called the Institutional Development Framework (IAD). See figure 2.

[2] Ostrom’s reference in the CPR work is Hardin’s conclusion on the tragedy of the commons: “Each man is locked into a system that compels him to increase his herd without limit – in a world that is limited. Ruin is the destination towards which all men rush, each pursuing his own interest in a society that believes in the freedom of the commons” (Hardin, 1968, p. 1244)

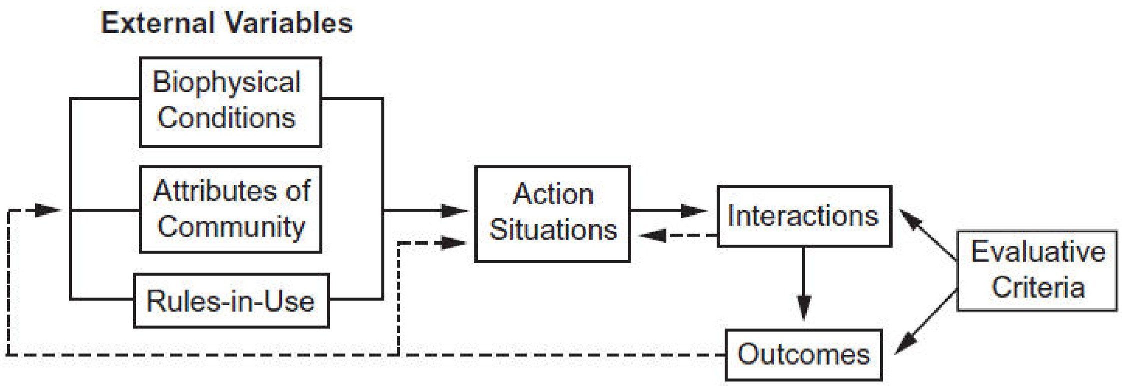

Figure 2: Institutional Development Framework of Elinor Ostrom (source Ostrom 2005)

We argue that this framework can be a guide to coordinating private and collective values in the biobased economy. Due to their features (low density and humidity), bioresources should be dealt with in local settings. Therefore, the local setting acts as an action situation, which– according to Ostrom – is conditioned by three sets of external variables: 1) The biophysical conditions, the local bioresource condition (wood, organic agricultural waste and industrial sectors), 2) the attributes of communities, which refers to specificities of tradition, culture and the social capital of the local community, and 3) "rules in use", a crucial part of the framework, the entrance point for making workable, balanced and sustainable local arrangements for the production, processing and application of bioresources.
The coordination of bioresources requires the development of new institutional arrangements able to balance the sustainable usage of bioresources, balanced local interests, and economically profitable transactions and outcomes. There are already institutionalized criteria of international organizations and national authorities for sustainable usage. For economically profitable transitions and outcomes we also have institutionalized economic standards and criteria. The basic mechanisms here are property and use rights of private goods and the contractual arrangements for their exchange (Fuchs 2003). The problem of balancing the values is more difficult because of the different perceptions of and engagements in bioresources. Here the societal or sustainability value of the bioresources is at stake, which is hard to quantify or to evaluate in a transactional relationship.[3] Markets and hierarchies cannot deal with these rival value positions, because both types of coordination optimize private and public values, respectively, each at the other's account. So, to balance a mixture of private and collective value interests, we still need to develop workable institutions. We argue that Ostrom's IAD framework can be an inspiration because it stresses the arrangement of two important ingredients: *process* of decision making and *content* of decision making. Incorporating the process arrangement is crucial because of procedural fairness, which happens to be a crucial factor in, for instance, the acceptance of innovative technologies (Huijts, 2013).

A contractual arrangement for local bioresource application could develop in four major steps.

- *Confrontation*: Explication of "what do I want". All participants unconditionally explain their core values and interests. This will bring the value conflicts to the table. Participants also commit to finding and reaching an agreement and arrangement together.
- *Identification:* Discussing, finding and agreeing common ground: What do we want to achieve in this local setting, and in what way?
- *Process arrangement:* Deciding an arrangement with rules on process and content. Rules are binding on the participants and need to be congruent with the legal jurisdiction.
- *Content arrangement:* Following the process arrangement, participants start working towards a commonly accepted outcome.

[3] Attempts have been made to evaluate societal values, but this is controversial.

The process arrangement step aims at an arrangement concerning how the participants are going to work together to find a commonly acceptable outcome. The process arrangement can be designed according to the following set of rules (Ostrom 2005):

1. Position rules (who takes part)
2. Boundary rules (entry and exit rules)
3. Scope rules (what type of outcome is at stake)
4. Authority rules (Who does what)
5. Aggregation rules (how to make decisions)
6. Information rules (how to communicate)
7. Pay-off rules (division of costs and benefits)

Ostrom considers these rules as the meta-language for institutional analysis, meaning that in real life, each rule type may have several empirical manifestations. But the rule types can also be used to design new institutional arrangements for balancing private and collective interests in the application of bioresources. The conference presentation will illustrate the rule-based framework with manure digestion and wood incineration examples.

4. Conclusion

Value and interest conflicts quite often frustrate the implementation of bio-resource utilization projects. Considering bio-resources as a common pool resource opens new perspectives to cope with such conflicts. The paper explains how the common pool perspective could add to the resolution of value conflicts in bioresource application. Part of the solution is to set up decision making about rules and procedures (the process) part of the coordination of bioresource utilization in local settings.

References

[1] eseia (2014), Innovation Challenge towards Rational Use of Bio-Resources in Europe. A Discourse book, Graz.
[2] Fuchs, D. A. (2003). An Institutional Basis for environmental Stewardship. The Structure and Quality of Property Rights. Dordrecht, Kluwer Academic Publishers.
[3] Huijts, N. M. A. (2013). Sustainable energy Technology Acceptance: a psychological perspective, Delft University of Technology.
[4] Ostrom,Elinor (2007), Governing the Commons. The Evolution of Institutions for Collective Action, New York, Cambridge University Press
[5] Ostrom, Elinor (2005), Understanding Institutional Diversity, Princeton, Princeton University Press
[6] Ostrom, Elinor (2010), Beyond Markets and States: Polycentric Governance of Complex Economic Systems, American Economic Review 100 (June 2010), p 641-672.
[7] Upreti, B. R. (2004). "Conflict over biomass energy development in the United Kingdom: some observations and lessons from England and Wales." Energy Policy **32**(6): 785-800.
[8] Williamson Oliver E. (1985): The economic institutions of capitalism, New York, the Free Press.

TD-O_05 Residual grass an overlooked bioresource – Environmental consequences of various conversion pathways

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1. Objectives

Recovery of biogas from organic residues is an acknowledged cost-effective mitigation technology for greenhouse gases [1]. Although agricultural residues have been widely studied [1,2], grass residues received little attention, remaining generally underutilized throughout Europe. The aim of this study is to assess the environmental consequences related to diverse uses of this substrate.

2. Methodology

A consequential life cycle assessment (LCA) was carried out for five management scenarios of meadow grass from natural areas, otherwise left un-harvested (reference management):

- Scenario 1 - biogas: The harvested grass is co-digested with 30% pig slurry; the digestate is stored and applied on farmland as a source of fertilizer and the biogas is used for combined heat and power (CHP) production, thereby avoiding marginal fertilizers, heat and electricity to be produced. Sensitivity analyses: (a) with extrusion pre-treatment for the grass (in which case the mixture becomes 50% slurry, 50% grass); (b) with extrusion, but only 1% slurry input (reflecting mono-digestion; the slurry input was considered necessary to ensure a stable digestion process). If not co-digested with the grass, it is considered that pig slurry would have otherwise been managed conventionally, i.e. stored and applied on land without any additional treatment, as described in [1].
- Scenario 2 - composting: The harvested grass is composted along with 15% wood chips in order to obtain a quality compost (open windrow composting), subsequently applied on household gardens. This compost does not replace mineral fertilizers; it was considered that without this (often free) compost, a lower yield for home gardens would simply be accepted. If not used for compost, the wood chips would have been used for CHP, thus avoiding marginal heat and electricity production.
- Scenario 3 – animal feeding: The harvested grass is used as animal feed, based on its protein and carbohydrate content. This involves that the grass replaces part of the protein and carbohydrate crops needed for animal feed. This scenario includes the indirect land use changes avoided as grass substitutes for wheat (marginal carbohydrate crop) and soy meal (marginal protein crop).
- Scenario 4 – integrated generation of solid fuel and biogas (IFBB): The harvested grass is here washed, stirred, heated and finally pressed where it is separated into two fractions: a liquid used for biogas production (co-digestion with straw) and a solid from which solid fuel pellets are produced (85% DM), as further described in [3]. Both the biogas and pellets are used for CHP. If not co-digested with the liquid fraction from the IFBB process, it is considered that straw would have been left on land and ploughed down.
- Scenario 5 – green biorefinery: The harvested grass is pressed, but the solid is here used for co-digestion with pig slurry, while the liquid is further processed (steam coagulation and decantation) to produce a protein-rich cake used for animal feed. As in scenario 3, indirect land use changes are included in this scenario.

All input and output flows were related to a functional unit being the management of one tonne of residual grass (meadow grass from natural areas). The geographical scope considered was Denmark, i.e. the data inventory for grass, manure management, straw and the applicable legislation were based on the Danish context. The life cycle impact assessment was carried out according to the EDIP 2003 methodology [4]. Background (or generic) LCA data were based on the Ecoinvent v.3 database, and the LCA was facilitated with the LCA software SimaPro 8. The key life cycle inventory (LCI) parameters considered in the LCA model are presented in Table 1.

Table 1: Key Life Cycle Inventory (LCI) parameters considered in the Life Cycle Assessment.

Parameter	Scenarios	Values and description	Reference
Anaerobic digestion	1-4-5	Two-steps mesophilic digestion; input mixture reaching 10% DM after the first digestion step; Fugitive losses, heat and electricity input, biogas composition and heating value modeled as in [1]	[1]
Pig slurry composition (ex-housing), BMP & LCI	1-5	Per tonne slurry: 69 kg DM; 5.3 kg N; 1.2 kg P; 2.9 kg K; 34 kg C; 55 kg VS BMP: 320 Nm^3 CH_4 per t VS Reference manure management: LCI as in [1]	[1]
Grass composition (freshly harvested) & BMP	1-2-3-4-5	Per tonne grass: 180 kg DM; 1.6 kg N; 0.6 kg P; 5.3 kg K; 72 kg C; 160 kg VS; BMP, fresh grass: 302 Nm^3 CH_4 per t VS BMP, extruded grass: 410 Nm^3 CH_4 per t VS	[5] [6,7]
Straw composition, BMP & LCI	4	Per tonne straw: 850 kg DM; 4.5 kg N; 0.8 kg P; 13 kg K; 380 kg C; 810 kg VS; BMP, straw: 195 Nm^3 CH_4 per t VS Reference straw management: ploughing, LCI as in [1]	[1]
Wood chips composition	2	Per tonne wood chips: 850 kg DM; 1.7 kg N; 1.4 kg P; 13 kg K; 420 kg C; 680 kg VS; Reference wood chips management: LCI as in [1]	[1,8]
Grass silage process	1-4-5	Modeled as the straw scenario in [1], but considering DM losses of 4.3% over 2 months, and CO_2 losses of 1.8% of the DM.	[1,9,10]
Marginal N, P and K fertilizers, electricity & heat (short-term)	1-2-3-4-5	N: Calcium ammonium nitrate; P: Diammonium phosphate; K: Potassium chloride; Heat: Natural gas-based domestic boiler; Electricity: coal-fired power plant	[1]
Avoided on-site grass decay	1-2-3-4-5	Modeled considering that: • 100% of the grass C would have been emitted as CO_2 • The difference in C sequestration between mature (unmanaged) grass and young (growing) grass corresponds to an avoided 5.5 kg CO_2 per t harvested grass	[11]
CHP	1-2-4-5	Only 90% of the net heat produced can be used (seasonal variation in demand) Gas engine efficiency: 40% for electricity; 46% for heat Efficiency, wood chip and IFBB pellets combustion: 27% electricity; 63% heat	[1]
IFBB process	4	Modeled essentially based on the data in [3]	[3]
Composting process	2	Modeled as described in [1]. Total N loss during process: 8% of the initial N, of which 15% is emitted as N_2O, 83% as NH_3 and the rest as N_2. Total C loss: 56% of the initial C (2% as CH_4, 0.3% as CO, the rest as CO_2)	[1]
Digestate/compost storage and application	1-2-4-5	Modeled as described in [1]	[1]
Indirect land use changes	4-5	Modeled as described in [2]	[2]

DM-dry matter; N-Nitrogen; P-Phosphorus; K-Potassium; C-Carbon; VS-Volatile Solids; CO_2-Carbon dioxide; CH_4-methane; CO- Carbon monoxide; N_2O-Nitrous oxide; NH_3-Ammonia: N_2-Nitrogen gas.

3. Results and discussion

Except composting (660 kg CO_2 eq. t^{-1} grass), all scenarios led to an improvement of the global warming potential (GWP_{100}), in comparison to the no-harvest situation (Fig. 1). Compared to (extruded) grass mono-digestion (-130 kg CO_2 eq. t^{-1} grass; scenario 1b), the IFBB system allowed an additional 50% GWP_{100} reduction, while the biorefinery led to twice the savings, and the animal feeding scenario to more than four times the savings of mono-digestion. For the latter two, it reflects the benefits of using grass for protein substitution (here soybean meal), and thus of reducing the pressure on land in sensitive ecosystems.

For all impact categories, composting led to increased environmental impacts, in comparison to the no-harvest situation (Table 2). This reflects the important loss of nitrogen (and particularly as N_2O, having a GWP_{100} ca. 300 times the one of CO_2) and carbon during the composting process. It also reflects the so-called lost alternative for the wood chips used during the composting process (purple bar in Figure 1); if not used for composting, it was considered that these would have been used for CHP, and would thus have avoided fossil resources (coal and natural gas) to be combusted. This involves that more fossil fuels are consumed in a scenario where grass is used for composting (compared to when grass is just left on land). Except composting, losses of nitrogen and phosphorus to freshwater were negligible for all scenarios. Acidification was increased for all scenarios but feeding, reflecting the production of ammonia during digestate handling. This, however, could be minimized through careful pH control. Similarly, the impact of the composting scenario on acidifying emissions could be improved by the use of a biofiler capturing the NH_3 releases (which could then be used as e.g. a fertilizer source).

Table 2: Characterized net LCA results. All figures are expressed per functional unit (1 t freshly harvested meadow grass).

Impact category	Unit	Sc. 1	Sc. 1a	Sc. 1b	Sc. 2	Sc. 3	Sc. 4	Sc. 5
Global warming (100 y)	kg CO_2 eq.	-110	-170	-130	660	-570	-200	-290
Acidification	m^2 UES	1.9	11	12	110	-3.5	22	11
N losses to freshwater	kg N	-0.013	-0.086	0.34	0.98	-0.32	0.32	-0.79
P losses to freshwater	kg P	-0.0059	-0.0031	-0.0017	0.086	-0.022	0.0056	-0.013

UES-unprotected ecosystems equivalent; Sc.-scenario

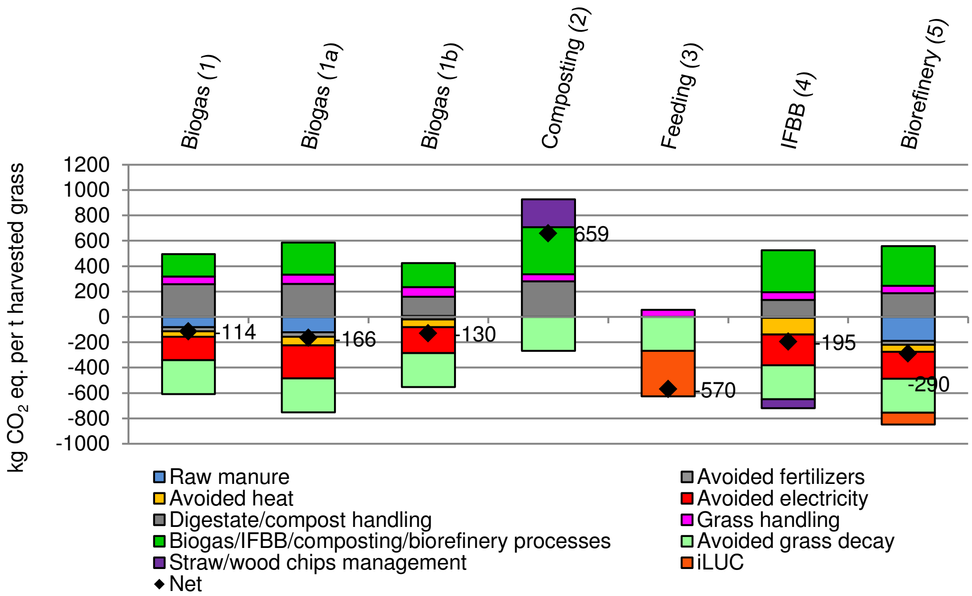

Figure 1: Breakdown of the LCA results for the Global Warming Potential (100 y).

Figure 1 highlights the benefits, in terms of GWP_{100}, of avoiding grass to decay on-site, and such benefits also applied for all other impact categories. An additional benefit of harvesting the grass, though not reflected in the LCA, is the enhanced biodiversity; by cutting and harvesting the grass, a new opportunity is created for other plant species to prosper. Figure 1 further reflects, as emphasized in previous studies (e.g. [1]), the tremendous environmental benefits of digesting manure instead of managing it conventionally; the scenarios triggering more manure to be digested thus lead to additional environmental benefits. Similarly, the scenarios triggering more electricity production (1a, 4) led to important environmental benefits (Fig. 1).

It is important to understand that the energy-related benefits shown in Fig.1 are directly related to the source of marginal energy avoided by the biogas or pellets produced. In a future extreme renewable energy context with a high share of fluctuating power (wind), the avoided energy would have GWP credits close to zero. However, biogas, being storable, would not displace continuous but flexible power [12], when not used for transport. As shown in [12], the marginal source of flexible power may come from biomass with potentially high environmental impacts. In such setting, producing pellets from the grass (scenario 4) appears less advantageous in comparison to biogas production, since these can only be used for continuous CHP production.

One important limitation of this study is that it was considered that 100% of the carbohydrate and protein value of the grass could substitute for marginal carbohydrate (wheat) and protein crop (soy), respectively. In reality, it is not the crude protein content that matters the most for animal feeding, but the quality of the protein in terms of amino acids. Further, the overall quality of the grass (contamination, etc.) may not meet the standards required to accept it 100% as a direct feed source (as assumed in scenario 3), so the benefits of scenario 3 may be slightly overestimated. Through this is not expected to change the overall ranking between scenarios, the model would benefit from refinements on these aspects.

4. Conclusion and outlook

In the context of a large-scale biogas deployment strategy, this study showed that it is more beneficial, on an environmental perspective, to use residual grass as a substrate for biogas than leaving it on land to decay. In fact, all management scenarios of residual grass, with the exception of composting, led to increased environmental benefits in comparison to today's no-harvest situation, for all impact categories assessed. Alternatives allowing to recover a maximum of protein (animal feeding, green biorefinery) generated the greatest environmental benefits, essentially due to the avoided land use changes this creates. The green biorefinery concept, allowing to simultaneously recover substantial energy, protein and fertilizers, was shown as a promising avenue for managing residual grass. Future work quantifying the cost-effectiveness of these biogas and biorefinery strategies to manage residual grass would be particularly helpful to support future investment decisions.

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References

[1] Hamelin L, Naroznova I, Wenzel H. (2014). Environmental consequences of different carbon alternatives for increased manure-based biogas. Appl Energy;114:774–82. doi:10.1016/j.apenergy.2013.09.033.

[2] Tonini D, Hamelin L, Astrup TF. (2015) Environmental implications of the use of agro-industrial residues for biorefineries: application of a deterministic model for indirect land-use changes. GCB Bioenergy: doi:10.1111/gcbb.12290.

[3] Hensgen F, Bühle L, Donnison I, Frasier M, Vale J, Corton J, et al. (2012). Mineral concentrations in solid fuels from European semi-natural grasslands after hydrothermal conditioning and subsequent mechanical dehydration. Bioresour Technol;118:332–42.

[4] Hauschild M, Potting J. (2005). Spatial Differentiation in Life Cycle Impact Assessment—The EDIP 2003 Methodology. Copenhagen, Denmark.

[5] Møller J, Thøgersen R, Kjeldsen AM, Weisbjerg MR, Søegaard K, Hvelplund T, et al. (2000). Fodermiddeltabel. Sammensætning og foderværdi af fodermidler til kvæg. Aarhus, Denmark.

[6] Triolo JM, Pedersen L, Qu H, Sommer SG. (2012). Biochemical methane potential and anaerobic biodegradability of non-herbaceous and herbaceous phytomass in biogas production. Bioresour Technol;125:226–32.

[7] Hjorth M, Granitz K, Adamsen APS, Moller HB. (2011). Extrusion as a pretreatment to increase biogas production. Bioresour Technol;102:4989–94.

[8] ECN-Biomass (2012). ECN Phyllis database. The composition of biomass and waste. Version 4.13. 2012. http://www.ecn.nl/phyllis/ (accessed May 15, 2015).

[9] Pakarinen O, Lehtomaki A, Rissanen S, Rintala J. (2008). Storing energy crops for methane production: Effects of solids content and biological additive. Bioresour Technol;99:7074–82.

[10] McDonald P, Henderson AR, MacGregor AW. (1968). Chemical changes and losses during the ensilage of wilted grass. J Sci Food Agric;19:125–32.

[11] Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, et al. (2008). Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science;319:1238–40.

[12] Wenzel H, Høibye L, Duban Grandal R, Hamelin L, Bird DN, Olesen A. (2014). Carbon footprint of bioenergy pathways for the future Danish energy system. Main report. http://www.ens.dk/sites/ens.dk/files/undergrund-forsyning/vedvarende-energi/bioenergi/analyse-bioenergi-danmark/carbon_footprint_of_bioeneergy_pathways_for_the_future_danish_energy_system_-_final_210314_l.pdf (accessed May 15, 2015).

TD-O_06 Influence of food waste characteristics on treatability through anaerobic digestion

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1. Objectives

This communication first aims at describing variations on food waste (FW) characteristics, in a general way and according to their localisation, collection source, season of collection and typology. Then, a study of the link between these characteristics and FW degradability was performed.

2. Methodology

This study was performed following 2 steps:

- Bibliographical compilation: A compilation of FW characteristics was made from 102 different samples, proposed in 70 studies extracted from scientific literature. A statistical study was performed using Statgraphics Centurion XVI ® software. The Kruskal–Wallis test was used to determinate the total variation of each FW characteristic. The Bonferroni correction was used to specify the categories that may generate the variations. The influence of FW characteristics mean values variation on the treatability through anaerobic digestion (AD) was commented based on literature observations.
- Experimental determination: FW were collected from a large canteen (RIA), a vegetarian restaurant (VEG) and a sourced-separated household food waste (SMICTOM). SMICTOM is collected once a week and some degradation of the waste might have already occurred before collection. 2 samples of each source were collected. A large characterisation in FW was performed, as shown in Table 1. Anaerobic biodegradability was measured by Biochemical Methane Potential (BMP) according to Standard methods. Aerobic biodegradability was measured at 40°C, with an air circulation of 70 $L.h^{-1}$, as developed by Tremier et al. [1]. Finally, a study of correlations of the determined characteristics was performed using Pearson matrix.

3. Results and discussion

3.1 Bibliographical compilation

Variation factors for FW characteristics: Compiled FW characteristics demonstrated to be highly variable as shown in Table 1. Only 2 characteristics may be considered as representatives to the whole FW: pH (5.1) and BMP (460 $NLCH_4.kgVS^{-1}$). These values can be used reasonably, whatever the type of FW, for process design or environmental assessment. Volatile solids (VS) showed a low coefficient of variation (CV) but did not follow a normal distribution. Thus it was not considered as representative. Neither of the other characteristics means values may be considered as representative. Thus causes of variations were searched for these characteristics.
The characteristics with a CV higher than 33% were analysed with the Kruskal–Wallis test based on 3 categories of variation: continents of analysed FW, collection sources and season of collection. The category “continent” showed an impact in 3 characteristics: total solids (TS), total ammonia nitrogen (TAN) and sodium content (Na); the category “collection source” impacts 6 characteristics: TS, VS, cellulose (CEL), carbon content (C), oxygen content (O) and TAN; and the category “season” impacts 3 characteristics :C, nitrogen content (N) and carbon-to-nitrogen ratio (C/N).
To perform the Bonferroni correction, subcategories were identified as follows: the category “continents” were composed by: Asia, North America (NA) and European Union (EU); “collection sources” by: restaurant food waste (RFW), household food waste (HFW), FW mixed with green waste (FWGW), FW of large producers (FWLP) and organic fraction of municipal solid waste (OFMSW) and “season of collection” by: summer and winter. No significant statistical differences were evidenced between the FW from RFW, HFW and FWLP. Therefore; OFMSW and FWGW show the largest impacts in characteristics, mainly because of impurities and GW content. Significant differences were stated as follow: TAN and Na from Asia; TS from EU; VS, O and CEL from OFMSW; TS, C and TAN from FWGW; and C, N and C/N from winter and summer. However, the chosen categories explain the variations of only 9 of the 37 in the literature compilation, thus other studies might be necessary to clarify the origins of variations in the other characteristics.

Table 1: FW characteristics means, standard deviation in parenthesis.

Characteristics	Acronym / unit	Literature compilation		RIA	VEG	SMICTOM
Physiochemical		$\bar{x}$ (σ)	CV (%)	$\bar{x}$ (σ)	$\bar{x}$ (σ)	$\bar{x}$ (σ)
pH	pH / -	5.1 (0.7)	13.7	5.2 (0.0)	5.9 (0.9)	4.7 (0.2)
Porosity	ø / %	-	-	67.9 (5.9)	59.6 (10.5)	44.1 (13.1)
Density	ρ / $kg.l^{-1}$	-	-	364.4 (63.0)	505.0 (18.3)	506.9 (37.0)
Total solids	TS / $g.kgWW^{-1}$	227.8 (99.6)	43.7	310.1 (23.2)	200.9 (10.7)	337.4 (5.3)
Volatile solids	VS / $g.kgTS^{-1}$	882.3 (81.7)	9.3	940.1 (19.5)	842.5 (94.7)	856.0 (36.3)
Total COD	CODt / $g.kgVS^{-1}$	1010.0 (89.0)	44.8	1438.5 (72.1)	1796.4 (7.1)	1169.1 (37.1)
Soluble COD	CODs / $g.kgVS^{-1}$	505.0 (157.0)	31.1	301.2 (0.9)	507.6 (65.4)	199.9 (42.2)
Biogas	Biogas / $NL.kgVS^{-1}$	-	-	763.8 (58.0)	858.3 (38.6)	653.1 (39.6)
BMP	BMP / $NLCH_4.kgVS^{-1}$	460.0 (87.6)	19.0	397.4 (31.6)	420.8 (42.0)	371.8 (27.0)
Methane content	CH_4% / %	-	-	52.0 (0.2)	49.0 (2.7)	56.9 (0.7)
Total Kjeldahl Nitrogen	TNK / $g.kgVS^{-1}$	18.1 (14.6)	80.7	21.1 (0.6)	20.9 (5.2)	26.9 (3.0)
Total Ammonia Nitrogen	TAN / $g.l^{-1}$	0.8 (1.0)	125.0	0.3 (0.1)	0.8 (0.2)	1.4 (0.2)
Microbial population						
Bacteria	- / copy number $16S.g^{-1}$	-	-	2.7E9 (3.0E8)	6.8E9 (6.3E9)	1.9E10 (1.9E8[a])
Fungi	- / copy number $18S.g^{-1}$	-	-	1.0E8 (2.4E7)	8.2E7 (6.8E7)	3.6E8 (4.2E6[a])
Elementary						
Carbon	C / %VS	51.5 (8.5)	16.5	52.5 (0.3)	64.1 (0.7)	51.7 (2.0)
Nitrogen	N / %VS	3.1 (1.3)	41.9	2.3 (0.2)	2.4 (0.5)	3.0 (0.4)
Oxygen	O / %VS	37.6 (7.6)	20.2	38.9 (1.3)	43.4 (3.6)	38.8 (3.3)
Hydrogen	H / %VS	7.9 (3.7)	46.8	6.7 (0.4)	7.3 (0.1)	7.5 (0.5)
Sulphur	S / %VS	0.4 (0.2)	50.0	0.2 (0.1)	0.2 (0.1)	0.2 (0.0)
Carbon-to-nitrogen	C/N / -	18.5 (5.9)	31.9	22.4 (1.4)	26.4 (5.8)	17.4 (3.1)
Biochemical						
Total Carbohydrates	- / %VS	61.1 (23.7)	38.8	69.7 (0.8)	68.5 (2.0)	64.3 (0.8)
Proteins	- / %VS	21.0 (13.0)	61.9	13.0 (0.5)	12.7 (3.4)	15.9 (2.0)
Fats	- / %VS	15.4 (8.0)	51.9	17.3 (0.3)	18.8 (1.4)	19.8 (1.2)
Hemicellulose	HEM / %VS	9.4 (5.0)	53.2	24.7 (15.6)	21.2 (4.9)	17.0 (1.8)
Cellulose	CEL / %VS	8.7 (8.2)	94.3	16.4 (0.7)	13.6 (7.4)	18.4 (2.9)
Lignin	LIG / %VS	6.6 (6.1)	92.4	11.0 (10.1)	15.3 (7.1)	4.9 (0.5)
Micronutrients						
Chloride	Cl^- / $g.kgVS^{-1}$	-	-	6.2 (0.0)	9.3 (6.5)	4.6 (2.2)
Nitrite	NO_2^- / $mg.kgVS^{-1}$	-	-	0.0 (0.0)	0.9 (0.7)	0.5 (0.3)
Nitrate	NO_3^- / $mg.kgVS^{-1}$	118.8 (49.7)	41.8	7.3 (10.3)	396.8 (196.2)	11.3 (13.1)
Sodium	Na / $g.kgVS^{-1}$	22.9 (29.0)	126.6	7.1 (0.2)	9.8 (2.9)	4.1 (3.0)
Potassium	K / $g.kgVS^{-1}$	13.6 (7.0)	51.5	10.0 (1.0)	26.0 (3.0)	11.7 (11.1)
Calcium	Ca / $g.kgVS^{-1}$	18.1 (13.0)	71.8	3.3 (0.1)	3.9 (0.2)	3.4 (1.1)
Magnesium	Mg / $g.kgVS^{-1}$	2.3 (2.0)	87.0	0.4 (0.0)	0.8 (0.2)	0.4 (0.5)
Metals						
Iron	Fe / ppm TS	482.5 (815.0)	168.9	-	-	-
Molybdenum	Mo / ppm TS	1.2 (1.1)	91.7	-	-	-
Zinc	Zn / ppm TS	86.6 (110.3)	127.4	-	-	-
Chromium	Cr / ppm TS	27.7 (71.4)	257.8	-	-	-
Copper	Cu / ppm TS	23.5 (37.2)	158.3	-	-	-
Nickel	Ni / ppm TS	9.9 (25.2)	254.5	-	-	-

RIA-large canteen food waste; VEG-vegetarian restaurant food waste; SMICTOM-sourced-selected household food waste; WW-wet weight; COD-chemical oxygen demand; NL-normal litres (0°C, 1 atm); BMP-biochemical methane potential. [a] – max error

Consequences of FW characteristics variation on their treatability through anaerobic digestion:
Regarding the general means, FW confirmed to be favourable for anaerobic digestion (AD). The VS (882.3±81.7 g.kgTS^{-1}), the total chemical oxygen demand (CODt) (1010.0±89.0 g.kgVS^{-1}) and BMP (460.0±87.6 NLCH_4.kgVS^{-1}) are higher than values expected for other traditional waste used in anaerobic digestion as cattle manure, sewage sludge and GW. However, special attention must be paid to pH, TAN and the carbohydrates contents to avoid process instability. pH value (5.1±0.7) is suitable to fermentative bacteria but is low compared to the optimal conditions of methanogen microorganisms (6.5 to 7.2). The TAN (0.8±1.0 g.l^{-1}) is higher than the value considered as optimal, but it is not yet considered as inhibitory. The high carbohydrate contents (61.1±23.7 %VS) may be rapidly accumulation of volatile fatty acids (VFA), acidifying the reactor.
Furthermore, the specific variations may induce some process instabilities. High TAN concentration in FWGW (2.6 g.l^{-1}), is considered like inhibitory and might cause a 50% reduction in the methane yield, even at low organic loading rate. Process instability caused by TAN often results in VFA accumulation, decreasing the pH. The high CEL from the OFMSW (25.2±0.0 %VS) might be degraded incompletely. According to Diaz *et al.* [2], the limited oxygen supply in the early hydrolysis step limits the hydrolysis of CEL, resulting in a production of methane during the first days but with an incomplete transformation. Finally, the C/N of FW collected during summer (13.6±4.7) is lower than the minimum advised for an accurate digestion process (15-20), and then nutritional deficits might be present.

3.2 Experimental determination

Experimental determinations show some significant differences with values from literature: Higher values for CODt on RIA, VEG and SMICTOM; CODs on RIA and SMICTOM; and hemicellulose (HEM), C and K on VEG and lowers values for NO_3^- on RIA and SMICTOM and Ca on RIA, VEG and SMICTOM.
The high experimental values of CODt and CODs showed that some FW have higher capacities of degradation but it also could mean that interferences in the calculation could have occurred. The high C and HEM content in VEG might be linked with their typology composition variations. The lower levels in Ca for RIA, VEG and SMICTOM might indicate that this measure was not performed in the same way as in the literature compilation, or that a nutrient deficit was present.

3.2.1 Typology influence in FW characteristics

As concluded in the literature review, geographical origins, source of FW and seasons cannot explains all the characteristics variations. Thus in the experimental study, correlations between physicochemical, biochemical and typological composition were searched.
To complete the bibliographical compilation data, a biochemical characterisation based on a modified Van Soest method was performed, as proposed by Tremier *et al.* [3]. It discretized the organic matter (OM) in a water soluble fraction (SOLe) and a water non-soluble fraction (NSOL). SOLe, in turn, is subdivided in molecules of size upper and lower than 1.5 kDa (SOLe>1.5kDa and SOLe<1.5kDa respectively), on the hypothesis that only molecules lower than 1.5 kDa can diffuse through the cellular membrane to be used for microbial metabolism. Molecules of superior size must be degraded by means of exoenzymes to be consumed. NSOL is divided in OM soluble in neutral detergent (SOLNDF), HEM, CEL and lignin (LIG). Figure 1 resumes this characterisation for RIA, VEG and SMICTOM.

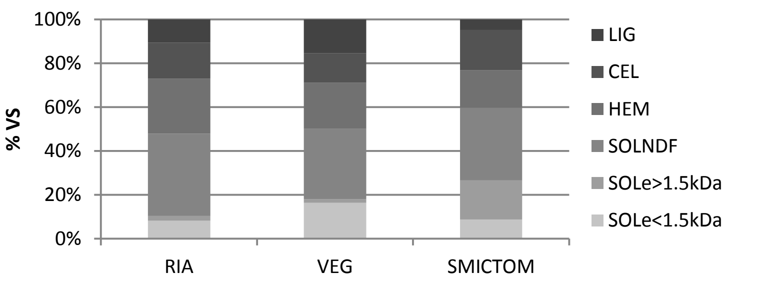

Figure 1: Biochemical fraction of RIA, VEG and SMICTOM

Concerning the typological fractions, results were: paper (15.0±8.9%WW), vegetables (31.3±20.6%WW), fruits (19.6±11.7%WW), starchy food (18.7±5.9%WW) meat (4.5±4.0%WW) and others (10.8±2.4%WW), which included milky preparations, sauces, bones, shells, coffee, green waste and biodegradables bags.

Using the Pearson matrix we found positive relations between: paper impact density; vegetables impact density, VS and SOLNDF; fruits impact VS and starchy food impact HEM. Thus, experimental results indicate that typology characterisation of FW may explain a part of characteristics mean values variations that were not identified on the bibliographical compilation.

3.2.2 Biodegradation characteristics

Based on the aerobic degradability, the cumulated oxygen consumption (AC) of FW was measured. AC indicates the total oxygen used to consume the available biodegradable OM in FW. The AC for RIA was 932.2 $gO_2.kgVS^{-1}$; for VEG 1040.6 $gO_2.kgVS^{-1}$ and for SMICTOM 869.6 $gO_2.kgVS^{-1}$. Making the ratio between AC and CODt, we found the proportion of biodegraded OM in aerobic conditions. We obtained 65% for RIA, 58% for VEG and 74% for SMICTOM. That means that CODt in SMICTOM is more available to be degraded. In the same way, the biodegradability ratio in anaerobic conditions (BD) between the experimental BMP and the theoretical BMP, calculated from the CODt (where 1 gram of degraded CODt produces 350 mL of methane), is 79% for RIA, 67% for VEG and 91% for SMICTOM. Again, SMICTOM seems to be more available to be degraded in anaerobic conditions, allows us to conclude that aerobic and anaerobic degradation have similar behaviours but anaerobic degradation seems to be more complete.
The difference showed by SMICTOM wastes might be explained by the aerobic biodegradation already underwent before collection, as proved by lower CODt, biogas production, CH_4 content (CH_4%) and by biochemical fractionation (more SOLe>1.5kDa, less SOLe<1.5kDa, less HEM). It seems that this fractionation and the oxidation followed-up by SMICTOM improved CODt degradation and avoid some process instabilities. Nevertheless, a controlled aeration might be applied to avoid the excessive OM degradation as was stated in SMICTOM.

3.2.3 FW characteristics correlation with biodegradation characteristics

Correlations of biodegradation analysis with the experimental FW characteristics are shown in Table 2. A higher pH shows to be favourable to biogas production and BMP. CODt was correlated positively to the biogas production as awaited, which support the use of this parameter to predict the FW biodegradation. Controversially, CODt was correlated negatively with CH_4%. It probably means that a part of the CODt is degraded quickly, especially the carbohydrates, producing more carbon dioxide (CO_2). Nevertheless, high presences of SOLe>1.5kDa seems to favour methane production, as state in SMICTOM. In the same way, it was stated that bacteria are linked negatively with carbohydrates, and fungi positively with SOLe>1.5kDa, showing probably a bacteria development by carbohydrates consumption and a fungi development that generate SOLe>1.5kDa from NSOL fractions by exoenzymes excretion. These results confirm the bibliographical observations to the treatability of FW through AD. However, a study of the biodegradation mechanisms of biochemical fractions and the biomass and enzymatic activity may help to understand clearly the AD of FW and the perspectives of optimization of the process.

Table 2: Correlation coefficient of FW characteristics based with some biodegradability characteristics.

	FW Characteristic	**CC**
	CODt	0,96**
Biogas	pH	0,87*
	C	0,81*
BMP	pH	0,87*
	TS	0,86*
	SOLe>1.5kDa	0,88*
CH_4%	BD	0,99**
	CODt	-0,9**
	Carbohydrates	-0,82*
Bacteria	Carbohydrates	-0.99**
Fungi	SOLe>1.5kDa	0.94*

Where; CC - Correlation coefficient; * - probability value below 0.05, ** - probability value below 0.01

4. Conclusion and outlook

In this study, FW characteristics demonstrated to be highly variable, that can be partially explained by the continent of collection, collection source, season variations and typological characterisation. It is necessary to be cautious in the utilisation of literature FW characteristics values for process design or environmental assessment. In general, FW proved to be very favourable to AD by its CODt, VS and BMP but presented some attention points. Is advised to identify the pH, TAN, carbohydrates, CEL and C/N content on FW and apply corrections or pre-treatments in case of inhibitory values. CODt, pH, and the biochemical fractions seem to be closely linked with biodegradability characteristics. AD in FW with low carbohydrates and high Sol>1.5kDa content seems to enhance the methane production and the biomass development, stated specially in samples with an initial aerobic pre-digestion. More detailed studies of these characteristics are advised to design most adapted treatments conditions to avoid instabilities.

References

[1] Tremier, A., et al. (2005): A respirometric method for characterising the organic composition and biodegradation kinetics and the temperature influence on the biodegradation kinetics, for a mixture of sludge and bulking agent to be co-composted, Bioresource Technology 96(2), 169-180.

[2] Diaz, I., et al. (2011): Effect of microaerobic conditions on the degradation kinetics of cellulose, Bioresource Technology 102(21), 10139-42.

[3] Tremier, A., et al. (2014): Characterization of the biochemical composition of solid organic resources to predict their aerobic biodegradation kinetics, 5th International Conference on Engineering for Waste and Biomass Valorisation, Rio de Janeiro, Brazil.

TD-O_07 Design of microalgae process for nutrient extraction from digestate through laboratory tests and modelling

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1. Objectives

Some previous researches studying nutrients extraction from digestate to produce microalgae biomass indicate an algae productivity varying from 2 to 360 mg L^{-1} d^{-1} and a N removal rate between 1 and 45 mgN L^{-1} d^{-1} using slurry or digestate [1] [2] [3]. However, no clear result was found concerning the influence of the operating parameters and the processes involved explaining such variations. Thus, the aim of this study was to understand and quantify the algae productivity and the processes involved in N removal according to the operating parameters applied in order to be able to design such a process.

2. Methodology

2.1 Laboratory pilot description

The parameters studied under controlled conditions at laboratory scale were (T1) the turbidity of the influent considered through the optical density (O.D.), (T2) the light provided, (T3) the N:P ratio of the influent, (T4) the level of CO_2 addition and (T5) the solid retention time (SRT) of microalgae in the bioreactor. All results obtained allowed to adapt and calibrate models from literature (as Contois and Monod kinetics) and finally, the model developed using Scilab software was used to design a real-scale treatment system.

The experimental set up consists of six 2.5L tubular reactors. The temperature was maintained at 25°C and the mixing set at 230 rpm. pH was monitored and regulated at 7±0.5 by the addition of pure carbon dioxide for trials T1, T2, T3 and T5. In addition, CO_2 was added at 45-min intervals during the day to avoid carbon limitation in the absence of regulation. To avoid acidification, which had been observed during preliminary tests when microalgae growth was limited, the pH was adjusted by adding 1.25 mL of 1M NaOH every second day to each reactor. For T4, pH regulation was performed using HCl (3N). Light was supplied by fluorescent bulbs and regulated according to the photosynthetic photon flux at the surface of the reactors. This was set at 50, 150 and 250 µmolE m^{-2} s^{-1} by shifting the light source to different distances from the reactor for trials T1 and T2 and at 250 µmolE m^{-2} s^{-1} for trials T3, T4 and T5. A 12h/12h light/dark regime was used for T1, T2, T3 and T5 while a 24h light regime was used for T4.

Figure 1: Experimental set up.

2.2 Microalgae inoculum

A complex phytoplanctonic ecosystem dominated by *Scenedesmus* sp. and *Chlorella* sp. was used for all experiments. It comes from an open high rate algal pond (INRA-LBE, France) dedicated to the treatment of synthetic wastewater. Other algae genera are present in small amounts along with bacteria.

2.3 Influent characteristics and trials description

The fixed O.D. levels inside the reactor at the beginning of the experiment were set on the basis of O.D. at 680 nm and varied between 0.2 and 1.3 for T1 and T2 and were fixed to 0.5, 0.2 and 0.8 for T3, T4 and T5, respectively. These values were obtained by adding different quantities of digestate in a growth medium. The composition of the growth medium was adapted from a "standard" 1/10 diluted digestate to obtain similar concentrations irrespective of the experimental conditions. Nutrients and trace elements were equilibrated by adding salts and Z8 media [4, 5] based on the amount of digestate added to achieve the desired color. Only nitrogen and phosphorus concentrations varied between the different trials. Initial NH_4^+ concentrations were fixed to 100, 100, 190, 60, 100 mgN.L-1 for T1, T2, T3 , T4 and T5, respectively, whereas PO_4^{3-} concentrations were fixed to 1, 1, 10, 18 mgP L^{-1} for T1, T2, T4 and T5 respectively and varied between 3 and 76 for T3.
Finally, for each run, the reactor was filled with microalgae inoculum (approximately 10% v:v to reach about $8.5x10^3$ microalgae mL^{-1}), digestate and growth media were supplemented with trace elements to reach the desired conditions, after which the light regime and pH regulation was activated. Trials T1, T2, T3 and T4 were carried out in "batch" mode and lasted 14 days while T5 was carried out in "semi-continuous" mode with solid retention time (SRT) varying from 2.5 to 4.5 days. Performances of the system, including mainly microalgae productivity and nutrients behaviours, were determined using average data between day 2 and day 14 for batch trials and data observed during at least 4 days after reach equilibrium for the semi-continuous trial.

2.4 Analytical measurements

Microalgae cells were counted manually using a Neubauer cell counting chamber each day allowing to determine microalgae growth and consequently productivity. Cations (Na^+, NH_4^+, K^+, Mg^{2+}, Ca^{2+}) and anions (Cl^-, NO_2^-, NO_3^-, PO_4^{2-}, SO_4^{2-}) were analysed every two days using a ionic chromatography. PCR, cloning and Capillary Electrophoresis-SSCP analysis were combined and performed to identified microalgae genus at the beginning and at the end of each trial.

3. Results and discussion

3.1 Microalgae productivity

As illustrated on Figure 2a, the combination of O.D. and light greatly influences microalgae growth during the batch trials. Consequently, maximum algae productivity varied between 0.3 and 139 mg L^{-1} d^{-1}, with the minimum obtained for O.D. and a flux of photon of 1.3 and 50 µmolE m^{-2} s^{-1}, respectively, while maximum was obtained for 0.2 and 250 µmolE m^{-2} s^{-1}. The self-shading effect is more important on light penetration than optical density for low O.D. while for higher O.D. the effluent turbidity limit the light accessibility by microalgae. Given this results, it seems acceptable to cultivate microalgae in a medium having an O.D. (680 nm) up to 0.6. In culture, the mixing has to be sufficient and the hydrodynamic of the system has to be adapted to allow a good light accessibility.
On the contrary, no impact of N:P ratio was observed on the microalgae growth during the 14-days batch tests (Figure 2b). In fact, N:P ratios varying from 3 to 76 gN gP^{-1} resulted in the same growth kinetic ($P < 0.05$; Student's t test) for the course of experiments. The maximum final microalgae concentration reached was $1.0x10^7$ cells mL^{-1} corresponding to an algae productivity of 83 mg L^{-1} d^{-1} for O.D. and a flux of photons equal to 0.5 and 250 µmolE m^{-2} s^{-1}, respectively. The similar microalgae productivity whatever the N:P ratio was mainly due to an adaptation of the N:P ratio of microalgae from 11 to 30 gN gP^{-1} according to the N:P medium as observed by means of analysis of N and P contents of microalgae at the end of the experiments. It was thus possible to highlight phosphorus storage by microalgae as reported by [6]. The P-storage accentuates the robustness of such a process against the phosphorus variability of the input.
The influence of the level of CO_2 addition was low compared to O.D. and light influence. In fact, microalgae productivity varied from 100 to 160 mg L^{-1} d^{-1} for O.D. and a flux of photon equal to 0.2 and 250 µmolE m^{-2} s^{-1} and a 24h light regime. The total inorganic carbon concentration varied

between 40 at the beginning to 10-25 mg C L^{-1} at the end. However, the trial without CO_2 injection led to productivity close to 0 and a total inorganic carbon concentration decreasing to 0 mg C L^{-1}. The trials carried out in semi-continuous mode showed a positive impact of the SRT on the microalgae productivity with an increase of 400% of this productivity when the SRT increase from 2.5 to 4.5 days. The low productivity obtained with the low SRT is probably due to the washing out of biomass.

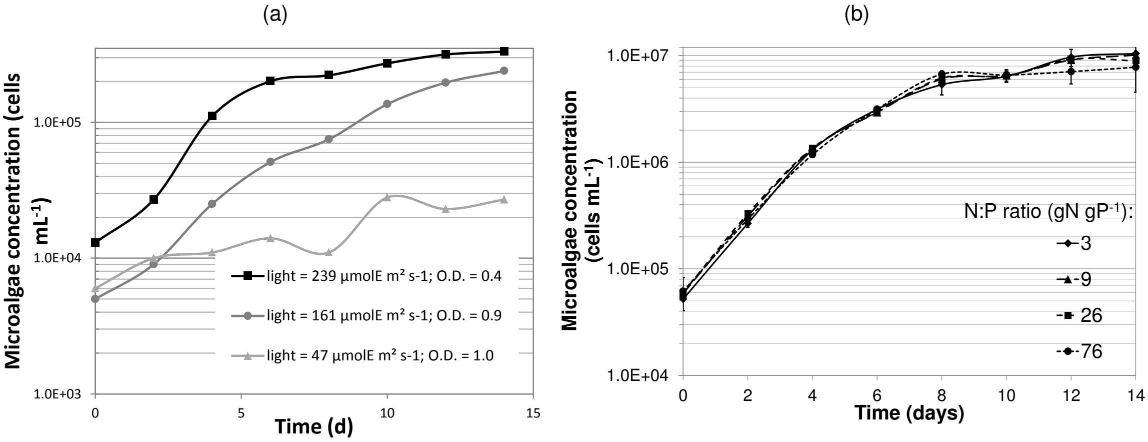

Figure 2: Impact of O.D. and light (a) and N/P ratio (b) on the evolution of microalgae growth over time.

3.2 Nitrogen removal processes

Additionally, the trials carried out in semi-continuous mode (T5) highlighted that nitrogen removal is not correlated with nitrogen assimilated within microalgae biomass. Such a result indicates the involvement of others processes than microalgae growth in N removal. As NH_3 stripping is considered negligible due to low pH and low gas transfer, nitrification-denitrification appears as the main pathway involved in N removal. These processes, e.g. nitrification-denitrification, were more important at low SRT. In fact, for SRT equal to 4.5 days, N removal due to assimilation was equivalent to N removal through nitrification-denitrification whereas N removal through nitrification-denitrification was 30 times more important for SRT of 2.5 d. These results and also results from the batch experiments showed that nitrification level decreases when microalgae growth increases, maybe due to a competition for inorganic carbon between algae and bacteria, as proposed by Choi *et al* (2010) [7]. Moreover, results from T3 highlighted the influence of initial phosphorus concentration on nitrification as illustrated on Figure 3. This impact has also been proposed by other authors [8]. In fact, microalgae have the ability to store internal cell phosphorus and consequently to deplete the medium when concentration of P is low, thus preventing nitrifiers growth while ensuring their needs.

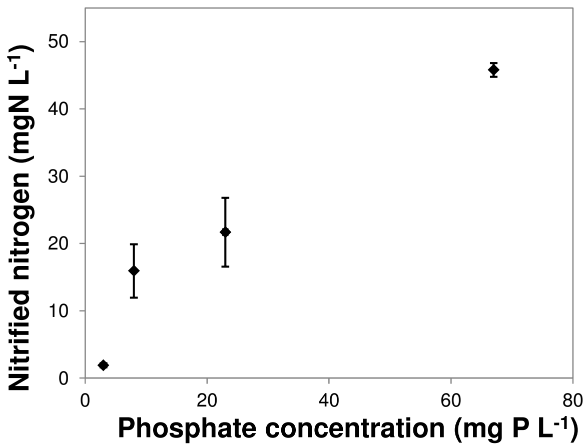

Figure 3: Impact of initial phosphate concentration on nitrification (represented by NOx).

3.3 Modelling and process design

Based on these data, a specific model taking into account the influence of light and O.D was developed using Contois and Droop model. Considering a flux of photons of 930 µmolE m^{-2} s^{-1} (annual average sun light in Rennes, France), a temperature of 25 °C, a O.D of the influent equal to

0.2, no CO_2 limitation and a NH_4^+ removal of 90% ensured by microalgae assimilation (50%) and nitrification-denitrification (50%), this model was used to design a raceway pond treating digestate from a commercial farm producing approximatively 4000 pigs per year. For a depth of 30 cm, the required surface for the raceway pond was about 0.8 ha. The calculated maximum microalgae concentration within the pond was between 1.7 and $2.3x10^5$ cells L^{-1} and associated nitrogen removal was estimated to 11 mgN L^{-1} d^{-1} for a SRT between 4-6 days. Because of the high concentration of NH_4^+ in digestate (up to 3-4 gN L^{-1}), the hydraulic retention time (HRT) necessary is higher than SRT requiring a recirculation of the effluent to the pond after microalgae separation. Such separation/recirculation was estimated between 375-500 m^3 d^{-1} for the modeled case.

4. Conclusion and outlook

From laboratory scale results, a model was developed and used to simulate a raceway pond at real-scale for nutrient extraction from digestate using microalgae culture. As an example, a surface of 0.8 ha and a recirculation from microalgae separators back to the reactor equal to 50 times the influent flow rate was determined for the treatment of digestate from a commercial farm producing about 4000 pigs per year. The results obtained highlighted the importance of phase separation for the development of such a process. In fact, from one hand, a very good separation of the digestate must be achieved in order to minimize the turbidity of the influent. On the other hand, important volume must be separated and recirculated after the pond. Because of the nitrification-denitrification was identified as major N removal process, a special attention should be paid to nitrous oxide emissions.

Acknowledgements

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References

[1] Molinuevo-Salces, B.; Garcìa-Gonzàlez, M. C.; Gonzàlez-Fernàndez C. (2010). "Performance comparison of two photobioreactors configurations (open and closed to the atmosphere) treating anaerobically degraded swine slurry." Bioresource Technology 101(14): 5144-5149.

[2] de Godos, I.; Blanco, S.; García-Encina, P. A.; Becares, E.; Muñoz, R. (2009). "Long-term operation of high rate algal ponds for the bioremediation of piggery wastewaters at high loading rates." Bioresource Technology 100(19): 4332-4339.

[3] Groeneweg, J. (1980). "Treatment of wastewater from live-stock production units with high-rate-algal-ponds." Inst. Biotechnol. No. 4: 165-170.

[4] Kotai, J. (1972). "Instructions for preparation of modified nutrient solution Z8 for algae." NIVA B-11(69).

[5] Staub, R. (1961). "Ernährungsphysiologish-autökologische Untersuchungen an Oscillatoria tubescens D.C." Schweiz. Z. Hydrol. 23: 82-198.

[6] Powell, N.; Shilton, A. N.; Pratt S.; Chisti Y. (2008). "Factors influencing luxury uptake of phosphorus by microalgae in waste stabilization ponds." Environmental Science and Technology 42(16): 5958-5962.

[7] Choi, O.; Das, A.; Yu, C.P. ; Hu, Z. (2010) "Nitrifying bacterial growth inhibition in the presence of algae and cyanobacteria." Biotechnol. Bioeng., 107 (6), pp. 1004–1011

[8] Risgaard-Petersen, N.; Nicolaisen, M.H.; Revsbech, N.P.; Lomstein, B.A. (2004). "Competition between ammonia-oxidizing bacteria and benthic microalgae." Appl. Environ. Microbiol., 70 (9), pp. 5528–5537

TD-O_08 Ecological evaluation of biogas from catch crops with Sustainable Process Index (SPI)

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1. Objectives

In many places agricultural energy production from biomass can result in competing situations between food and energy. Furthermore a one-sided use of land such as pure monocultures can lead to heavy pressures on soil and environment and as a consequence endanger ongoing agricultural cropping. Future challenges in bio resource management will have to

- sustain intact arable land and food production [1],
- without losing economic feasibility
- and to further develop farming processes so that they can bring increased economic and ecological benefits [2].

Hence agricultural future for energy, material and food production is connected to each other due to basic physical, ecological and economic means. A confrontation with this challenge and an interaction are needed to find pathways out of dependency dilemmas. Alternative options must be forced by all stakeholders involved in farming processes.
Extending biomass as resources for energy, agricultural land does not necessarily have to be exploited horizontally. Additional cropping potential could also be developed when some farming techniques and sequences are changed [3]. One example which could play a possible role in this debate is shown in this work where cover crops are sown in field experiments during the projects Syn-Energy I [4] and II [5] between regular cropping periods. No further land is needed but plants proposed for energy use were planted in intermediate periods between the periods main crops grow. These cover crops were used to produce biogas having different energy provision utilisations. An ecological evaluation was made to evaluate the overall process including cultivation of main and cover crops along with biogas production and its utilisation.

2. Methodology

Available ecological evaluation methods are manifold and can show different aspects of ecological pressure, impact and cost [6]. The scenarios were evaluated regarding the environmental compatibility with the Sustainable Process Index (SPI) [7]. The SPI is defined accordingly to two principles of generally accepted demands human kind needs to fulfil to embed man-made activities sustainably [8]:

1. Human activities must not alter long term storage compartments of global material cycles in quality as well as in quantity. If this principle is not adhered to resources will be depleted and substances accumulated in ecosphere, overstraining the natural cycles.

2. Flows to local ecosphere have to be kept within the qualitative and quantitative range of natural variations in environmental compartments. If such flows exceed the amount a compartment can integrate the accumulating substances will alter the compartment. This alteration can lead to a local environment that is no longer able to sustain flora and fauna.

This number can be taken to identify the ecological pressure of human activities. The larger this number is the higher the environmental impact. Detailed mathematical calculation implementing these assumptions is explained by Narodoslawsky and Krotscheck, 1995 [9]. Final result is an aggregated number accounted in a total ecological footprint in m^2 SPI.
The practical evaluation was carried out with the freely available online tool SPIonWeb [10]. This tool consists of a graphical user interface and a database including typical life cycle data to create and interlink man-made process cycles [11].

With Syn-Energy I and II field experiments during the years 2009 and 2015 data about yields, emissions and erosion rates was collected. These field experiments were executed in different scenarios of soil cultivation, cultivation techniques and kinds of field crops. In field experiments two crops which are typically grown in Austria were sown as main crops and these were wheat *(triticum)* with an intermediate yield of 6 t/ha with 88 % DM (dry matter) content and whole crop silage of maize *(zea mays)* with an intermediate yield of 50 t/ha with 30 % DM content were sown in succession with fallow land or cover crops. Cover crops were used in different amounts and compositions including seeds for egyptian clover *(trifolium alexandrinum),* crimson clover *(trifolium inkarnatum),* grain pea *(pisum arvense/sativum),* common vetch *(vicia sativa),* sorghum *(sorghum),* common sunflower *(helianthus annuus),* phacelia *(phacelia),* mung bean *(vigna radiate),* persian clover *(trifolium resupinatum),* white mustard *(sinapis alba),* oil radish (*rhaphanus sativus var. oleiformis),* lopsided oat *(avena strigosa),* triticum *(triticale),* buckwheat *(fagopyrum),* broad bean *(vicia faba),* sweet pea *(lathyrus saltivus)*. These crops are typically suggested and already many of them tested to be grown as cover crops [12]. Some examples also show that when using specific cover crops in fermentation for biogas, methane yields of 800 m³/ha on average can be achieved [13].
Summer cover crops were sown in succession with wheat and winter cover crops in succession with maize. These combinations were chosen because of differences in growing and nutrition demand of the main crops. Regarding the use of machinery different cultivation techniques like ploughing, gleaning, grubber, rotary harrow and direct seeding were applied. Table 9 shows the sequences from when to when soil was cultivated for main crops with or without cover crops, which kind of plant was planted and which technology was used.

Table 9: Dates of soil cultivation, seeding and harvest of summer cover crop and winter wheat for the calculation of erosion with BoBB [14].

Date	without s-cc	s-cc mulched	s-cc harvested, ploughing for main crop	s-cc harvested no ploughing
21.07.	gleaning	gleaning	grubber	grubber
22.07.	gleaning	gleaning	seeding s-cc	seeding s-cc
29.07.	gleaning	grubber	s-cc	s-cc
30.07.	gleaning	seeding s-cc	s-cc	s-cc
16.9.	ploughing	s-cc	s-cc	s-cc
15.10.	fallow	s-cc	harvest s-cc	harvest s-cc
16.10.	rotary harrow	ploughing	ploughing	grubber
17.10.	fallow	rotary harrow	rotary harrow	fallow
18.10.	seeding winter wheat	seeding winter wheat	seeding winter wheat	seeding winter wheat
20.07.	harvest winter wheat	harvest winter wheat	harvest winter wheat	harvest winter wheat

ha: hectare; s-cc: summer cover crops

Table 10 shows the sequences from when to when soil was cultivated for main crops with or without cover crops, which kind of plant was planted and which technology was used.

Table 10: Dates of soil cultivation, seeding and harvest of winter cover crop and maize for the calculation of erosion with BoBB [15].

Date	without w-cc	w-cc mulched	w-cc harvested, ploughing	w-cc harvested, no ploughing	w-cc harvested, ploughing	w-cc harvested, no ploughing
04.10.	gleaning	gleaning	ploughing	gleaning	ploughing	gleaning
05.10.	gleaning	ploughing	rotary harrow	grubber	rotary harrow	grubber
06.10.	gleaning	fallow	seeding w-cc	seeding w-cc	seeding w-cc	seeding w-cc
9.10.	gleaning	seeding w-cc	w-cc	w-cc	w-cc	w-cc
10.10.	ploughing	w-cc	w-cc	w-cc	w-cc	w-cc
10.04.	fallow	grubber	w-cc	w-cc	w-cc	w-cc
15.04.	rotary harrow	fallow	w-cc	w-cc	w-cc	w-cc
24.04.	fallow	fallow	harvest w-cc	harvest w-cc	w-cc	w-cc
25.04.	fallow	fallow	grubber	grubber	w-cc	w-cc
26.04.	seeding maize	seeding maize	seeding maize	seeding maize	w-cc	w-cc
01.05.	maize	maize	maize	maize	harvest w-cc	harvest w-cc
02.05.	maize	maize	maize	maize	grubber	grubber
03.05.	maize	maize	maize	maize	seeding maize	seeding maize
30.09.	harvest maize	harvest maize	harvest maize	harvest maize	harvest maize	harvest maize

ha: hectare; w-cc: winter cover crops

For winter and summer cover crops the yields varied from 2.5 t DM/ha up to 6 t DM/ha dependent on specific side parameters as there are cover crops seed mix, climate, soil, cultivation techniques and local weather conditions.

The data collected during the field experiments about yields, fertiliser demand, humus, soil quality and emission sources and sinks was entered in different scenarios created with the online tool SPIonWeb. Scenarios were made for the main crops wheat and maize. As a reference scenario a typical BAU (business as usual) case with fallow land between typically cultivated main crop periods was taken. Two further scenarios were made where on the same field main crops were successed with cover crops. One of these scenarios should show how much the ecological pressure changes, when natural gas is substituted with biomethane produced from these cover crops. The other one should show differences when these cover crops are just left on the field for mulching without using them for another purpose. To get a better understanding for what happens when using main crops or residues from main crops for biomethane additionally two variations of the BAU scenario were evaluated. An assumption of AMON et al.(2006) who indicates that a maximum of 20% of arable land should be taken for energy production was taken as a limiting factor for one variation [16]. In this relation the project team decided to make an own assumption determining that this dedicated part of the land could then be used for biomethane production substituting an equivalent amount of natural gas which would have been otherwise produced differently to supply the ongoing demands for energy. The reason behind this idea was that if in the scenarios without energy production from crops a comparison with scenarios where biomethane is produced from crops would not be fair at all.

3. Results and discussion

3.1 SPI footprint of biogas from cover crops

During the projects Syn-Energy I and II possible contributions to tackle these issues were tested. On different test areas throughout Austria different mixtures of cover crops were sown in the time gaps between typical growing periods (fallow) of two main crops maize and wheat. Beneficial effects for soil, water, erosion and weed management could be measured. Further processing of cover crops in bio fermentation processes and the cycled use of digestate as fertilizer returned to the field reduces amounts of conventionally used external fertiliser.

The ecological evaluation of Syn-Energy II uses the following assumptions that are derived from project results as well as experiences from other projects carried out by the authors (Detailed deduction of these assumptions is outside of this cope of this paper and can be taken from the homepage of the Climate and Energy Fund of the Austrian government [17].)

- Winter wheat with summer cover crops and maize with winter cover crops, each scenario considered with two kinds of soil cultivation and harvesting methods, having yields of main crops (winterwheat: 5,3 t DM (dry matter); maize: 15 t DM) and cover crops (winter: 3 t DM; summer: 4 t DM)
- Biogas manure of winter-cover crop is used by 30 % as fertiliser for the following main crop, whereas for summer-intercrop 80 % are used for the main crop.
- It is assumed that summer cover crops with a minimum share of 50 % legumes and 2 tonnes of legume dry matter yield per hectare have a fixation performance of 70 kg N/ha, winter cover crops (e.g. forage rye with *trifolium incarnatum*) fix 20 kg N/ha.
- A reduction in the use of mineral nitrogen fertiliser can so be reached due to a N-fixation of the legumes and a reduction of wash-out and emissions
- Consequent intercropping reduces weed burden whereby the use of herbicides is reduced by 20 to 50 %.

Figure 7 shows assumed natural cycles with important emission and interactions in the soil water air system (brown part, left side). The directly connected and influencing system (green part, right side) gives an overview about the men-made agricultural process options considered. Besides business as usual cropping of main crops for nutrition use only, in this case study main crops are still untouched but supplemented with cover crops. Cover crops can then be processed to biogas processes (fermenter, combined heat and power (CHP), biogas cleaning, tractor powered by fuel produced from biogas) differently used in the ecological footprinting scenarios afterwards.

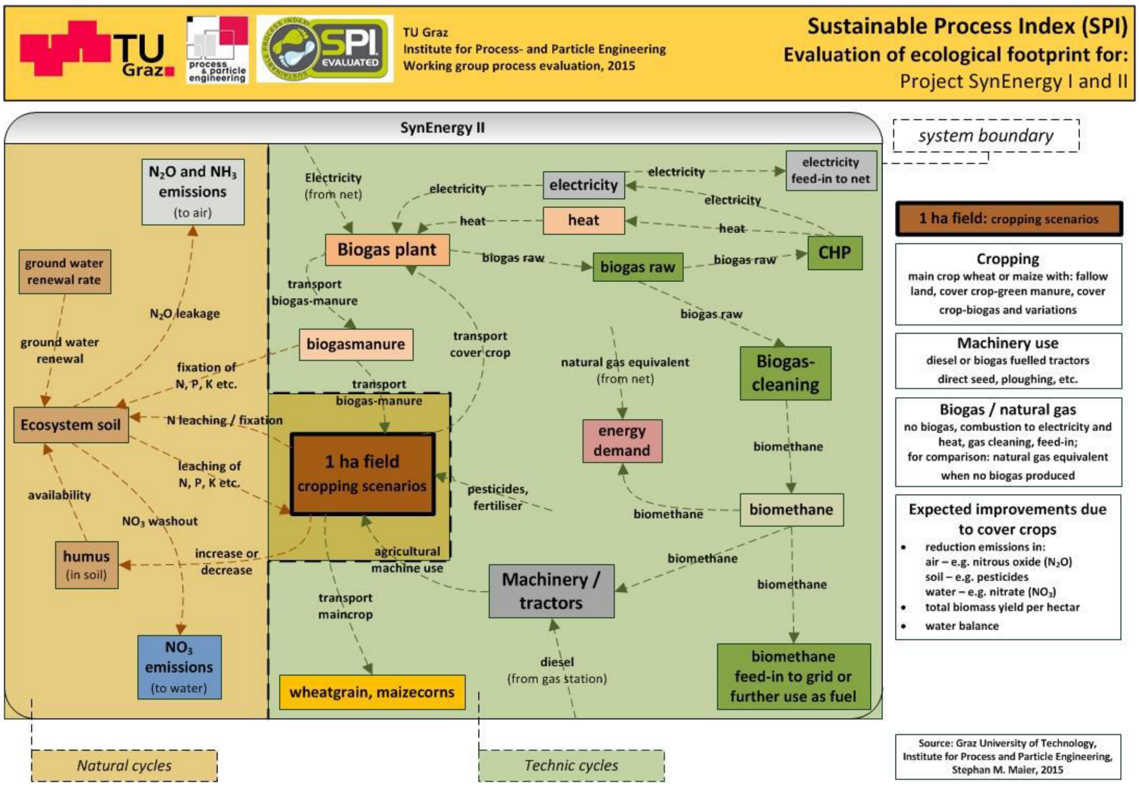


Figure 7: Maximum cultivation, emission cycle and energy network of considered scenarios.

The total agricultural production process cycles on the test areas from soil cultivation and seeding to harvest of the main crops and cover crops were evaluated with Sustainable Process Index (SPI) which already has been successfully tested in different fields of application [18]. Measured data by project partners who did research about cover crops on the testing areas including biogas potential, changes in humus system, erosion, N_2O (nitrous oxide), NH_3 (ammonia) emissions and NO_3 (nitrate) leachate was ecologically evaluated in SPI [19].

3.2 Side parameters and scenarios

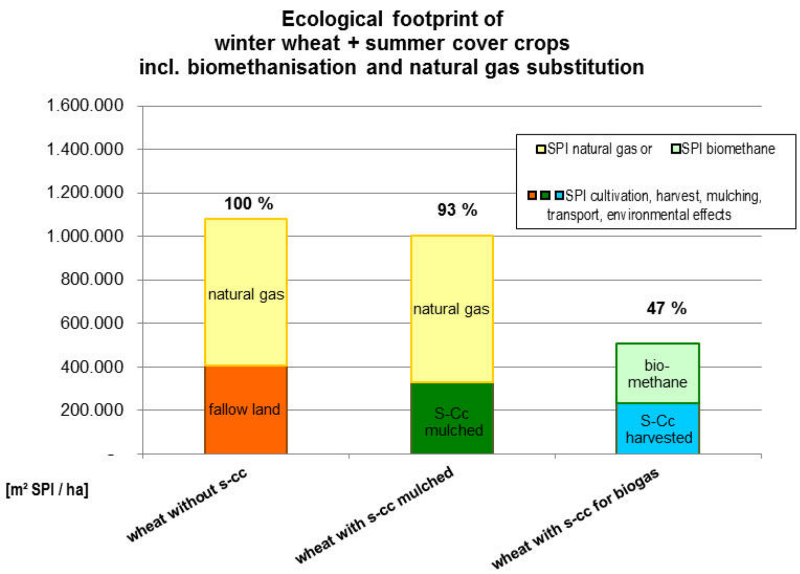
It has been assumed that there are three main types of soils, a heavy (mainly consisting of clay, many fine particles, compact), medium (compound of clay, humus, sand and clastic sediments) and light (mainly sand, inconsistent) soil. Fuel consumption as well as nitrate leachate is dependent on the type of soil available for cultivation. In the current study an average cover crop yield of 4.5 t DM (dry mass) has been chosen, while in case of green manure a cover crop yield of 2.5 t DM has been used and the cover crops were directly worked into the ground as mulch to increase soil fertility. In case of BAU (business as usual) scenario there is a fallow period and cover crop production for green manure production. Similarly overall fuel consumption for each scenario has been calculated for cultivation in medium soil type. The use of heavy duty tractors (70 kW to 110 kW) and machinery has been integrated in all processes.
The evaluated scenarios are described for the wheat scenarios (System I) as follows:

- Conventional (BAU): wheat followed by fallow land; natural gas equivalent; 1,260 m³ natural gas equivalent,
- Main crop wheat in succession with summer cover crops mulched as green manure for fertilisation; 1,260 m³ natural gas equivalent,
- Main crop wheat in succession with summer cover crops harvested for production of 1260 m³ biomethane; biogas manure returned to field as fertiliser; direct seeding without ploughing; self-loading wagon; tractors fuelled with biomethane.

The evaluated scenarios are described for the maize scenarios (System II) as follows:

- Conventional (BAU): 15 t DM maize per ha followed by fallow land; 1,260 m³ natural gas equivalent,
- Main crop maize in succession with winter cover crops mulched as green manure for fertilisation; 1,260 m³ natural gas equivalent,
- Main crop maize in succession with winter cover crops harvested for production of 1,260 m³ biomethane and biogas manure returned to field as fertiliser; ploughing, chopper; tractors fuelled with diesel; biomethane,
- Conventional (BAU) variation 1: maize followed by fallow land; 20 % of arable land for 1,260 m³ biomethane production; 80 % of arable land for food or fodder,
- Conventional (BAU) variation 2: maize followed by fallow land, maize straw for 1,260 m³ biomethane.

The biogas production as well as the specific process steps and the specific ecological footprint in m² SPI are shown in the following two figures Figure 8 and Figure 9.
The description of the scenario results of system I can be seen in Figure 8. In system I wheat was set as a main crop alternated with summer cover crop for biogas production. The ecological footprint was calculated for 1 ha agricultural land containing medium emission values of all three classes of soil (heavy, medium and light soil). Additional use of cover crops in fallow periods opens additional potential to produce biomass and hence energy regionally. This option can reduce energy dependencies on fossil fuels as well as the ecological footprint. Using cover crops for mulching instead of biogas production can reduce the ecological footprint by 7 % compared to the conventional process. Cover crops for biogas production can reduce ecological pressure by up to 53 % compared to conventional processes. When comparing just the ratio of the ecological footprint of cultivation, harvest, mulching, transport and environmental effects without natural gas and biomethane the footprint can go down by 19 % for mulched cover crops up to 42 % for harvested, fermented and then mulched cover crops. When natural gas is not considered in the scenarios without cover crops and mulched cover crops the footprint of harvested and fermented cover crops would be 20 % to 35 % higher.

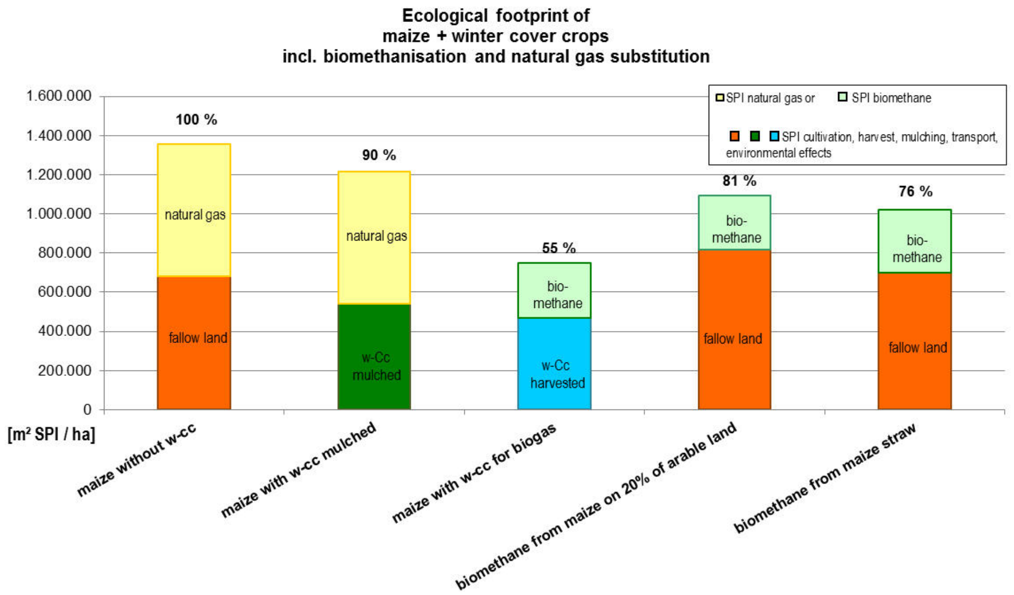


ha: hectare; s-cc: summer cover crops; [SPI m² / ha]: result of Sustainable Process Index in m²SPI per hectare

Figure 8: System I: SPI Scenarios – wheat as main crop and summer cover crops for biogas production per ha cultivation area

Figure 9 shows information about the scenario results of system II. Using cover crops for mulching instead of biogas production can reduce the ecological footprint by 10 % compared to the conventional one. Ecological pressure due to maize cropping can be reduced by up to 45 % with cover crops to biogas scenarios compared to the conventional one. When comparing just the ratio of ecological footprint of cultivation, harvest, mulching, transport and environmental effects without

natural gas and biomethane the footprint can go down by 20 % for mulched cover crops up to 31 % for harvested, fermented and then mulched cover crops.
Also in the additional fallow land scenarios the total ecological footprint could be reduced by 19 % using 20 % of the main crop maize for biomethane up to 24 % using maize straw compared to the conventional BAU scenario. Providing biomass for the fermentation process increases the ecological pressure on field where it grows by 20 % when 20 % of the main crop is used for biomethane production and it grows slightly by 3 % when the field residue maize straw is used for biomethane production. When natural gas is not considered in the scenarios without cover crops and mulched cover crops the footprint of harvested and fermented cover crops would be 10 % to 28 % higher.



ha: hectare; w-cc: summer cover crops; [SPI m² / ha]: result of Sustainable Process Index in m²SPI per hectare

Figure 9: System II: SPI Scenarios – maize as main crop and winter cover crops for biogas production per ha cultivation area.

In both systems I and II the comparison of ecologically evaluated conventional with biogas driven processes shows a potential to reduce ecological pressure between 2 % and 53 % depending on how much fossil fuels are replaced in the individual process steps. This scenario configuration shows that due to the high ecological footprint of fossil fuels the potential to reduce the footprint by the application of green manure is lower than that of the biogasification of cover crops. Using cover crops directly after growing for mulching on field can help to draw near to natural nitrogen cycles and improve soil quality.

4. Conclusion and outlook

Substitution of fossil fuels with biogas from biomass from field without using the main crop for energy purposes can have several benefits. Producing energy from cover crops means no additional competition for land use. This can be an opportunity to better guarantee food security and energy can be provided from biomass on the same area where food and fodder is grown. These results will be discussed with relevant stakeholders of the agricultural sector.
In scenarios with cover crops seeding it was possible to reduce the amount of additional nitrogen fertilisers. Differences in cultivation techniques showed that erosion, humus, nitrous oxide emissions and nitrate leachate resulted in different ecological footprints.

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References

[1] Sage, C. (2012): Environment and Food, Routledge, Abingdon/Oxon, pp 1-320, ISBN: 978-0-415-36311-2 (hbk).

[2] Narodoslawsky, M.; Niederl, A.; Halasz, L. (2008): Utilising renewable resources economically: new challenges and chances for process development, Journal of Cleaner Production, 16/2, pp 164-170, doi: 10.1016/j.jclepro.2006.08.023, (http://www.aidic.it/CISAP4/webpapers/85Halasz.pdf).

[3] Gatta, G.; Gagliardi, A.; Soldo, P.; Monteleone, M. (2013): Grasses and legumes in mixture: an energy intercropping system intended for anaerobic digestion, Italian Journal of Agronomy; volume 8:e7, pp 47-57, doi:10.4081/ija.2013.e7, (http://www.agronomy.it/index.php/agro/article/view/ija.2013.e7/485).

[4] Klimafonds (2011): Klima und Wasserschutz durch synergetische Biomassentuzung – Biogas aus Zwischenfrüchten, Rest- und Abfallstoffen ohne Verschärfung der Flächenkonkurrenz, Blue Globe Report, Erneuerbare Energien #10/2011, https://www.klimafonds.gv.at/assets/Uploads/Blue-Globe-Reports/Erneuerbare-Energien/2008-2011/BGR0102011EEneueEnergien2020.pdf, Download 10 / 08 / 2015.

[5] Klimafonds (2012): Biogas aus Zwischenfrüchten: Ein Beitrag zum Klimaschutz und zur Erhöhung der Wertschöpfung bei gleichzeitiger Sicherung der Lebensmittelversorgung, https://www.klimafonds.gv.at/assets/Uploads/2SynEnergyOekocluster.pdf and further accessible documentation, Download 10 / 08 / 2015.

[6] Čuček, L.; Klemeš J. J.; Kravanja, Z. (2015): Chapter 5 - Overview of environmental footprints, In Assessing and Measuring Environmental Impact and Sustainability, Butterworth-Heinemann, Oxford, pp 131-193, ISBN 978-0-12-799968-5, doi: http://dx.doi.org/10.1016/B978-0-12-799968-5.00005-1. (http://www.sciencedirect.com/science/article/pii/B9780127999685000051).

[7] Narodoslawsky, M. (2015): Sustainable process index, Assessing and Measuring Environmental Impact and Sustainability, Chapter 3, edited by Jiří Jaromír Klemeš, Butterworth-Heinemann, Oxford, pp 73-86, ISBN 9780127999685, doi: http://dx.doi.org/10.1016/B978-0-12-799968-5.00003-8. (http://www.sciencedirect.com/science/article/pii/B9780127999685000038)

[8] SUSTAIN (1994) Forschungs- und Entwicklungsbedarf für den Übergang zu einer nachhaltigen Wirtschaftsweise in Österreich, Verein zur Koordination von Forschung über Nachhaltigkeit, Endbericht der Wissenschaftlergruppe "Sustain", Inst. f. Verfahrenstechnik, Technische Universität Graz, OCLC-Nummer: 732633651, 1994, pp 1-155. (http://www.worldcat.org/title/forschungs-und-entwicklungsbedarf-fur-den-ubergang-zu-einer-nachhaltigen-wirtschaftsweise-in-osterreich-endbericht-des-interdisziplinaren-forschungsprojektes/oclc/732633651).

[9] Narodoslawsky, M.; Krotscheck, C. (1995): The sustainable process index (SPI): evaluating processes according to environmental compatibility, *Journal of Hazardous Materials*, Vol. 41, Issues 2–3, pp 383-397, ISSN 0304-3894, http://dx.doi.org/10.1016/0304-3894(94)00114-V. (http://www.sciencedirect.com/science/article/pii/030438949400114V).

[10] SPIonWeb (Sustainable Process Index on Web), http://spionweb.tugraz.at, Graz University of Technology 2013 – 2015, last accessed on 21/04/2015.

[11] Kettl, K.-H. (2012): Evaluation of energy technology systems based on renewable resources, Dissertation, Institute for Process and Particle Engineering, Graz, Austria, pp 1-186.

[12] Peu, P.; Picard, S.; Girault, R.; Labreuche, J.; Béline, F.; Dabert, P. (2013): Catch crops for agricultural biogas production, case study for *Brassicaceae sp.*, proceedings 2013, pp 44-51, doi:10.1016/j.biombioe.2015.04.029, (http://www.ramiran.net/doc13/Proceeding_2013/documents/S7.02..pdf).

[13] Molinuevo-Salces, B.; Larsen, S.U.; Ahring, B.K.; Uellendahl, H. (2013): Biogas production from catch crops: Evaluation of biomass yield and methane potential of catch crops in organic crop rotations, Biomass and Bioenergy, Volume 59, pp 285-292, ISSN 0961-9534, http://dx.doi.org/10.1016/j.biombioe.2013.10.008. (http://www.sciencedirect.com/science/article/pii/S0961953413004303)

[14] BoBB, Bodenerosion, Beratung, Berechnung (soil erosion, consulting, calculation) download available on http://baw.at/index.php/ikt-download/1679-erosionsberechnungstool-bobb.html. BoBB is based on a slightly changed version of Revised Universal Soil Loss Equation (RUSLE), last accessed on 21/04/2015.

[15] BoBB, Bodenerosion, Beratung, Berechnung (soil erosion, consulting, calculation) download available on http://baw.at/index.php/ikt-download/1679-erosionsberechnungstool-bobb.html. BoBB is based on a slightly changed version of Revised Universal Soil Loss Equation (RUSLE), last accessed on 21/04/2015.

[16] Amon, T.; Kryvoruchko, V.; Hopfner-Sixt, K.; Amon, B.; Bodiroza, V.; Ramusch, M.; Hrbek, R.; Friedel, J. K.; Zollitsch, W.; Boxberger, J. (2006): Biogaserzeugung und Potentiale, Ländlicher Raum, Online-Fachzeitschrift des Bundesministeriums für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, Teil 1: gem. Werkvertrag Nr. 439; Wien 27.02.2006, pp 3-4.

[17] Climate and Energy Fund of the Austrian government, https://www.klimafonds.gv.at/foerderungen/foerderlandkarte/detail/?projectID=46143, last accessed on 20/08/2015.

[18] Heckl, I.; Halász, L.; Szlama, A.; Cabezas, H.; Friedler, F. (2015): Process synthesis involving multi-period operations by the P-graph framework, Computers & Chemical Engineering, Available online 9 May 2015, ISSN 0098-1354, http://dx.doi.org/10.1016/j.compchemeng.2015.04.037. (http://www.sciencedirect.com/science/article/pii/S0098135415001386)

[19] Narodoslawsky, M.; Niederl, A. (2006): Sustainable Process Index, Renewable-Based Technology: Sustainability Assessment, Chapter 10, Ed: Dewulf, J.; van Langhove, H., John Wiley & Sons, 2006, pp 159-172, doi: 10.1002/0470022442.ch10. (http://onlinelibrary.wiley.com/doi/10.1002/0470022442.ch10/summary).

TD-O_09 Needs for production of grain protein crops in Europe and possibilities for more bio-based green economy including nitrogen use

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1. Objectives

Nowadays the EU has a 70% deficit in protein rich grains, which are imported like soybean and soybean meal. Both are used as a feed for increased meat production with amplified imports of soya beans that increased from 1961 to 2013 from 2.7 to 37 M t (15% of world soybean production), respectively [1, 3]. In Slovenia the existing production of protein feeds cover just 2.8% needs, and share of fields under grain legumes is 0.25%, import of soybean and meal are about 97.5% [1, 3]. The aim of the paper is to introduce the protein feed crops into crop rotation [2], especially soybean and pea.

2. Case study description

A review of protein supply in EU countries, such as export, import and production data for Slovenia are analyzed, including potential importance of N symbiosis on the fields, exchange of functional biodiversity on the fields, inputs of N with organic manures and costs for N fertilizers. In Slovenian case the total balance of own protein feed potential was calculated and the farmers opinion of protein feeds were presented. For soybean production two scenarios (10% and 20% of arable land i.e. 17,500 and 35,000 ha, respectively were took into account.

3. Observations and discussion

3.1 Role of soybean in European Union

EU has a 70% deficit in protein rich grains, solving by imports of soybean and soybean meal [4].
The largest proportion of the EU arable land is under cereals and has remained stable over the past at around 57%. On the other hand there have been the major changes in all the other crops in term of the proportion of land. Protein crops declined from 5.8 M ha in 1961 (4.7% of the arable area) to 1.9 M ha in 2011 (1.8% of arable area) [4].
In this period the combined production of beef, pork and poultry meat has increased from 17 to 43 M t and therefore demand for protein crops increased accordingly. This demand has been met with slightly bigger production (for 1 M t in period from 1961 to 2011). The biggest part was met with amplified imports of soybean that increased from 2.7 to 37 M t. This is almost 15% of world soybean production and those imports could be even higher if two thirds of Europe's cereal harvest would not be used as the animal feed [5].
Estimation of Danube Soya Association is that in EU around 36 M t of soybean is used yearly and 30% of it is GMO-free. GMO-free soyabean demand on EU market is mostly covered by Brazil. This could cause risk of contamination since GMO soybean production is not prohibited in Brazil and there are some environmental concerns connected to its production. They have stated that just around 5% of GMO-free soya demand is covered and there are a lot of opportunities on the market for increasing GMO-free soybean production. This demand is also increasing as the food industry adopts GMO-free policies [6].
The research of Henseler *et al.* revealed that trade disruptions with the biggest importers like Brazil, Argentina and USA would cause 30% increase of the import price. Moreover, the most affected sectors in EU would be pig and poultry industry. Animal production in general would decrease in that case [7].
Another similar study concluded if EU would stop import from three main suppliers (USA, Brazil and Argentina) the price for EU food industry would go up around 220% for soybean, 211% for soybean meal and 202% for soy oil [8].
The major factor for extension of large proportion of arable area in cereal production and deprivation of other crops was good annual yield increase that was about 0.15 t/ha/yr [3]. It has been facilitated by the switch to autumn sowing, the availability of inexpensive nitrogen fertilizers, investment in plant breeding and well established pesticide sector. Combinations of low cost of nitrogen fertilizers and imported feed protein have promoted cereals in the past [9].

On the other hand farmers who grow protein cops are confronted to various agronomic challenges. Yields of protein crops are considered as unstable in many studies. Some studies showed that some protein crops are not as competitive as cereals against weeds and some are not as competitive in drought stress, pest and diseases (citates in [3]).
Different yields after all affect farmer's gross margins. Per ha gross margins are higher for cereals than for protein crops mostly because CAP reforms have reduced the support for protein crops during last decades [4].
The recent developments on the market provide some opposing effects. The first argument is the price of nitrogen fertilizers more than doubled since 2000 meanwhile, wheat price has not followed this trend [4].
Another is that the price of soybean in global market increasing since 2007 and at the same time the import quantities have fallen. The same trend is expected in the future due to growing international demand. The producer price for soybean is increasing accordingly to the price of imported soybean and is reducing the comparative advantage of wheat and other cereals [4].

3.2 Environmental impacts of soybean

Many studies reveal that production of soybean and other protein crops could be beneficial for European agriculture.
Cultivation of protein crops could considerably reduce the use of synthetic nitrogen fertilizer. Compared to cereals, protein crops do not require high N fertilizer inputs to reach the full yield potential, cereals need to receive around 100 to 200 kg N/ha. The residual N combined with other positive effects on soil could be expressed as fertilizer nitrogen equivalent that accounts around 120 kg N/ha [10].
Crops following protein crops in rotation have better yields. For example even if wheat and other cereals are fertilized for optimum yield the yield was 15 to 25% bigger [11]. There is almost no nitrous oxide released and nitrates leached in water in soybean production. Also the biodiversity benefits are known [4].
Protein crops can lead to 20-25% reduction of pesticide costs for the succeeding crop and 7% less ecotoxicity [12].
Also the positive effects on soil have been studied. Protein crops are providing carbon storage, benefits to soil structure and composition, and also better ability for phosphate intake. In EU grown protein crops their production is associated with significantly reduced fossil energy use, greenhouse gas (GHG) emissions, ecotoxicity, eutrophication and acidification [4].

3.3 European policy of protein legumes

In the past the development of Common Agricultural Policy (CAP) direct price support shifted to area based support that was decoupled from production. Those were the main drivers connected with decline in protein crops production. One of the causes was also Blair House Agreement (1992) which restricted supported area for protein crops in EU with agreed with World Trade Organization (WTO) and USA. Later on, in the period from 2002-2004 to 2008, the surface covered with protein crops fell from 1.8% to 0.8% and this appeared to be the response of decoupling [4].
At present protein crops are supported through Article 38 which allows supporting specific crops with up to 10% share in the single payment scheme. At the same time they can use also the complementary national direct payments but for now just 2 countries use it [4].
Currently CAP is in the state of negotiations. Proposals from CAP are to use protein crops as a greening and crop diversification component that could be implemented in ecological focused areas (EFA). EFA could represent from 3 to 15% of arable land in present negotiations and protein crops could be a part of it because of nitrogen fixation properties.
Policy instruments are one of the most important factors influencing the cultivation of protein crops in Europe.

3.4 Example of Slovenian farmer opinion about introduction of grain legumes into crop rotation

This research was done in the region of Savinjska Valley on the base of questionnaire carried out in the frame of national project whose main goal was to define the present situation in Slovenia and recognize the future development objectives for production of protein fodder crops [13, 4]. The questionnaire was distributed among dairy farms, which have more than 5 hectares of cultivated fields and are included in »Produced/manufactured GMO free« milk production. The result of the questionnaire shows, that only 23% of included farms grow grain legumes. The main reason for low production of grain legumes is in the lack of agricultural areas. On the other hand, the result of the questionnaire shows that soybean is used as feed by 47% of those who took part in the questionnaire. The other half of the farmers thinks that growing soybean causes excessive costs. The farmers are also inadequately informed about the growing of soybean and also about its feed value. Only 10% of farmers who took part in the questionnaire is informed how common European policy increase the production of leguminous crops. Only 37% of them predict that the growth of the production of leguminous crops by the year 2020 will increase mostly because of the financial incentive. On the basis of these results it can be concluded that farmers do not believe in the effectiveness of financial incentives. In the year 2015, 2% of payments from national envelope will intend for supporting the production of protein crops. In 2015 the height of financial support per hectare of area occupied with protein crops is 419 EUR.

3.5 Self-sufficiency of protein feed and possibilities for more bio-based green economy including N use in Slovenia

Analysis of Slovenian protein supply chain potential is based on existing production of oil crops (cca. 5 000 ha seed rape (nowadays used only for biodiesel exported to Austria and Italy) and 5000 ha oil pumpkins), which cakes can give (15 000 t with 34% and 4 000 t with 60% crude protein, respectively) in total 7 500 t crude proteins. After consumption this feed represents quality organic manure (in total approximately 145 t N i.e. 478 t CAN (27% N) i.e. 5 560 bags of CAN i.e. 76 728 EUR) that could be used for incorporation in soils, which will reduce use of mineral N fertilizers, increase water infiltration, etc. Even the total balance of N for soybean production is negative, in case of Danube Soya scenario on 17 500 or 35 000 ha fields with grain legumes (Table 1) approx. 14 000 or 28 000 t of symbiotic N could be assimilated, instead of buying 250 550 or 560 000 bags of CAN in the sum of 1 062 or 2 128 M EUR, respectively.

As additional value of symbiotic N used by intercrops (for example 10% of cereals, 4 000 ha x 60 kg N) 2 400 t of N matches 48 000 bags of CAN i.e. 662 400 EUR. However, changed natural system can reduce N mineral inputs in total to 2.13 M EUR and increase functional biodiversity for 1/4 [2]. Imported soybean and soybean meal (mainly from Brazil) into EU is produced on 18 M ha. In case of different scenarios of own production in EU (in total 10, 20, 30%, etc.) similar trends of green bio-based economy and decreasing costs could be achieved at the field production level.

Table 1: Potential N symbiotic fixation for soybean production and intercrops in two scenarios for Slovenia vs. No. of bags of CAN and their costs.

Scenario	Field area	Symbiotic N	Bags of CAN	Saved M EUR for mineral fertilizers
	ha	000 kg/ha	No.	M EUR
1	17 500	1 400	250 550	1.062
2	35 000	2 800	560 000	2.128
Intercrops (pea + cereals)	4 000	240	48 000	0.662

4. Conclusion

Because of imported 70% of protein feeds in EU, the main political decision of good agriculture practice (GAP) is supporting introduction of protein crops into rotation systems.

To conclude, increasing production of grain legumes can be important additional self-sufficiency measure for creating greener bio-based economy, while increased functional biodiversity on the fields, more natural symbioses N instead of using artificial N fertilizers, saving of energy and cost for artificial N fertilizers are expected.

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References

[1] EIP-AGRI focus groups: Grain legumes. Final reports 14.4.2014, 29.10.2014 (http://ec.europa.eu/eip/agriculture)

[2] Bavec, F., Bavec M. (2015): Underutilized crops and intercrops in crop rotation as factors for increasing biodiversity on fields. In: Biodiversity in Ecosystems - Linking Structure and Function. Intech.13 p.

[3] Bavec, S. (2014): Certification of European GMO-free soya to foster its production and use in shorter supply chains and traditional food products. M.Sc. thesis, Groupe ESA, Angers; France

[4] EP - European Parliament (2013): The environmental role of protein crops in the new common agricultural policy study. [On-line]. European Parliament - Directorate general for internal policies. Agriculture and rural development (http://www.europarl.europa.eu/studies)

[5] Von Witzke, H., Noleppa, S. (2010): EU Agricultural production and trade: can more efficiency prevent increasing, land grabbing' outside Europe? [On-line]. Research Report. Humboldt University Berlin, agripol – network for policy advice, Berlin. (http://www.agrar.hu-berlin.de/fakultaet/departments/daoe/ihe/Veroeff/opera-final_report_100505.pdf.)

[6] DS-Danube Soya (2012): Aufgaben/Function. [On-line] Danube Soya. (http://www.donausoja.org/aufgaben-function)

[7] Henseler, M., Piot-Lepetit, I., Ferrari, E., Gonzalez Mellado, A., Banse, M., Grethe, H. Parisi, C., Helaine, S. (2013): On the asynchronous approvals of GM crops: Potential market impacts of a trade disruption of EU soy imports. Food Policy, 41,166–176

[8] Kalaitzandonakes, N., Kaufman, J., Miller, D. (2014): Potential economic impacts of zero thresholds for unapproved GMOs:The EU case. Food Policy, 45, 146–157

[9] Brouwer, F. (2006): Main trends in agriculture. [On-line]. LEI Agricultural Economics Research Institute. (http://www.mtt.fi/sasspo/SASSPO_HKI_BN1-1.pdf)

[10] Kopke U., Nemecek T. (2010): Ecological services of faba bean. Field crops research. 115: 217-233

[11] Peoples, M.B., Brockwell, J., Herridge, D. F., Rochester, I. J., Alves, B. J. R., Urquiaga, S.,Boddey, R. M., Dakora, F. D., Bhattarai, S., Maskey, S. L., Sampet, C., Rerkasem, B., Khan, D. F., Hauggaard-Nielsen, H., Jensen E. S. (2009): The contributions of nitrogen-fixing crop legumes to the productivity of agricultural systems. Symbiosis, 48, 1-17

[12] Von Richthofen, J.-S., Pahl, H., Bouttet, D., Casta, P., Cartrysse, C., Charles, R., Lafarga, A. (2006): What do European farmers think about grain legumes? Grain Legumes, 45, 14–15

[13] Bavec, F. (2011-2014): Alternative field crops under different production systems, crop rotations as a basic principal for adaptation to climate changes and qualitative feed - food supply (V4-1137).

[14] Blagotinšek, M. (2015): Introduction of grain legumes into crop rotation on dairy farms. University of Maribor, Diploma work, 48 p.

TD-O_10 Effect of source segregation and conventional separation of pig excreta on biogas yield of solids

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1. Objectives

Our objectives were:

- to compare source segregation with four conventional separation methods, including screw press, centrifugation, flocculation and combined flocculation with centrifugation, for treatment of pig slurry in terms of separation efficiency for dry matter, organic matter, and nutrients;
- to determine and compare the characteristics of derived solids and liquids;
- to assess the biodegradability and hydrolysis rate constants of solids and liquids by performing anaerobic digestion batch tests; and
- to evaluate the influence of source segregation and conventional separations on volumetric biogas yield of solids.

In order to make valid comparison between separation methods, all tests were performed with manure from the same batch, guaranteeing equal characteristics.

2. Methodology

Fresh source segregated solid (SSS) and liquid (LSS) were collected from the conveyor belt system (Aanink and Ogink, 2011 [1]) of fattening pig farm in Sterksel, The Netherlands. Slurry (S) was created by mixing a fixed ratio of SSS:LSS:water of 0.44:0.56:0.38. The dry matter content of the slurry was 8-9%.

A tomato strainer with a 1.3 mm pore diameter was used as a lab-scale simulation of a screw press separator. Centrifugation of the slurry was carried out for 1 minute at a G-force of 3,500 G using a laboratory centrifuge. For flocculation separation, PAM (Superfloc C-2260, Kemira Kemwater, Findland) at a 40 mol% charge density was applied to the slurry. In order to test flocculating followed by centrigation, after flocculation with PAM C-2260, the container was centrifuged at 3,500 G for 1 minute. Separation efficiencies E(x) for *x* (wet weight, dry matter (DM), organic matter (OM), soluble COD (sCOD), and nutrient content) were compared between methods. The separation efficiency *E(x)* expresses the percentage of the amount of parameter *x* that is accumulated in the solid fraction, and is calculated as:

$$E(x) = \frac{m(x)_{solid}}{m(x)_{solid} + m(x)_{liquid}} \times 100 \quad (\%) \qquad \text{Eq. 1}$$

where $m(x)_{solid}$ and $m(x)_{liquid}$ are mass of parameter *x* (wet weight, DM, OM, TN, TP, N-NH_4, P-PO_4) in the solid fractions and in the liquid fractions, respectively.

The specific biochemical methane yield (BMP) of the solids and liquids was determined in batch test at mesophilic conditions following VDI 9630 standard. The hydrolysis rate constant k_h of the solids were determined according to Sanders, 2001 [2]. In order to compare measured hydrolysis rates for two substrates ($k_{h,1}$ and $k_{h,2}$), the t-test for two independent samples was used.

3. Results and discussion

3.1 Characteristics of solids and liquids

Table 1 summarizes the characteristics of collected materials, derived solids, and liquids from conventional separations from source-segregated solid.

Table 1: Composition and anaerobic degradation characteristics of solids and liquids; average values, n/a = not available.

	Source separation[a]		**Synthesized slurry[a]**	**Conventional separation of synthesized slurry[a]**							
				Solids				*Liquids*			
	SSS	LSS	S	SSP	SC	SF	SFC	LSP	LC	LF	LFC
Composition[b]:											
DM, g/kg	260	17.2	84.9	256	220	190	213	69.0	27.1	17.1	16.4
OM, g/kg	218	8.1	68.0	235	187	162	182	52.5	17.0	9.01	7.98
OM/DM, %	84	47	81	92	85	85	85	76	62	53	49
COD_{tot}, g/kg	323	16.4	105	301	258	221	282	85.7	42.4	n/a	20.2
COD_s, g/kg	50.2	14.2	22.1	18.8	21.8	21.8	21.8	n/a	22.2	n/a	17.4
TN[c], g/kg	10.8	4.54	5.25	7.16	8.46	7.75	9.27	5.03[c]	3.92	3.01	2.89
NH_4-N, g/kg	3.13	3.94	2.41	1.89	2.39	2.43	2.35	2.47[c]	2.98	2.46	2.49
PO_4-P, g/kg	3.93	0.1	1.34	1.56	3.78	2.9	3.52	1.31[c]	0.25	0.12	0.09
TP, g/kg	0.41	0.07	0.24	0.12	0.34	0.26	0.29	0.04[c]	0.12	0.06	0.07
COD/OM ratio	41.6	254	59.1	28.0	38.5	40.7	43.4	72.9[c]	144	176	177
Anaerobic degradation[d]:											
BMP_{39}, mL CH_4/g OM	303	429	298	253	283	274	290	313	531	580	631
BMP_{39}, mL CH_4/g COD	206	207	193	198	205	205	187	191	213	n/a	250
BMP_{119}, mL CH_4/g OM	336	453[c]	352	304	331	328	331	362	590	659	708
BMP_{119}, mL CH_4/g COD	228	218	228 ±8.1	238	240	240	214	220	237	n/a	281
Degradability 39 days[e], %	59	59	55	57	59	59	53	54	61	n/a	71
Degradability 119 days[e], %	65	62	65 ±0.02	68	69	69	61	63	68	n/a	80

[a] SSS and LSS: solid and liquid from source segregation; SSP and LSP: solid and liquid from screw press; SC and LC: solid and liquid from centrifugation; SF and LF: solid and liquid from flocculation; SFC and LFC: solid and liquid from flocculation + centrifugation;
[b] n = 2;
[c] not measured but calculated from the mass balance of the separation experiment;
[d] n = 3; mesophilic conditions (37°C). BMP values are already subtracted for yield of blank;
[e] calculated based on the fact that 350 mL CH_4 equals 1 g of COD at standard conditions.

Table 2: Slurry separation efficiency[a], SE(x) in %, for mass, DM, OM, TN, TP, NH_4-N, and PO_4-P; average values of duplicates with standard deviation between brackets.

	Mass (wet weight), %	**Dry matter, %**	**Organic matter, %**	**TN, %**	**NH_4-N, %**	**TP, %**	**PO_4-P, %**
Source segregation (SSS)	44[b] (2.0)	92 (0.5)	96 (0.4)	66 (0.0)	38 (0.0)	97 (0.0)	84 (0.0)
Screw press (SSP)	10[c] (0.4)	30 (0.8)	34 (0.9)	39 (0.9)	8.2 (0.3)	12 (0.4)	36 (0.9)
Centrifugation (SC)	33[c] (1.7)	80 (1.2)	84 (1.0)	51 (1.9)	28 (1.6)	88 (0.8)	58 (1.9)
Flocculation (SF)	44 (0.5)	90 (0.2)	93 (0.1)	67 (0.4)	44 (0.5)	95 (0.1)	77 (0.3)
Flocculation+Centrifugation (SFC)	36 (0.4)	88 (0.2)	93 (0.1)	64 (0.4)	34 (0.4)	96 (0.1)	70 (0.4)

[a] expressed as the relative amount that is accumulated in the solid fraction;
[b] not measured but derived from data of Aarnink and Ogink (2007) [1];
[c] n = 4.

Regarding the source segregation system, the DM content and OM/DM ratio were 26% and 84% for SSS and 1.7% and 47% for LSS. These values are equal to the solid (25%, 85%) and liquid (3%, 45%) of the similar system in Aarnink and Ogink, 2011 [1]. The LSS was a brownish liquid containing low OM/DM ratio (47%), which meant as out of half of the OM was inorganic compounds.
Regarding the conventional separations, the solid from the screw press (SSP) appeared driest and had mainly large particles whereas solids from centrifugation (SC), flocculation (SF), and flocculation-combined centrifugation (SFC) were wet and pasty, similar to the source segregated solid SSS. The SSP and SSS had highest DM content (26%) of all solids. SSP had highest OM/DM ratio (92%) vs. 84-85% of all other solids. The liquids obtained from all separations showed DM content in the range of 2-3% and OM/DM ratio of 47-62% except for the liquid from screw press (LSP) where it had quite high DM content (6.9%) and high OM/DM ratio (76%).
The nutrient contents (TN, TP, $N\text{-}NH_4$, $o\text{-}PO_4$) in terms of g/kg DM were in the same range for SSS, SC, SF, and SFC but significantly lower for SSP (Table 1). SSP had the highest DM content (26%), but lowest nutrient contents indicating that the big size particles (F >0.2 mm), which were found to be the majority (72% mass distribution of total dry matter) in SSP, are not as rich in nutrient as smaller size particles. Except for the screw press, the TP content in the liquids were very low (5.50-9.21 g/kg DM) compared to the TP content in the solids (15.1-17.2 g/kg DM). $N\text{-}NH_4$ content was higher in SSS (3.1 g/kg wet weight) and LSS (3.94 g/kg wet weight) than in other solids and liquids (1.89-2.89 g/kg). This was because of the dilution effect when creating slurry from LSS and SSS. Similar to TP content, the $o\text{-}PO_4$ content was relatively lower (0.04-0.12 g/kg) in the liquids than in the solids (0.19-0.41 g/kg). Also, in general, the $o\text{-}PO_4$ content in the solids was little in compared to the TP contents.

3.2 Separation efficiency

Table 2 shows separation efficiencies of each method for wet weight, DM, OM, TN, TP, $N\text{-}NH_4$ and $o\text{-}PO_4$. For the screw press E(OM) and E(DM) were only 30±0.8% and 34±0.9%, respectively. These values were 84±1.0% and 80±1.2% for centrifugation. Flocculating with PAM increased E(DM) to 90±0.2% and E(OM) to 93±0.1%. Further centrifugation after flocculation did not improve separation efficiency for DM and OM. For source segregation, E(DM) and E(OM) were 92.2% and 95.5%, respectively, which were even greater than for flocculation.
Separation efficiencies were very high for TP (96-97%), but lower for TN (64-67%). The very high E(TP) in these methods indicates that most TP can be recovered in the solids (i.e. in the particles) whereas maximum two third of the TN can be separated into the solids.

3.3 Anaerobic degradability

In this study, BMP of the solids and slurry S were 304±4.6-352±12.4 ml/g OM. Except for the solid from screw press SSP where the BMP value was 304±4.6 ml/g OM, the solid from source segregation SSS, centrifugation SC, flocculation SF, or combined centrifugation flocculation SFC gave similar methane yields (328±7.1-336±8.9 ml/g OM). There were not significant differences in BMP values of all solids, indicating there is not an effect of separation methods on the degradability of all derived solids. In addition, it means that separation did not affect the BMP values of solids.
Except for the screw press, BMP of the liquids was much higher than that of the solids and varied widely between 418 ml/g OM for source-segregated liquid (LSS) to 631 ml/g OM for the liquid from flocculation combined with centrifugation (LFC) (Table 1). The significant difference between BMP values of liquids and solids resulted from the fact that the major portion of easily degradable soluble organic in slurry S went to the liquids, while the major particulates retained in the solids.
Methane contents in biogas were around 65-69% for solids and 70-80% for liquids. The biodegradability of the solids (Table 1) did not vary much between types of solids. The biodegradability calculated was 61-68% after 119 days.

3.4 Hydrolysis rate constant

The hydrolysis rate constants were measured with high R^2 values (>0.96). The lowest hydrolysis rates were for SSP (0.04 d^{-1}). The other solids showed faster hydrolysis rates compared to SSP, but still with those values, they are considered slow solubilised substrates for digester. The t-test results showed that hydrolysis rate of SSS was statistically significantly higher than SSP, SC, SF and SFC. Probably the loss of some small particles and colloidals into the liquid fractions during the separations did cause overall lower hydrolysis rates of the remaining solids as compared to SSS as the smaller the particles are the faster hydrolysis rates they have (Sanders, 2001).

3.5 Methane production distribution

Results showed that methane production of the solids SSS, SSP, SC, SF and SFC are 94±2.4, 30±0.4, 75±2.9, 87±1.9, and 85±2.4%, respectively, compared to 100% of methane production distribution in slurry S. The methane production distribution in LSS was lowest (6 %) and significantly higher for LF (12 %), LFC (15 %), LC (27 %) and LSP (68 %) (Figure 1).
The volumetric methane yield is the product of its OM content (ton/ton substrate) and its BMP (m^3 per ton OM). It showed that the volumetric methane yield was significantly higher for SSS (73±1.9) and SSP (72±1.1) than for SC (62±2.3), SF (53±1.1), SFC (60±1.7) and S (24±0.8). This suggested that with the same reactor volume, digesting solid from source segregation and screw press produces a higher volume of methane (i.e., higher gas yield/m^3 reactor/day) than solids from centrifugation, flocculation or flocculation combined with centrifugation.

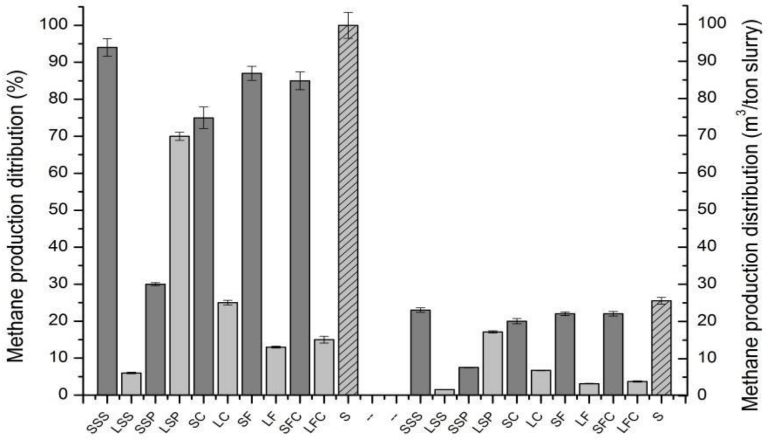

Figure 1: Methane production distributions in the solids and liquids expressed in % of total methane production distribution in S (100%) and in term of m^3/ton original slurry.

4. Conclusion and outlook

Separation efficiencies (E(OM)) of flocculation and flocculation combined centrifugation are similar to source segregation. The specific methane yields of the solids in term of ml/gOM did not vary much but SSS showed fastest hydrolysis rate, requiring less retention time in continuous reactor. The volumetric methane yield of SSS and SSP are much higher than other solids, however E(OM) of screw press was very low (34.3%).

Acknowledgements

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References

[1] A.J.A. Aarnink and N.W.M. Ogink. (2007): Environmental Impact of Daily Removal of Pig Manure with a Conveyer Belt System, International Symposium on Air Quality and Waste Management for Agriculture, 16-19 September 2007

[2] I. Angelidaki & W. Sanders. Assessment of the anaerobic biodegradability of macro pollutants. Environmental Science and Bio/Technology 3: 117–129, 2004

TD-O_11 Biogas production from large-scale retail trade wastes – Preliminary assessments related to Piedmont reality

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1. Objectives

Approximately 1000 anaerobic digestion (AD) plants are currently operational in Italy. 80% of them are still fed with mixtures of manure in combination with energy crops. Nonetheless, the environmental sustainability of this technology relies on the partial or total substitution of dedicated crops with by-products or co-products in the feeding mixture [1], [3]. The objective of the present research was to assess the regional availability of organic waste from the large-scale retail trade as possible feedstock for AD plants.

2. Methodology

The number of existing hyper and supermarkets in Piedmont Region (north-western Italy) were acquired from the official data of the Large Retail Trade Italian Association (Federdistribuzione). With the support of the personnel in charge of the hyper and supermarkets' waste management, the amount and type of organic leftovers produced on the regional territory was assessed. The expired and unsold organic products were grouped into animal-based (meat, milk..) and non-animal derived-leftovers (fruit, bread..). According to the amount of available leftovers and to their specific methane (CH_4) potential, an estimation of the regional CH_4 yield was produced. For fruit, vegetables, fish and meat' specific CH_4 potential, data were collected from the literature, where-as it was directly assessed for bread and milk-derived wastes (a mixture of 50% milk, 30% yo-gurt and 20% mozzarella cheese). Waste bread was directly received from a Torino supermar-ket, whereas dairy leftovers' samples coming from a facility in Alessandria specialized in collect-ing and managing this type of waste. Bio-methane potential (BMP) assays were performed in mesophilic conditions, according to VDI 4630 (2006) [7]. Two liters batch reactors were filled with a mixture of biomass and digestate (inoculum). All organic samples were tested in triplicate. Three reactors were filled up with inoculum only and used as blank. The methane yield of the tested organic leftovers was recorded along 60 days.

3. Results and discussion

3.1 Regional organic wastes availability and typologies

In Piedmont the most diffused retail typology is represented by the free service (49%) and supermarkets (27%) (Table 1).

Table 1: Typology of retail in Piedmont.

Typology	Number of retail
Superstore (>8000 m^2)	8
Superstore (between 2500 and 8000 m^2)	93
Supermarkets (between 400 and 2500 m^2)	518
Free service (between 100 and 400 m^2)	917
Discounts	348

According to the survey, the regional production of organic leftovers from the large-scale retail trade is of approximately 37 kt/year, 24 kt of which represented by non-animal based products and the remaining 13 kt by animal-based leftovers (e.g.…)meat, milk that corresponding about to 8,5 kg per resident and year. As shown in figure 1, non animal based products are mostly composed by unsold bread (about 60%), vegetables (21,5%) and fruit (18%). The remaining 0,5% consists of packaged foodstuff. The latter isn't suitable as feedstock for anaerobic digestion, due to the presence of packaging materilas. The separation of package from the remaining organic prod-uct is indeed time consuming, costly and requires specific equipment.

The fraction of organic waste consisting of fruits and vegetables is very variable, both in terms of quantity and composition being subject to seasonality. In the summer period the production of this type of waste increases, as well as fruits' average water content. At the contrary, in winter the fruit leftovers' availability decreases, with increasing dry matter content. Nonetheless, the amount of available leftovers as well as their humidity are key factors when they're meant to be used to feed anaerobic digestion plants throughout the year. Now this products is collected to composting plant.

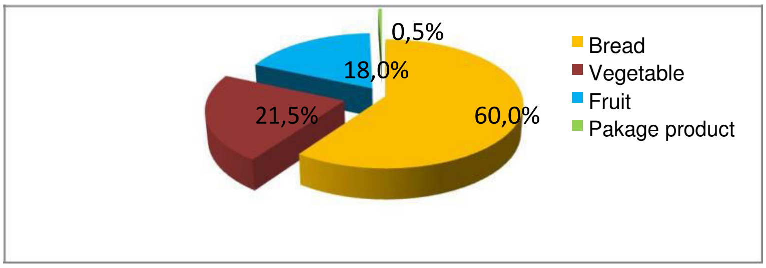

Figure 1: Components of non-animal based leftovers

Figures 2 and 3 show respectively the main components of fruit and vegetables leftovers avail-able on the regional territory. With special regards to the fruit compound, approximately 65 % of organic waste consist of high humidity materials (e.g. lemons, oranges, melons, strawberries and grapes), therefore these biomasses are difficult to preserve from aerobic/anaerobic degra-dation along storage.

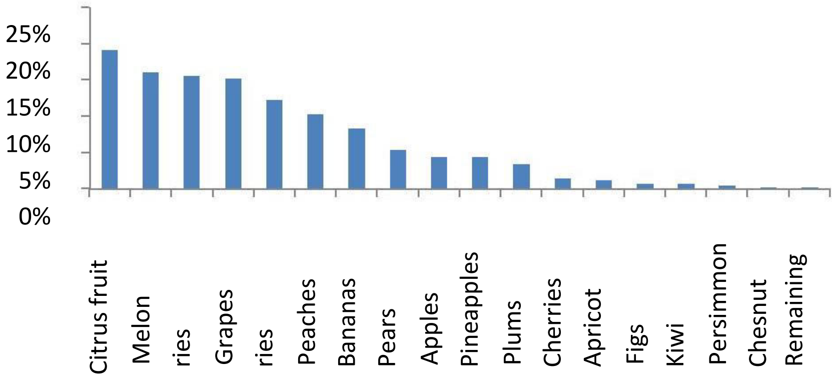

Figure 2: Average composition of fruit leftovers (percentages expressed on wet weight).

The same figure applies to vegetables leftovers. In this case lettuce and tomatoes alone repre-sent the 35% of the unsold organic materials.

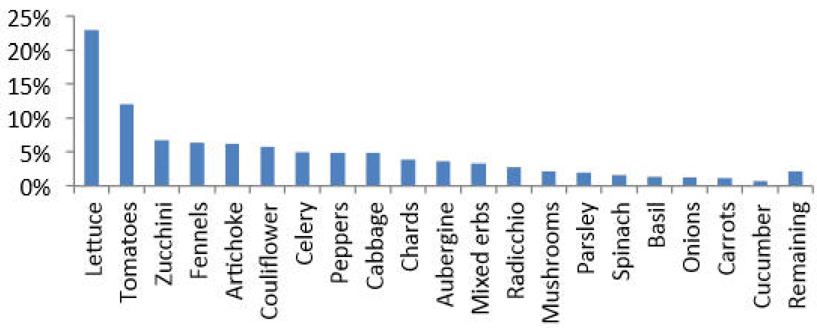

Figure 3: Average composition of vegetable leftovers (percentages expressed on wet weight).

The production of animal-based leftovers is less influenced by seasonal factors, as the composition of meat, fish and dairy products remains steady regardless to the climatic conditions. Meat-based leftovers are the most abundant feedstock (representing 51% of the animal-based organic wastes) whereas fish and milk are equally represented in the remaining unsold organic waste mixture. All animal-derived leftovers are suitable for feeding anaerobic digestion plants, being characterized by a high organic matter and nitrogen content and the presence of fats. Nevertheless, even though these biomasses are characterized by high specific methane potentials, their fast degradation kinetic may lead to an accumulation of fatty acids in the medium, thus representing an hazardous factor for the anaerobic digestion process stability. Now this products is collected to fertilizers plant and companies specialized in pets feed.

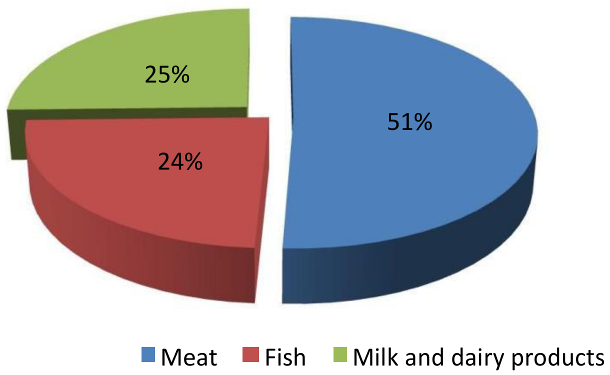

Figure 4 – Average composition animal product

3.2 Assessment of the regional leftovers' methane potential

Results of BMP assays carried out by DiSAFA are reported in table 2. Bread and milk yielded up to 700 and 1050 m^3 CH_4·t SV $^{-1}$ respectively. These value are two to threefold higher than those of an average corn silage (300-400 m^3 CH_4·t SV $^{-1}$). According to literature (table 2), organic leftovers from the mass-retail market yields from 230 to 480m^3 CH_4 per t VS.

Table 2: Specific methane yield of major large-scale retail organic waste.

Organic leftover	Specific yield (m^3 CH_4·t SV $^{-1}$)	Reference
Bread	730 (700-750)	DISAFA internal report
Dairy mix	1000 (1000-1050)	DISAFA internal report
Fruit	385 (320-450)	[4]
Vegetables	250 (230-280)	[2]
Meat	315 (230-400)	[5]
Fish	460 (440-480)	[6]

In table 3 the annual availability of organic wastes and their average organic matter content are summarized.
According to the produced leftovers, their organic matter composition and specific methane yield, approximately 10,600x10^3 Nm^3 of methane can be produced per year on a regional basis. 83% of the overall production might be achieved by the anaerobic digestion of unsold bread only, given its high availability (approximately 14.500t per year) and organic matter content.
Despite the 9600t of available fruit and vegetable leftovers and their high specific methane yield, their contribution to the potential regional methane production is relatively low (≈3%) due to their high water content. Expired animal-based products (dairy-mix, fish and meat) account for the re-maining 14% of the overall producible methane.

Table 3: Average total solid composition, organic matter composition and methane yield of the available leftovers.

Biomass	Yearlyavailability (t)	TS content (%)	VS content (kg t^{-1})	Yearly availability VS (t)	Yearly Yield (m^3 CH_4) x10^3
Bread	14,500	75 (70-80)	830 (800-850)	12,000	8,800
Fruit	4,300	12 (9-15)	145 (80-210)	400	100
Vegetables	5,300	15 (5-20)	75 (50-150)	630	250
Meat	6,600	25 (15-40)	230 (230-240)	1,500	450
Fish	2,400	25 (20-50)	230 (200-250)	550	250
Dairy mix	3,800	15 (10-30)	200 (120-600)	750	750

4. Conclusion and outlook

According to the survey, approximately one supermarket every 3000 inhabitants is available on the regional territory and a significant amount of organic leftovers are available annually as feedstock for anaerobic digestion plants. The energetic valorization of animal and non animal based unsold organic wastes would allow the production of approximately 10 millions m^3 of me-thane per annum. Given the 24.000GWhel. Piedmont electric energy requirement per year (Terna, 2014), by burning the producible methane by Combined Heat and Power Units (CHPs) it would be possible to cover only the 0,2% of the internal energy demand. Neverthe-less the use of such by products is still limited due to several factors, such as the lack of lo-gistic for their supply to the A.D. plants and the current normative in force. When used in combination with animal manure, agricultural by-products and/or energy crops as feedstock for agricultural A.D. plants, organic leftovers from the mass-retail market currently narrow the possible utilizations of digestate, by turning it into a waste. Thus, their energetic valorization need first to be supported by proper normative. Furthermore it must be underlined that most of the mentioned by-products are already recycled for animal feed production or composting. Thus the economic convenience of their energetic utilization must be further investigated.

References

[1] De Meester S., Demeyer J., Velghe F., Peene A., Van Langenhove H., Dewulf J., 2012.The environ-mental sus-tainability of anaerobic digestion as a biomass valorization technology. Bioresource Tech-nology (121), October 2012, Pages 396–403

[2] Garcia-Peña E.I., Parameswaran P., Kang D.W., Canul-Chan M., Krajmalnik-Brown R., 2011. An-aerobic digestion and co-digestion processes of vegetable and fruit residues: Process and microbial ecology. Bioresource Technolo-gy 102, 9447-9455

[3] González-García S., Bacenetti J., Negri M., Fiala M., Arroja L., 2013. Comparative environmental per-formance of three different annual energy crops for biogas production in Northern Italy. J Cleaner Production, 43 (2013), pp. 71–83

[4] Gunaseelan V.N., 2004. Biochemical methane potential of fruits and vegetable solid waste feedstock. Biomass and Bioenergy 26, 389-399

[5] Luste S. , Luostarinen S., Sillanpää M., 2009. Effect of pre-treatments on hydrolysis and methane pro-duction po-tentials of by-products from meat-processing industry. Journal of hazardous materials 164, 247-255

[6] Kafle G.K., Kim S.H., Sung K.I., 2013.Ensiling of fish industry waste for biogas production: A lab scale evaluation of biochemical methane potential (BMP) and kinetics. Bioresource Technology 127, 326-336

[7] VDI 4630, 2006. Fermentation of organic materials, Characterisation of Substrate, Sampling,

[8] Collection of Material Data, Fermentation Tests, VDI Gesellschaft Energietechnik. [2] Writer, (2014): Compost is the best, Journal 4, 13-19.

[9] DiSAFA, 2014. Biogas and methane potential of organic waste. Internal report.

[10] Terna, 2014. Dati provvisori di esercizio 2014.

TD-O_12 New tool for improving management of biogas digesters – A heat transfer- and biogas production model for anaerobic digestion

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Abbreviations: HRT, hydraulic retention time; RMSE, Root Mean Square Error; MBE, Mean Bias Error; MAE, Mean Absolute Error.

1. Objectives

Accurate prediction of slurry temperature in anaerobic digesters plays a key role in the management, design, and optimization of digester performance. The objective was to develop a flexible and accurate model for the prediction of digester temperature that can be used to design new digesters and select designs for a specific environment. The model should offer accurate predictions for a variety of digester configurations, with the possibility of coupling it to biogas production models. The model can be used for simple digesters traditionally not heated as presented here, and also for the design of a traditional CSTR.

2. Methodology

A heat transfer model was developed based on [1], with significant changes. The model uses a finite-difference approach with a fixed time-step (5 minutes). Heat sources and sinks include solar radiation, soil, external heating and slurry inflow and outflow. Heat transfer processes include radiation and convection. Convective heat transfers are based on convection coefficients, which may vary widely in value, and these must be estimated prior running the model. Solar radiation is calculated for each hour based on latitude, longitude, altitude and cloudiness.

The model can use detailed weather data supplied by the user, or a built-in three-parameter sine approximation based on the annual maximum and minimum mean air temperature and the hottest day of the year. The applied methodology enables the use of the model with detailed or approximated inputs, making it useful for decision-support on digester management practices in areas where data availability differ.

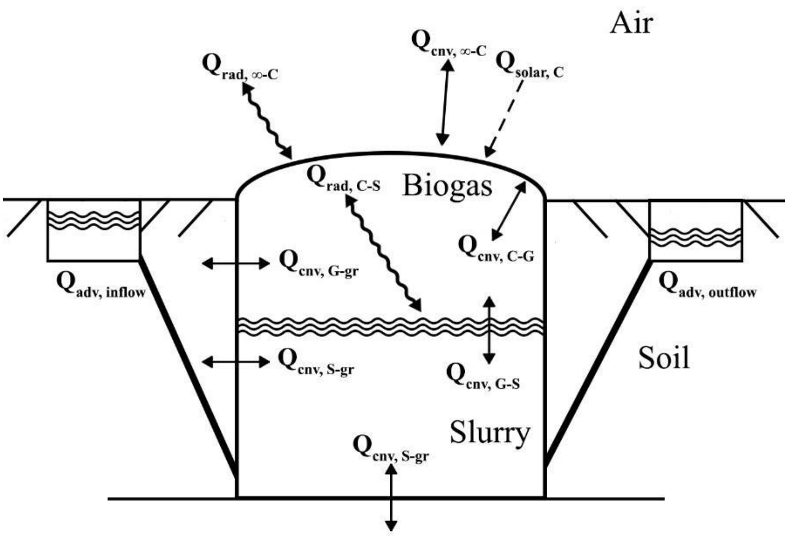

Figure 1: Heat transfer processes included in the model. The digester presented is located below ground, with only the cover exposed to ambient air. Abbreviations: *rad*, radiative heat transfer; *cnv*, convective heat transfer; *adv*, heat transfers by advection; *solar*, radiative heat transfer by solar insolation. *C*, cover; *G*, gas; *gr*, ground; *S*, slurry; *Q*, heat transfer process.

3. Results and discussion

3.1 Model Overview

This heat transfer model for anaerobic digestion uses one or two parameter files: a general parameter file; and an optional weather file containing daily average ambient air temperatures. The parameter file lets the user define digester geometry and dimensions, slurry content, slurry loading rate and digester location amongst others. A second parameter file contains heat transfer coefficients and related parameters that will typically not be changed by the user. The model creates three output files containing average daily temperature of the slurry, detailed hourly temperature of the slurry, and heat transfer rates for each time step. The model is written in Fortran90, and a one-year simulation runs on a modern laptop computer in less than 2 seconds, including Read/Write operations. Executable versions can be made for Windows, Mac, or Unix-like operating systems.

3.2 Model Simulation of Uninsulated Anaerobic Digester

Psychrophilic (<20 ºC) and psychrotrophic (20-30 ºC) conditions of anaerobic digestion are characterized by low biogas production, and typically occur in the winter months of subtropical, highland climates [2]. For rural areas with simple anaerobic digesters, and low availability of technology, the implications can be biogas shortage for cooking and heating. Prior studies have resulted in the development of heat transfer models to a variety of digester configurations [1, 3, 4], for the optimization of digester design and management. However, these models typically are not publicly available and furthermore require expensive proprietary software for execution. The novelty of this proposed simple heat transfer model is its simplicity, little computational demand, wide accessibility and flexibility in input parameters.

Predicted slurry temperatures were compared to measured slurry temperatures obtained from a study by Pham et al. [2]. The digester was buried to a depth of 2.6 m, at the Thuyphuong Pig Research Centre, National Institute of Animal Science, Hanoi, Vietnam, with only the digester cover exposed. Digester volume was 7.0 m3, with a working volume of approximately 5.0 m3. The feedstock used was pig manure, with a feeding rate of 0.14 m3 day-1, and a HRT of 40 days. Data were collected from July 2012 to February 2013.

The model was run using the parameters presented in Table 1, with subsequent comparison to the measured data from [2]. Convection coefficients were based on correlations presented in [1]. While measurements were conducted from July 2012 to February 2013, the heat transfer model has been evaluated for a full 365 days.

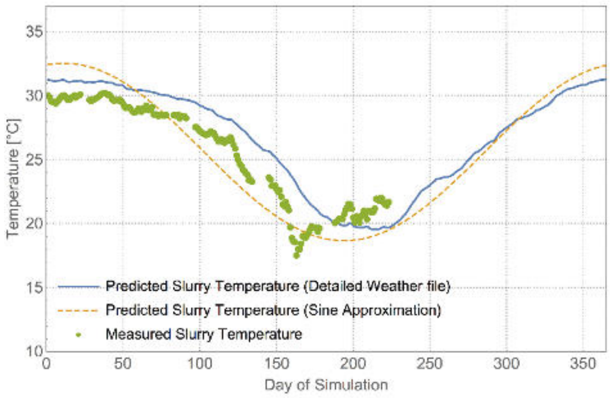

Figure 2: Predicted slurry temperatures simulated with air temperatures assessed from a detailed weather file supplied by [2] (solid line), and a three-point sine approximation (dashed). The first day of simulation corresponds to 27th July 2012.

Parameters for the sine approximation, annual minimum and maximum mean air temperature, were obtained from Noi Bai International Airport, Hanoi, Vietnam, through Wolfram Alpha LLC, Wolfram Research, Champaign, IL. Over an 8-year period from the 1st January 2006 to 1st January 2014, the average maximum and minimum temperature was 31 °C and 19 °C respectively. The hottest day of year was estimated to be day 180 (June 28th). The model successfully calculates the magnitude and annual response of the daily slurry temperatures, for both detailed and approximated weather data (Figure 2). Using a detailed weather file,
RMSE = 1.9 °C, MBE = 1.3 °C, and MAE = 1.7 °C. Results were similar with the sine-prediction of daily mean

ambient air temperature: RMSE = 1.9 °C, MBE = -0.3 °C and MAE = 1.7 °C. For both simulations the Nash-Sutcliffe Model Efficiency Coefficient was 0.8, indicating that the model in both cases is a better predictor than the observed average slurry temperature.

Table 1: User-supplied parameters and heat transfer parameters used for simulation shown in Figure 2 and 3.

Heat Transfer Parameters				
Parameter	Slurry	Air		Biogas
Density [kg m^{-3}]	1000	1.205		1.156
Specific Heat Capacity [J kg^{-1} K^{-1}]	4180	1010		1682
	Cover to Air	Cover to Biogas	Slurry to Biogas	Slurry to Substrate
Convective Heat Transfer [W m^2 K^{-1}]	10	1	2	15
User Supplied Parameters				
Parameter	Value	Parameter		Value
Height of Digester [m]	2.5	Slurry Production [ton day^{-1}]		0.075
Length of Digester [m]	2.0	Latitude [°]		21.2
Width of Digester [m]	1.3	Longitude [°]		105.8
Slurry Depth in Digester [m]	1.5	Time Zone		+ 7.0 GMT
Meters Above Sea Level [m]	100	Placement [m]		-2.5

The sinusoidal approximation of the annual temperature regime is in this instance a valid replacement of the detailed weather file, had it not been available. The mean error quantities indicate that the model accuracy should be good enough to distinguish if digester design or management may reach psychrophilic or psychrotrophic conditions, even though some residuals are larger than the average indicated by the RMSE, MBE and MAE.

3.3 Demonstration of Model Application to Different Designs and Management

The three-parameter sinusoidal approximation of the annual ambient air temperature regime is demonstrated in Figure 3a. With data collected from Noi Bai International Airport, Hanoi, Vietnam, this approximation is a reasonable substitute for the detailed weather file, should it be unavailable. In remote rural areas this could be useful for simulation of different designs and management practices, as nearby weather stations may not exist.

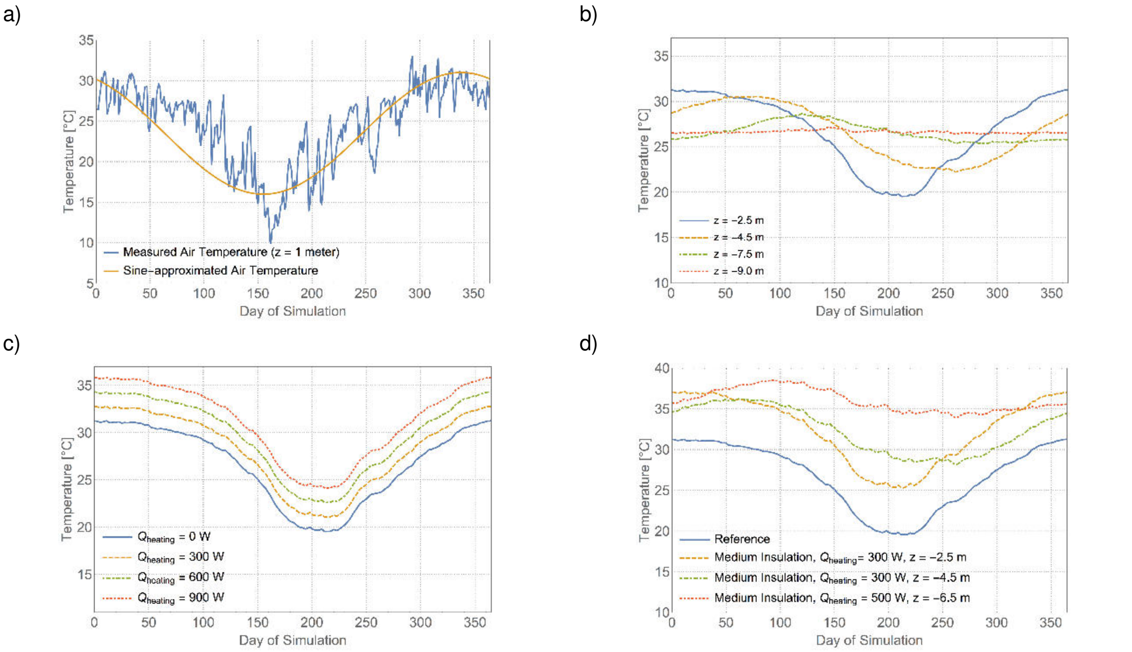

Figure 3: Demonstrations of model application. a) Comparison of input weather file with observed air temperature, and the three-parameter sinusoidal approximation of the annual temperature regime. Figures b) – d) show predicted slurry temperature. b) Effect of burying the digester further below ground, inaccessible to the ambient atmosphere. Reference depth is z = -2.5 m. c) Simulations with external heating added to the slurry in the anaerobic digester. The specified power for heating is applied throughout the whole year. d) Simulations with more complex design and management practices. Simulations are a combined result of variation in external heating, digester depth and insulation thickness.

Figures 3b through 3d present simulations with different digester design and management, all of which have been simulated based on the detailed weather file. Figure 3b illustrates the reference scenario (as shown in Figure 2), dug 2.5 m into the ground. To demonstrate application of the model, remaining calculations have been performed at -4.5 m, -7.5 m and at -9.0 m. As the depth of the digester increases, the annual temperature fluctuation begins to decrease in amplitude, and get displaced from the original temperature phases. At -9.0 m, where the solar insolation has little effect as opposed to soil surface, the predicted slurry temperature approaches the mean earth temperature of the location. The anaerobic digestion conditions are still psychrotrophic, but biogas inhibition due to temperature shock should no longer be a problem.

Although external heating may not be available in remote rural areas, it could still be relevant to estimate the power required to elevate predicted slurry temperatures to an acceptable level. Figure 3c illustrates the predicted slurry temperatures at various heating inputs. The model applies the external heating input to the entire simulation period, and not just the cold winter months. For this reason the predicted slurry temperature profile is translated to elevated temperatures. Supplying external heat with 900 W will relieve the digester from psychrophilic conditions during the cold winter months, and allow the digester to be maintained in the mesophilic temperature range, to a greater extent than before.

Figure 3d presents simulations with the combined effect of varying digester design and management parameters. The reference simulation (blue) corresponds to that presented in Figure 2. The results indicate that entirely mesophilic conditions can be achieved by placing the digester 6.5 m below ground, adding a medium thickness layer of insulation and heating with the effect of 500 W. The addition of a medium thickness layer of insulation was achieved by reducing the slurry to substrate heat transfer coefficient by a factor of 4. Demonstrations of model predictions can thus be based on manipulating a single parameter, or looking for the combined effect of several variables. This is particularly helpful in the designing of new digesters, but can also indicate whether a change of digester management practices could enhance the annual biogas production, also in the cold winter months.

4. Conclusion and outlook

Preliminary calculations for an uninsulated digester at the Thuyphuong Pig Research Centre, Hanoi, Vietnam were simulated, and model performance assessed by RMSE, MAE and MBE. This, together with the Model Efficiency Coefficient, indicate that the model perform well, even when seen in the light of the simplifications made to build a simple model. The flexibility of the model and modest amount of input parameters makes it a useful decision-support tool for biogas digester design and management practices in developing countries. Future work will include coupling it with the biogas production model presented by [5], and improving the methods of determining the convective heat transfer coefficients. The project is still a work in progress, and model performance evaluation with additional data is pending. The model will be publicly available online when completed.

Acknowledgements

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References

[1] Terradas-Ill, G.; Pham, C. H.; Triolo, J. M.; Martí-Herrero, J.; Sommer, S. G. Thermic model to predict biogas production in unheated fixed-dome digesters buried in the ground. *Environ. Sci. Technol.* 2014, *48* (6), 3253–3262 DOI: 10.1021/es403215w.

[2] Pham, C. H.; Triolo, J. M.; Sommer, S. G. Predicting methane production in simple and unheated biogas digesters at low temperatures. *Appl. Energy* 2014, *136*, 1–6 DOI: 10.1016/j.apenergy.2014.08.057.

[3] Perrigault, T.; Weatherford, V.; Martí-Herrero, J.; Poggio, D. Towards thermal design optimization of tubular digesters in cold climates: a heat transfer model. *Bioresour. Technol.* 2012, *124*, 259–268 DOI: 10.1016/j.biortech.2012.08.019.

[4] Axaopoulos, P.; Panagakis, P.; Tsavdaris, A.; Georgakakis, D. Simulation and experimental performance of a solar-heated anaerobic digester. *Sol. Energy* 2001, *70* (2), 155–164 DOI: 10.1016/S0038-092X(00)00130-4.

[5] Rennuit, C.; Sommer, S. G. Decision Support for the Construction of Farm-Scale Biogas Digesters in Developing Countries with Cold Seasons. *Energies* 2013, *6* (10), 5314–5332 DOI: 10.3390/en6105314.

TD-O_13 A Flemish case study of possible synergies between high quality green compost and green energy

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1. Objectives

The SYNECO-project tested the feasibility of

- anaerobic digestion of fine fractions of green waste before composting, and
- using sieving overflow (pre- or post-composting) as biomass for incineration.

Monitoring the process and the analysis of obtained compost and biomass should lead to improved quality protocol and a persistently high quality of the end products[4]. These results & a decision-tool aim to help transform the composting sector to a more synergetic and economically viable player.

2. Methodology

Repeated biogaspotential-tests (wet mesophile versus dry thermophile anaerobic digestion) were conducted on fine fractions of green waste. As to incineration, Vlaco – through mass balances and an inventory of green waste of participating compost sites – defined the current levels of output (compost, mulch, biomass,..) versus available levels of bulking agent in green waste. During several periods of the 2-year project 'reference' compost batches (with 'black biomass' taken as overflow after composting) as well as 'experimental' compost batches (with 'white biomass' taken as overflow before composting) were set up and monitored. Input materials, biomasses and composts were analysed and compared. Finally, an online tool was constructed in GAMS (linear programming) – simulating the highest possible free cash flow – to help composters optimalise their compost-biomass balance. Or:

- Method 1: 3 x 2 biogaspotential-tests fine fractions of shredded green waste (<20,<30,<40mm)
- Method 2: mass balances (I/O) of composting sites & sampling green waste (% bulking agent)
- Method 3: 3 x analysis before composting of shredded green waste & biomass (overflow fractions >20, 20-80, 20-100, >30, 30-60, >40, and > 60mm)
- Method 4: 3 x monitoring composting process
- Method 5: 3 x analysis of composts and biomasses (overflow) after composting
- Method 6: literature study and interviews Vlaco and -members for construction decision-tool

3. Results and discussion

3.1 Low biogaspotential green waste

As shown in figures 1-3 – wet and dry biogastests over 3 'seasons'[5] x 6 samples – no coherent relationship was found between maximum particle size (used sieve) of green waste and biogaspotential.

[4] High organic matter-standards in Flanders (>16%) – impeding substantial substraction of bulking agent
[5] March-may '13 (spring), august-september '13 (summer), january-march '14 (winter)

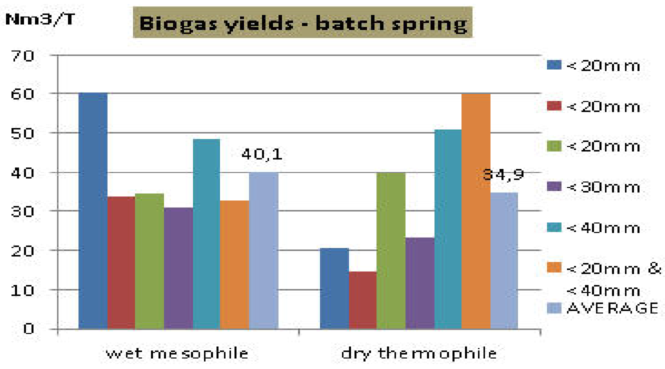

Figure 1: Biogas-yields (wet mesophile and dry thermophile) of fine fractions shredded green waste taken in spring '13.

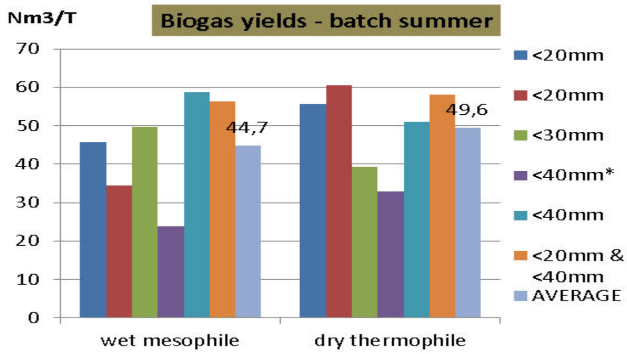

Figure 2: Biogas-yields (wet mesophile and dry thermophile) of fine fractions shredded green waste taken in summer '13.

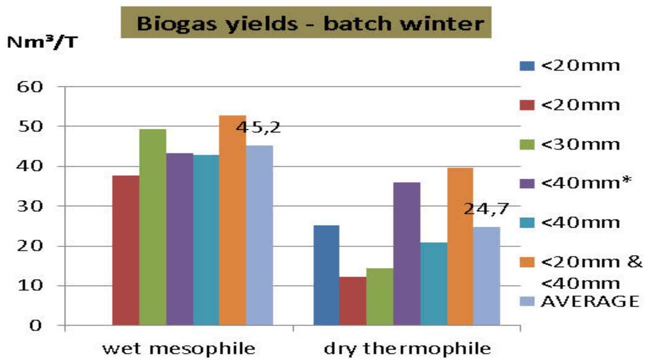

Figure 3: Biogas-yields (wet mesophile and dry thermophile) of fine fractions shredded green waste taken in winter '14.

Regardless of the surprising differences in biogas-yields between the two anaerobic digestion-techniques, wet as well as dry fermenting of various subfractions of green waste have lead to fairly low biogas yields averaging between 24,7 and 49,6 Nm³/T fresh matter. Highest biogaspotentials, measuring around 60 Nm³/T, were found in both wet mesophile and dry thermophile tests. Although on average wet digestion outperformed the dry digestion twice, the highest average biogas yield was obtained through dry, thermophile fermentation of green waste originating from the 'summer' batch. Moreover these summer samples were the only ones still showing considerable biogas accumulation at day 21 (end dry test) and day 25 (end wet test). Biomethane-contents oscillated between 51% and 55%. In comparison with other (co-)digested inputs, the anaerobic digestion of fine fractions of green waste yield low amounts of biogas/biomethane. It is furthermore plausible that green waste contains more sand and accidental plastic and metal objects which cause technical problems especially for wet digestion (more than 80% of Flemish digestion facilities). Given, ultimately, the large cost of an investment of an additional digestion-unit (or extra handling & transportation costs), this synergetic scenario in the optimization of green composting was discarded in further tests and simulations.

3.2 Synergy of pre-/post-composting biomass (overflow) for incineration and compost?

3.2.1 Availability and necessity of bulking agent in green composting

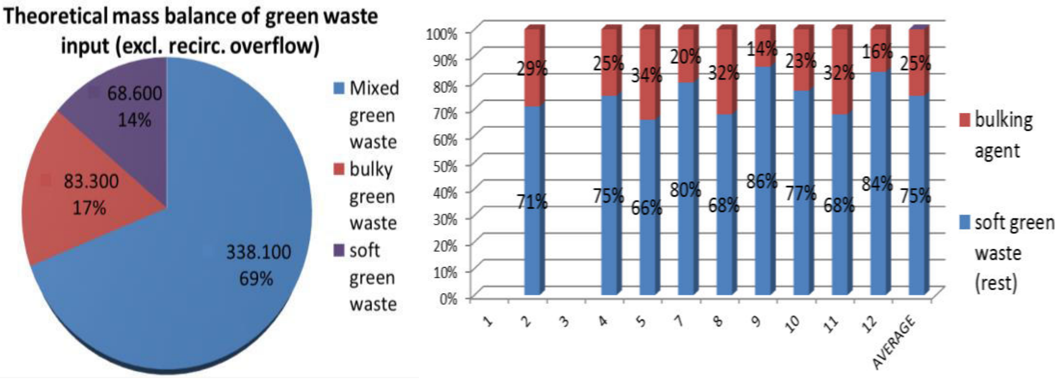

Figure 4(a,b): theoretical mass balance ’11-’12 green composting and sorting analysis bulking agent in green waste ’12-’13.

Mass balances (input) of 12 composting sites were extrapolated (Fig 4a) to a Flemish average (26 sites) of 69% mixed green waste input, 17% woody green waste input, and 14% soft green waste input – together 490.000 tonnes (2012). A better knowledge of the share of ‘bulking agent’ was obtained by 3 sorting campaigns[6] at 9 sites during which the weight-percentage bulking agent (hedge clippings, branches, prunings and roots) was determined in total fresh green waste input: green waste received by composting sites consists on average of 1/4th (25%) bulking agent (Fig 4b). It should also be noted that differences between seasons and composting sites are considerable, e.g. winter with highest average % bulking agent (56%). Combining these findings with mass balances-data of tonnes recirculating sieve overflow (equally considered bulking agent) Vlaco established a yearly share of 1/3rd (32%) bulking agent in the set-up material of green composting in Flanders. Backtracking the compost qualities of compost sites with lowest share of bulking agent in spring and summer, specifically looking at levels of organic matter (OM)[7] of compost, a rule of thumb was confirmed stating 20% bulking agent at compost set-up as a prudent share to obtain compost with a sufficiently high amount of OM. This would ultimately mean a potential of biomass for incineration of around 10% of the tonnes of green waste set-up for composting every year, thereby potentially raising 31 GWel and 80 GWth or an equivalent yearly energetic consumption of respectively 9.000 households (electricity) and 3.500 households (heat).

3.2.2 White biomass and experimental (EXP) composting versus referential (REF) composting and black biomass

Actual synergies between obtaining ‘good’[8] compost and extraction of significant amounts of biomass for incineration (green energy) were tested in 3-rounds[9] at 10 compost sites in which biomass was defined as the sieve overflow. Vlaco offered guidelines for a more intensive experimental composting – a higher turning frequency of the pile – since literature and prior experience pointed to the necessity to offset lower amounts of bulking agent with higher turning frequencies (or forced aeration or composting in rows instead of tables). Most importantly we found (Table 1[10]):

[6] Dec’12-feb’13 (‘winter’), april’13 (‘spring’), aug-sept’13 (‘summer’)

[7] Minimum Flemish requirement (16% OM) & Vlaco quality label-requirement (18% OM)

[8] Reference are the FOD (B)-requirements: >16% OM, max 1 weed seed, DM >50%, OUR <15

[9] Start-up: march-may ’13 (spring), august-september ’13 (late summer), january-march ’14 (winter)

[10] Table 1 & 2: ADM: absolute dry matter; ‘Paired T’-test: to indicate significant difference, i.c. if < 0,05

Table 1: Average values of some main characteristics of reference versus experimental compost.

round	compost averages	EC	DM	density	OM	OM	C/N	OUR	N_total	P_total	K_total	Fe_total	Al_total	pH-H2O	germ. Seeds
		µS/cm	%/FM	g/l fresh substr	% (FM)	%/ADM	-	mmol/kg OM/hour	%/ADM	mg/kg ADM	mg/kg ADM	mg/kg ADM	mg/kg ADM	-	#/l
spring	REF	615,88	68,78	425,18	23,36	33,88	18,36	8,66	1,06	1833,21	7914,05	6177,63	3151,81	7,88	0,25
	EXP	613,50	66,73	457,64	21,88	32,45	17,53	9,06	1,05	1840,59	7470,05	6426,69	3449,50	7,94	0,13
paired T		0,97	0,32	0,12	0,36	0,40	0,23	0,72	0,94	0,88	0,25	0,19	0,02	0,85	0,35
late summe	REF	1046,60	52,03	518,28	17,60	33,79	12,37	8,81	1,52	2593,20	12091,34	5234,30	3620,70	8,68	0,00
	EXP	1083,60	53,60	533,42	17,77	33,67	12,79	6,12	1,50	2581,69	13287,59	6475,00	4067,30	8,60	0,00
paired T		0,58	0,64	0,16	0,88	0,97	0,79	0,03	0,46	0,89	0,42	0,23	0,24	0,72	1,00
winter	REF	581,57	56,44	458,32	19,49	34,57	18,42	5,76	1,06	1684,12	5311,79	5760,00	3177,57	8,52	0,00
	EXP	482,79	55,06	487,37	17,70	32,17	17,24	3,61	1,05	1659,84	4751,91	5602,07	3358,36	8,49	0,00
paired T		0,09	0,525	0,027	0,004	0,042	0,096	0,507	0,745	0,825	0,205	0,813	0,510	0,907	1,000

- Reference composts and experimental composts are physico-chemically usually very similar irrespective of composting technique (table, row, tunnel, forced/passive aeration) and turning frequency which seem to imply year-round possibilities to extract overflow before composting (white biomass). In winter though statistically significant differences in notably OM indicate that monitoring is necessary and/or using higher size sieves (>40mm or higher) for extracting white biomass is more prudent. Also in summer, due to the low ratio of C/N of input material (grass) and in compost, it is not advisable to take out of the loop sizeable amounts of overflow, neither before nor after composting. A final sieving at the end of the composting process proved always necessary,
- A higher frequency of turning than 1x/month seems advisable to reduce risk of survival of weed seeds, higher vulnerability to humidity, and to improve compost maturity. A higher frequency (and/or forced aeration and/or row composting) can reduce required duration/surface of the compost process without jeopardizing compost quality. Too frequent turning though (2x/week or more for row composting & 1x/week or more for table composting) undermines a build-up and maintaining of sufficient heat and thus hygienisation of the compost,
- Smell was never problematic in experimental composting,
- There are some statistically significant differences between ‘black’ and ‘white’ biomass (Table 2) depending on which season/batch is considered.

Table 2: Average values of some main characteristics of ‘black’ versus ‘white’ biomass.

round	biomass averages	OM	ash	DM	density	C/N	K_total	Mg_total	Ca_total	Na_total	stones > 5 mm	# impuri-ties > 2 mm	Cl	S	lower heating value (ct mbar)
		%/ADM	%/ADM	%/FM	g/l fresh substr	-	mg/kg ADM	mg/kg ADM	mg/kg ADM	mg/kg ADM	%	%	mg/kg ADM	mg/kg ADM	MJ/kg FM
spring	BLACK	89,0	11,0	61,0	190,5	78,9	8094,8	1210,9	8333,9	482,4	0,1	1,0	4010,7	846,1	9,3
	WHITE	87,1	12,9	65,1	144,8	58,4	6287,1	1056,3	8514,9	288,5	0,1	2,3	2105,8	814,5	9,3
paired T		0,70	0,70	0,41	0,20	0,13	0,17	0,43	0,88	0,00	0,90	0,44	0,26	0,73	0,99
late summe	BLACK	78,5	21,5	45,9	305,0	41,2	11968,5	1759,3	12203,2	562,0	0,3	0,4	1426,3	785,3	5,3
	WHITE	74,9	25,1	56,2	138,7	35,6	15683,4	1835,1	11742,1	473,2	0,0	0,8	1123,1	1348,8	7,3
paired T		0,60	0,60	0,11	0,01	0,34	0,08	0,73	0,74	0,49	0,26	0,45	0,60	0,02	0,17
winter	BLACK	83,1	16,9	55,3	230,6	59,8	5889,8	1283,6	10821,1	430,0	1,7	0,5	609,9	912,5	7,8
	WHITE	84,9	15,1	48,0	252,3	61,3	4171,2	1140,9	9832,5	245,5	0,5	0,7	1073,5	741,6	6,0
paired T		0,69	0,69	0,31	0,36	0,88	0,03	0,59	0,72	0,02	0,33	0,67	0,01	0,19	0,30

In general woody overflow extracted during the second batch (summer) shows lower levels of OM (and C/N) and higher levels of ash and alkaline elements (K, Mg, Ca, Na) and thus seems less appropriate for incineration. Biomass-fractions of >20mm or more are energetically more interesting – lower sieves tend to give biomass with lower OM and dry matter (DM) and higher alkaline elements as well as S(ulphur) – although higher sieve fractions (>40mm or more) can on the other hand have a higher % of impurities. Strong acceptance standards and an additional purification (e.g. windshifting) is advisable,

- Biomass from the 1st batch (spring) has excellent average calorific values (9,3 MJ/kg FM) whereas biomass form third batch (winter) has a more moderate lower heating values (7,8 and 6 MJ/kg FM). It is highly recommended to maximize dry matter by choosing the right moment to sieve (precipitation) and/or allowing the biomass to dry before selling as input for incineration,

- From the perspective of temperature/hygienisation experimental batches are advisably no taller than 3 meters,
- All crucial observations during the SYNECO-project:
  - were summarized and formalized in a Code of Good Practice and Quality Protocol, and
  - used for conceiving the internet-based decision-supporting tool (GAMS software) which simulates highest free cash flows for green composting and extracting biomass (overflow) for incineration.

4. Conclusion and outlook

Biomethanisation of subfractions is disregarded as a synergetic option. For green composting Vlaco considers min. 20% bulking agent at set-up as a rule of thumb. Extracting fractions white/black biomass is defendable especially during winter and spring although OM-levels in compost need to be continually monitored. In this synergy a more intense composting process (forced aeration and/or higher turning frequency) has clear advantages although increasing turning frequency may result at some point in lack of hygienisation. Between late spring and late autumn biomass-overflow is not advised. Described synergies will theoretically yield higher net cash flows mainly due to the possibility of more input and thus higher gate fees.

Acknowledgements

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References

[1] Huybrechts & Vrancken, (2012), Best Beschikbare Technieken voor composter- en vergistingsinstallaties (BAT)
[2] Gybels, R., Wouters, R., Schuurmans, B. & Verbeke, W. (2012): Houtige biomassa voor energie in Limburg. Eindrapport van het MIP2-project “Limburgs groen voor een groene economie”, 159 pp
[3] Vandaele E. (2009): Ecological and economical benefits of compost – Abstract, 9 pp (http://www.vlaco.be)
[4] Vandenbroucke I. (2009): Optimalisering compostering, 31 pp (http://www.vlaco.be/node/630)
[5] Vlaco, QA-data compost of Flemish composting sites

TD-O_14 Environmental infrastructural investments in peri-urban areas – How to overcome nimbyism in case of waste to energy biogas plants

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1. Objectives

Finding new locations for environmentally friendly infrastructural facilities is not an easy task. NIMBY (not in my backyard) social resistance syndrome usually accompanies them: everybody wants to live in a high quality place but nobody wants to bear the associated costs. For facilities processing organic waste resistance of the local population has been identified as the biggest siting problem. In the near future erecting of new infrastructural facilities for waste processing will become a challenging task in Poland, *i.a.* due to the EU requirements to reduce quotas of landfilled organic fraction of municipal solid waste (OFMSW). Their development has been constrained as local inhabitants fear odour impacts, intensification of traffic as well as a loss of the property's value. This article outlines a multicriteria approach to location procedures for anaerobic digestion (AD) plants processing organic waste, with a special focus on social acceptance criteria. The integration of spatial issues within the analytical hierarchy method (AHP) is supposed to provide a methodology to negotiate location trade-offs between concerned inhabitants, municipalities and investors.

2. Introduction: Nimbyism and trade-offs to find the best site

Rapid urbanization has resulted in siting dilemmas and problems to find new locations for environmentally friendly infrastructural facilities (such as renewable energy sources- RES or waste processing plants). Urbanisation and intensification of economic development lead to local conflicts, especially in densely populated areas. Any industrial, intensive agricultural facility or technical infrastructure can be classified as a potentially conflict bearing undertaking. In particular suburban areas and small towns are potential NIMBY arenas. Typically, conflicts are evoked by a sense of unfair distribution of benefits and costs resulting from the emergence of a new investment. In fact environmental benefits are of global nature, while the costs are born locally. Conflict situations have their origin in non-uniform (and thus unjust) spatial distribution of local benefits [1].

Inhabited areas generate waste and have higher energy demands, therefore, from the economic point of view (costs of logistics), new infrastructure should be located in their proximity. However, the urbanisation process and urban-sprawl result in decreasing amount of available greenfields in city outskirts and city-village transition areas. For spatial planners location of environmental infrastructure facilities is, thus, challenging and requires elaboration of substantive arguments.

In case of AD facilities it is not only to make people accept the new investment site but also to actively involve them in the waste management process. The majority of the European AD plants fuelled with OFMSW (*c.* 60%) use source separated organics (SSO), which means they require active participation of the local population in the selective collection process. A typical example is the AD plant in Bayern, Germany. Within the Munich metropolitan region two AD plants are fuelled by OFMSW. One is located within the city of Munich and operated by the city utility company AVM, where SSO of the city of Munich are processed. The other one (in operation since 1997) is located in Kirchstockach and takes in SSO from the city ring administered by the Landkreis München. Biotonne is directed to the plant from a distance of up to 40 km from the area of *c.* 667 km^2 (Munich province) from over 400 thousand residents. A well-developed network of roads and highways around Munich allows for efficient logistics. The annual capacity of the plant amounts to 30.000 Mg per year. The plant operator stressed the need to introduce proactive educational measures in close cooperation with local authorities. Such activities include open days, and were organized in Kirstockach already a couple of times. The facility has also been exhibited during the "Energy days" festival, when local people could learn about the AD technology and separate collection of kitchen and garden waste.

In analysed European regions with a higher population density, where such infrastructural investments are inevitably integrated into the urban tissue, the trend is to use existing brownfield sites such as composting, MBTs' sites, WWTPs (especially in smaller towns) for the purpose of their extension. 10 case studies in Europe were analysed in detail from the point of view of environmental, organizational, economic, technological, social and spatial criteria. This part of the project enabled to outline initial multicriteria evaluation scheme.

Table 1: Chosen spatial parameters in 10 European case study analysis.

		Scandinavia			Western Europe			Southern Europe			CEE
NUTS3 Typology		IN	IN	PRC	PU	PU	PU	PU	PU	PU	PRC/UP
Population density	Inh./km^2	120	114	82	496	496	838	694	694	1941	60
Brownfields/ Greenfields		B	B	B	B	B	B	G	B	G	B
Landuse requirements	ha	1.5	12.2	2.3	4.6	0.5	4.9	3	1.2	3.9	1.2
Distance to inhabited areas	m	500	400	100	250	230	200	300	300	200	1000
AD Plant typology		SSO	COD	COD	SSO	COD	SSO	SSO	MBT	SSO	M
Plant size	'000 Mg/a	50	85	50	30,5	2	25	80	90	25	85

PRC - Predominantly rural (region) close to a city, PRR - Predominantly rural remote (region), IN - Intermediate (region), PU - Predominantly urban (region)

MBT mechanical-biological treatment plants with AD of organic fraction of mixed municipal solid waste, SSO- AD plants fuelled by source selected organics (SSO), COD- AD plants fuelled by SSO in codigestion with other agricultural materials.

3. Results

3.1 Identification of relevant criteria to alleviate the effects of nimbyism

The key element of this research is to elaborate a multicriteria decision analysis tool to be used in location procedures in Poland. The preliminary version of the tool was designed based on experiences from other countries (study visits, literature reviews) as well as own analyses. The Analytic Hierarchy Process (AHP) combined with a spatial analysis tool (GIS) will be applied to justify the choice of the best site. The AHP is a multicriteria method that was originally developed in 1977 by Satty [2]. In short, this technique is based on the hierarchical structure of the problem followed by the evaluation of options by pair-wise comparisons. In the literature it can be found using this approach to evaluate localization issues using a large number of different criteria [3], [4].

In this project the pair-wise comparison of tier I criteria: *C1. Impact on inhabitants C2. Impact on the environment C3. Economic feasibility* is performed. Tier 2 and 3 subcriteria relevant to the impact on inhabitants include issues such as:

- B1 IMISSIONS/QUALITY OF LIFE, with a C level (tier III) criteria: distance to inhabited areas and other receptors (noise, odour), distance to protected drinking water intake areas, interactions (cumulated impact);
- B2 MATERIAL AND INTANGIBLE ASSETS ON A LOCAL LEVEL, with a C level (tier III) criteria: impact on agricultural land (in particular ecological farming), recreation, amenity and tourism (loss of trading income), loss of real property's value, new employment;
- B2 CULTURAL HERITAGE/LANDSCAPE, with a C level (tier III) criteria: distance to cultural heritage places, distance to conservation areas, visual influence zones, visual or other intrusion (*e.g.* changes to landscape character, noise in a quiet setting), visual effects, changes in views of a landscape (townscape).

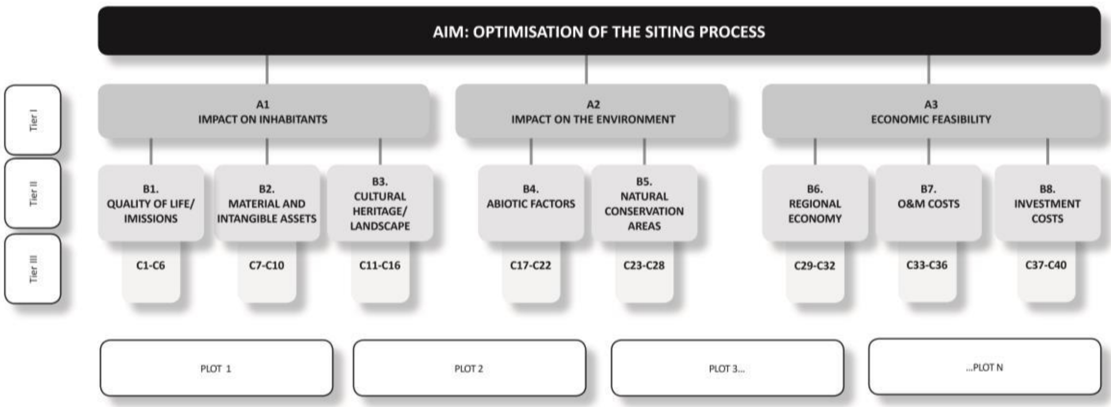

Figure 1: A structure for a multicriteria evaluation of plots available for the location of a AD plant.

Case study municipalities are to be located in waste management regions (administrative units abbreviated RGO in Polish), defined by the Polish law as regions with at least 150 thousand inhabitants. Such administrative entities are statistically reviewed in order to calculate the level of waste generation and the possibility to obtain OFMSW fraction for AD. The optimisation of the siting process takes into account local aspects such as exclusion areas (*i.a.* water bodies, areas with slopes exceeding 10%), preferential areas (*i.a.* brownfield and derelict sites), distances to inhabited areas and other sensitive receptors (*i.a.* schools, hospitals, recreational sites).

3.2 Choice of a case study municipality and plots

The AD plant in Poznan has been planned as the extension of the existing landfill site (plot 1 in Figure 2). The landfill was opened in 1984 and its total area is of almost 49 ha including recultivated part of 11,6 ha and new, expanded part of 20,9 ha. The annual amount of deposited MSW is approx. 130.000 Mg. The plot chosen as one of possible locations of the plant by the investor is locate in the south-eastern part of the landfill within the administrative borders of the city of Poznan. The closest residential area of single-family housing is about 800 m in the Suchy Las Commune and the allotment gardens on the west. Crucial is the vicinity of the Meteorite Nature Reserve, which is famous for its seven meteor craters. Inhabitants' hopes that the nuisance associated with the existing facility would disappear after its closure were dashed. Protests were triggered, *i.a.* by the decision to cofinance the investment from the EU funds as this is an area of a high recreation, amenity and touristic value. Also the inhabitants of the neighbouring Morasko Radojewo and Suchy Las raised their objections regarding a close proximity to the planned investment.

The siting process has begun in 2010, later becoming a part of the Regional Waste Management Plan 2012. The new AD facility will be able to process 30.000 Mg of biodegradable waste, 18.000 Mg of green waste and 12.000 Mg of OMSW from households of the city of Poznan. In the process of AD followed by composting the plant will produce 2.300 MWh electricity to be used for installation self consumption and the same amount will be sold to the grid. Heat will be produced in the amount of 3.500 MWh and the compost (13.500-15.000 Mg) will be made available to the municipality (green areas of the city), inhabitants and farmers.

In order to find the most suitable location for the plant three sites will be investigated. First location (1) is situated close to the main landfill of Poznan (4) situated within the borders of the Suchy Las Commune neighbouring the city of Poznan. The second location (2) is on the lot close to the border with the Czerwonak Commune, this plot is a part of a CHP plant (6) and next to waste-to-energy plant (5), currently under construction. It is an industrial area in the north-eastern suburbs of the city with many services, warehouses and factories. However, this plot is also not very far from the Warta River and its natural banks thus under the risk of flooding. The third location (3) is close to the main wastewater treatment plant (7) located in the neighbouring Czerwonak Commune. The main obstacle in this case is its location on the edge of the Warta River and a close distance to two big residential areas built in 1980’s as a block-of-flats for workers.

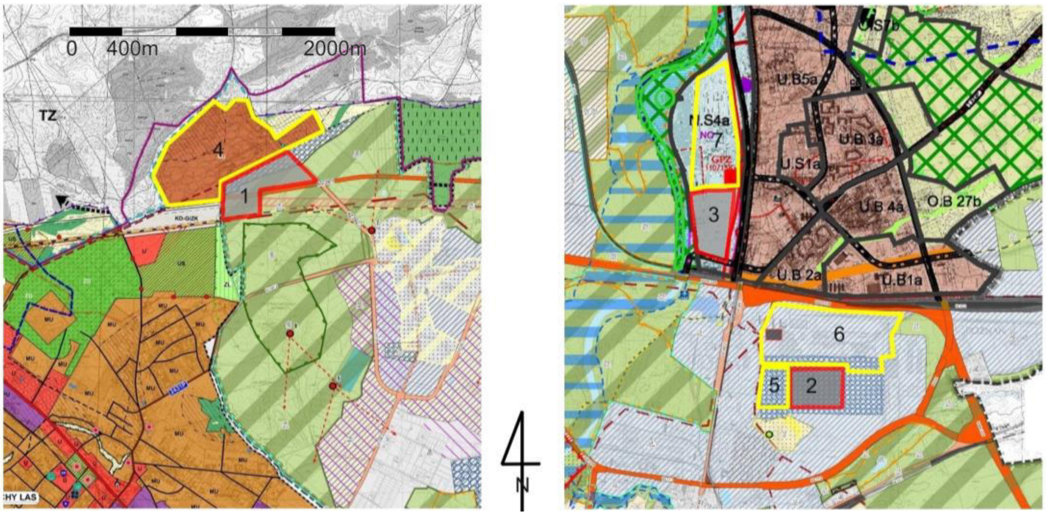

Figure 2: Part of Study of the Conditions and Directions of the Spatial Management of the Poznan City, Suchy Las and Czerwonak Communes with possible plots for the AD plant (1-3) and neighbouring infrastructure facilities (4-7).

All chosen locations are close to other waste treatment facilities, however, they are also not very distant form the residential and nature conservation areas. Certainly the use of brownfields areas or other industrial sites should be prioritised in the location selection process; however, they should not be sole criteria for plant siting. Finding a suitable location for an AD plant has been accompanied with numerous objections as regards transport logistics, noise and odours nuisance from the plant's operation. In order to improve the level of acceptance, the local authorities have started ongoing public consultations in 2014.

4. Conclusion and outlook

From the so far research it can be concluded that organizational, legal and socio-economic factors constrain the further development of such infrastructural investments in Poland. Investment procedures are also complicated, long lasting and do not encourage new developments. On top of that the very process of choosing an appropriate location is quite difficult.

Therefore, the project team elaborated a methodology for a multicriteria decision analysis tool to choose the best site for waste treatment and green energy facilities (anaerobic digestion of the organic fraction of municipal solid waste). The AHP (analytical hierarchy process) was chosen as the most suitable to create an appropriate tool. The final decision making tool should reflect different interests of inhabitants, investors and the communes and thus alleviate the effects of nimbyism. The selected case study areas are locations, which have already encountered social resistance (like Poznan) or the ones where NIMBY syndrome is very likely- like the suburban commune of Jabłonna (within the Warsaw metropol region) or Domosław (agricultural area with a high recreational value).

This is an ongoing project, complete results of which including case studies together with the sensitivity analysis of the proposed tool will be available by the end of 2015. However, the preliminary results will be presented during the conference.

Acknowledgments

The research leading to these results has received funding from the Polish-Norwegian Research Programme operated by the National Centre for Research and Development under the Norwegian Financial Mechanism 2009-2014 in the frame of Project Contract No [Pol-Nor/210329/46/2013]. URL: http://mbist.arch.pw.edu.pl/index.php/en/

References

[1] Basta C. (2012): Risk and spatial planning. *Handbook of Risk Theory*. Springer, Dordrecht.

[2] Cabala P. (2010): USING THE ANALYTIC HIERARCHY PROCESS 1 . General Characteristics of the Method. *Operations research and decisions* 1(1)

[3] Yang J. and Lee H. (1997): Academic papersFAn An AHP decision model for facility location. *Facilities* 15(9), 241–254

[4] Ying X., Zeng G.M., Chen G.Q., et al. (2007): Combining AHP with GIS in synthetic evaluation of eco-environment quality-A case study of Hunan Province, China. *Ecological Modelling* 209(2-4), 97–109

TD-O_15 Ensiling as pretreatment to improve the anaerobic biodegradability of catch crops

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1. Objectives

The main objective of this study was to determine the anaerobic biodegradability of three catch crops: *Lolium multiflorum. (Ryegrass), Brassica napus (Oilseed rape) and Avena strigosa (Black oat)*) and to assess the effect of ensiling, in order to use them as co-substrate in manure-based biogas plants while maximizing biogas production.

2. Methodology

2.1. Experimental set-up

Biochemical methane potential (BMP) tests of the three catch crops studied were carried out at 37 ºC (in duplicate) according to Soto et al. (1993) [1] and Angelidaki et al. (2009) [2]. 1.2 L-glass vials were filled with 0.5 L of a mixture of cow manure as inoculum (1.5 gVSS/L), silage and fresh catch crops as substrates (2.5 gCOD/L) as well as deionized water. The mixture was supplemented with macro/micronutrients and bicarbonate following Ferrer et al. (2010) [3]. A control vial without substrate was included to assess the residual methane (CH_4) potential of the inoculum. The flasks were stirred and bubbled with a N_2/CO_2 gas mixture (80/20 v/v) in order to remove O_2 before they were closed with rubber stoppers. The test lasted an average of 40 days. Anaerobic biodegradability and methanogenic index (M) of each sample were calculated as described elsewhere [1].

2.2. Analytical methods

Total solids (TS), volatile solids (VS), total chemical oxygen demand (COD) and total kjeldahl nitrogen (TKN) concentrations were measured according to Standard Methods [4]. Biogas production was measured with a volumetric milligas counter. Biogas composition (CH_4, H_2, and CO_2) was determined by gas chromatography once a week. pH was measured on the leachate after crop:distilled water extraction (1:5 v/v) for 0.5 h.

2.3. Feedstocks

The inoculum was taken from an anaerobic lab-scale digester treating cow manure working at steady state. Catch crops were grown and harvested between successive plantings of maize at the Mas Badia Field Station (La Tallada d'Empordà, Girona, Spain) and ensiling was performed immediately after harvesting at the laboratory during almost three months. The main characteristics of the feedstock used are summarized in Table 1.

The fermentation process was confirmed by the decrease of pH values after silage process, although the rest of parameters did not show significant differences, except for COD whose differences may be related to an analytical error. Even though the pH of silage catch crops was low, the initial pH in each vial was around the neutrality due to the 1:5 v/v dilution and the buffer capacity of cow manure.

Table 1: Characterization of the feedstock used.

	pH	COD (g kg^{-1})	TS (%)	VS (%)	TKN (g kg^{-1})	Biomass yield (t DM Ha^{-1})
, CO2 is emitted	6.35	227	20	18	4.5	2.69
Ryegrass silaged	4.01	292	21	18	4.5	-
Oilseed rape	5.72	270	13	11	3.5	3.14
Oilseed rape silaged	3.98	270	13	11	3.6	-
Black oat	6.29	346	17	15	4.6	3.03
Black oat silaged	3.67	307	17	16	4.8	-

COD-Chemical Oxygen Demand; TS- Total solids; VS-Volatile solids; TKN- Total kjeldahl Nitrogen; DM-dry matter.

3. Results

The anaerobic biodegradability obtained for the three catch crops studied and the cumulative methane (mL of CH_4) production in all the BMP assays carried out are shown in Figure 1 and Figure 2, respectively. As can be seen in Figure 1, it appears that ensiling improved the biodegradability and methane yield of catch crops under anaerobic conditions. In fact, the anaerobic biodegradability of ryegrass, oilseed rape and black oat was increased by 5%, 16% and 17%, respectively, which resulted in an increase of the accumulated volume of methane by 9%, 38% and 70%.

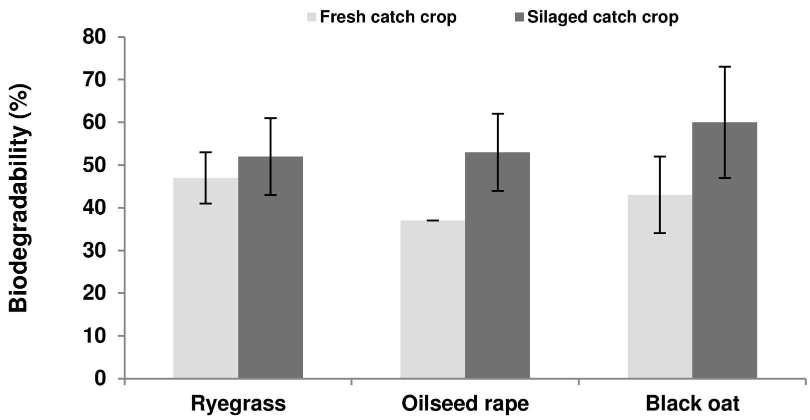

Figure 1: Anaerobic biodegradability for fresh (blue) and silage (green) catch crops.

Table 2 summarizes the methane yields obtained in the BMP assays. In terms of $L_{CH4}kg_{VS}^{-1}$. Methane yields of silage catch crops increased by 36%, 41% and 44%, for ryegrass, oilseed rape and black oat, respectively, whereas methane yield expressed as $L_{CH4}kg_{COD}^{-1}$ was improved by 10%, 43% and 40%, respectively. The obtained values (see Table 2) were within the range of those values obtained in previous studies for different catch crops, ranging from 250 to 450 $L_{CH4}kg_{SV}^{-1}$ [5]. In any case, regarding anaerobic digestion of crops, factors such as temperature, mixing, particle size, organic load, lignocellulosic content, part of the plant utilized for the anaerobic digestion and crop storage strategy, should be considered in order to optimize methane potential.

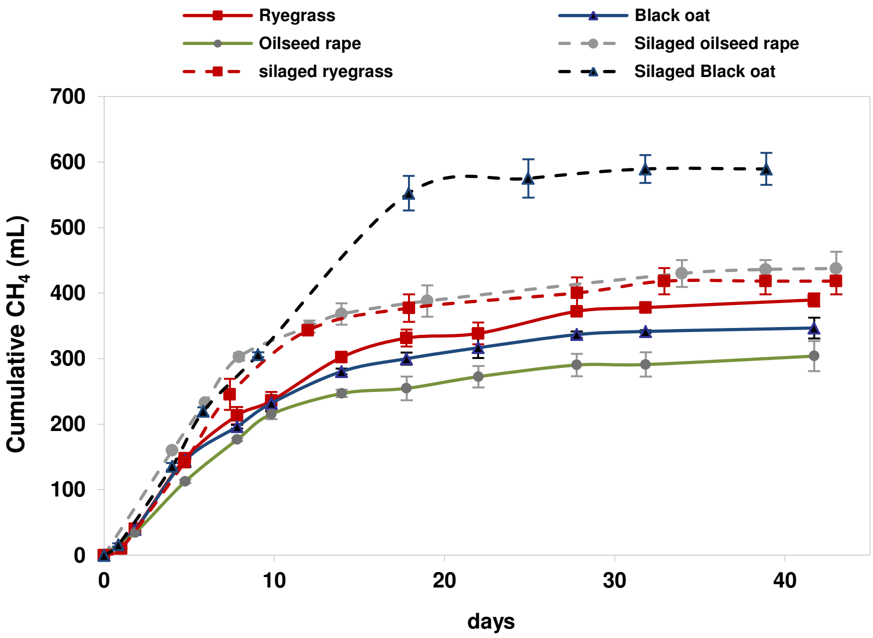

Figure 2: Cumulative methane production for fresh (continuous line) and silage (dotted line) catch crops.

During the silage process lactic acid, acetic acid, methanol, alcohols, formic acid, H^+ and CO_2 are formed. These products are important precursors for methane formation [6]. Another reason for the increase in specific methane yield could be a predecomposition of crude fibre in course of the silage process, which improves the biodegradability and the availability of nutrients for the methanogenic metabolism. In any case, analysis of lignin, cellulose and hemicellulose is necessary in order to a better understanding of methane yield production from different crops. Note that, in spite of the higher amount of cumulative methane achieved by silage black oat, methane yield was only slightly higher than silage oilseed rape or even lower in terms of $L_{CH4}kg_{COD}^{-1}$. Also methane yield per hectare was lesser than those obtained for silage oilseed rape (Table 1).

Table 2: Methane yields and anaerobic biodegradability of silage and fresh catch crops obtained in the BMP tests.

(*)	Ryegrass	Silaged ryegrass	Oilseed rape	Silaged oilseed rape	Black oat	Silaged black oat
Methanation potential (LCH$_4$ kgVS$_{added}^{-1}$)	195±2	264±15	301±8	424±27	271±12	391±17
Methanation potential (LCH$_4$ kgCOD^{-1})	152±2	167±10	120±3	172±11	139±6	194±9
Methanation potential (m^{3}CH$_4$ t^{-1}fresh matter)	35±0	49±3	32±1	47±3	46±2	60±3
Methanogenic potential (%M)	43±5	48±8	34±0	49±8	40±9	56±13
Anaerobic biodegradability (%B)	47±5	52±9	37±0	53±9	43±9	60±14
m^3 CH$_4$ ha^{-1}	468	634	813	1145	732	1056

* mean values±standard deviation

Nevertheless, the main parameter to consider when using crops as substrate for biogas production is the volume of methane per hectare. Methane yield per hectare is dependent on the biomass yield per hectare and the specific methane yield of the biomass. The biomass yield per hectare depends primarily on the time of establishment, the weather conditions (temperature and rainfall) during the growing season, the amount of available nitrogen (or mineral nitrogen added) in the soil and the time of harvest. The catch crops used in this study were grown and harvested between successive plantings of maize with a minimum cultivation and harvesting efforts, without using fertilizers and watering.

Economic calculations made for ryegrass showed that its methane yield per hectare should be above 700 m^3 CH$_4$ ha^{-1} (specific methane yield above 350 m^3 t^{-1} of VS and biomass yield on VS above 2 t ha^{-1}) to obtain an economic favourable balance when using this crop as a co-substrate for manure-based biogas plants [2].

In our study, biomass yield on VS for ryegrass was 2.4 t_{VS} ha^{-1}. As can be seen in Table 2, specific methane yield of fresh ryegrass was almost 195 m^3 t^{-1} of VS and methane yield was therefore around 468 m^3 CH_4 ha^{-1}. Although is quite lower in comparison with 700 m^3 CH_4 ha^{-1}, this methane yield is high taking into account the efforts of cultivating this catch crop in economic terms and the fact that methane yield in terms of m^3 t^{-1} of VS was almost half of 350 m^3 t^{-1} of VS . On the other hand, when ryegrass ensiling was performed, methane yield per hectare increased by 35% (634+ m^3CH_4 ha^{-1}). Additionally, the specific methane yields of oilseed rape and black oat increased by 41% and 44%, respectively. Besides the economic assessment, performing a life cycle analysis of the process is essential in order to determine the impact of the overall activity on the environment, including nutrient balances.

In spite of the fact that the oilseed rape biodegradability was very similar to those obtained for ryegrass, and 7% lower than the biodegradability of black oat, its higher biomass production per hectare, resulted in the highest CH_4 production per ha (1145 $m^3{}_{CH4}$/ha for the silaged oilseed rape). However, it should be tested in continuous test to optimize the operational parameters (% mixture, organic load rate, etc.) to maximize the biogas production.

4. Conclusions and outlook

Ensiling can be use as strategy to store as well as to increase the anaerobic biodegradability of vegetable substrates such as catch crops. Thus, it can be used as an economically feasible pretreatment in order to use them as co-substrates in manure-based biogas plants while maximizing biogas production. The anaerobic biodegradability was improved by 5-17% whereas methane yield per hectare of silage crop was 30-47% higher than those obtained for fresh catch crops. Silage oil seed rape appears to be the best option for producing biogas by anaerobic digestion due to its higher methane yield per hectare of crop (1145 $m^3CH_4ha^{-1}$).

Next step will be to optimize operational conditions of the anaerobic co-digestion of manure and these catch crops at continuous mode in order to elucidate the possible synergisms and the viability of applying the process at industrial scale in a full-scale biogas plant.

Acknowledgements

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References

[1] Soto, M., Méndez, R., Lema, J., 1993. Methanogenic and non-methanogenic activity tests. Theoretical basis and experimental set up. Water Research 27 (8), 1361–1376.

[2] Angelidaki, I., Alves, M., Bolzonella, D., Borzacconi, L., Campos, L., Guwy, A., Jenicek, P., Kalyuzhnui, S., van Lier, J., 2009. Defining the biomethane potential (BMP) of solid organic wastes and energy crops: a proposed protocol for batch assays. Water Sci. Technol. 59 (5), 927–934.

[3] Ferrer, I., Palatsi, J., Campos, E., Flotats, X., 2010. Mesophilic and thermophilic anaerobic biodegradability of water hyacinth pre-treated at 80 ºC. Waste Management 30, 1763–1767.

[4] APHA, 2005. Standard Methods for the Examination of Water and Wastewater, twentyfirst ed. Washington DC, American Public Health Association. American Water Works Association and Water Environment Federation.

[5] Molinuevo-Salces, B., Larsen, S.U., Ahring, B.K., Uellendahl. H. (2013). Biogas production from catch crops: Evaluation of biomass yield and methane potential of catch crops in organic crop rotations. *Biomass and Bioenergy* **59**, 285-292.

[6] Amon T., Amon B., Kryvoruchko V., Zollitsch W., Mayer K., Gruber L. (2007) Biogas production from maize and dairy cattle manure-Influence of biomass composition on the methane yield. Agriculture, Ecosystems & Environment 118:173-182.

TD-O_16 Anaerobic co-digestion of pig manure and organic waste materials as affected by different hydraulic retention time

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1. Objectives

The aim of the study was to study transformation of lignocellulosic biopolymers and biogas production yield effected by hydraulic retention time (HRT) on co-digestion of pig manure and organic waste materials in continuous stirred tank reactors (CSTR).The results were used to develop a kinetic model that predicts biogas production along with retention time.

2. Methodology

2.1 Feedstock and inoculum

Pig manure and slaughterhouse waste were obtained from Fangel biogas plant (Odense, Denmark). Brewery waste was collected from a local brewery plant (Denmark) and supermarket waste (Danish ham) was purchased from supermarket. The organic waste material is composed of 45% Danish ham, 45% brewery waste water and 10% slaughterhouse waste. The feed for CSTR consists of 75% pig manure and 25% organic waste material based on wet mass. Final composition for feed is 75% pig manure, 11.25% Danish ham, 11.25% brewery waste and 2.5% slaughterhouse of total mass. Inoculum was collected from an industrial biogas plant on Funen Island, which processes 75 % of animal manure as prime feedstock and 25 % of industrial food processing waste by mass base at a mesophilic temperature.

2.2 Anaeorbic digestion in batch and continuous mode

Batch tests to determine Biochemical methane potential (BMP) were carried out according to VDI 4630 (2006). Anaeorbic digestion in continuous mode using 20 L of laboratory scale CSTR was performed at four different HRT, 15, 20, 30 and 45 days at 37 °C. The organic loading rates were 3.88, 2.91, 1.94 and 1.29 g.l^{-1}day^{-1} at 15, 20, 30 and 45 days, respectively. Gas yield, gas concentration, VFA and TAN were monitored throughout the anaerobic digestion. The reactors was fed after 14 days of degassing and operated for the period of three times HRT with 12 l working volume.

2.3 Chemical characterization

The methane concentration was measured by the gas chromatograph (7890A, Agilent technology, USA) equipped with a thermal conductivity detector and a 30 m x 0.320 mm column (J&W 113-4332, Agilent technology, USA).

Dry matter (DM), VS, TKN, TAN, Crude lipid were measured according to APHA standard [1].

Volatile fatty acid (VFA) concentrations from C2-C5 were determined using a gas chromatograph (Hewlett Packard 6890, Italy) with a flame ionisation detector. Ethanol was measured using high-performance liquid chromatography (HPLC, Agilent 1100, Germany). Hemicellulose, cellulose and lignin were determined according to Van Soest characterization [2,3].

2.4 Modelling of methane production kinetic in CSTR

From mass balance equation for microbial growth in CSTR with the assumption of constant working volume, influent and effluent without any accumulation in the system, it is concluded that:

$$\mu = \frac{1}{\theta} = D \qquad \text{Eq. 1}$$

Where μ= specific microbial growth (day^{-1}), θ= hydraulic retention time and D= Dilution rate (day^{-1}).

The maximum specific microbial growth is equal to reciprocal critical HRT.

$$\mu_{max} = \frac{1}{\theta_c} \qquad \text{Eq. 2}$$

Specific microbial growth is explained by the kinetic model suggested by Grau et al. [7] below:

$$\mu = \mu_{max}\frac{S}{S_0} \qquad \text{Eq. 3}$$

Combining Eq. 1, 2 and 3 results in:

$$\frac{S}{S_0} = \frac{\theta_c}{\theta} \qquad \text{Eq. 4}$$

On the other hand, the correlation between methane yield at any time versus substrate concentration in influent and effluent is expressed in Eq. 5.

$$\frac{B}{B_{max}} = \frac{S_0 - S}{S_0} = 1 - \frac{S}{S_0} \qquad \text{Eq. 5}$$

The term S/S_0 is substituted by θ/θ_c. Then, the rate of methane production, r_{CH4}, is calculated by multiplying organic loading rate, S_0/θ, and methane yield, B.

$$B = B_{max}\left(1 - \frac{\theta_c}{\theta}\right) = B_{max}\left(1 - \frac{OLR}{OLR_c}\right) \qquad \text{Eq. 6}$$

$$r_{CH_4} = S_0 B_{max}\left(\frac{1}{\theta} - \frac{\theta_c}{\theta^2}\right) \qquad \text{Eq. 7}$$

Where B= methane yield (NL CH_4 kg^{-1} VS), B_{max}= maximum methane yield (NL CH_4 kg^{-1} VS), r_{CH4}= rate of methane production (NL CH_4 L^{-1} day^{-1}), OLR= organic loading rate (g/l day) and OLR_c= critical organic loading rate (g/l day).

2.5 Degradation kinetic of lignocellulosic fractions

The first order kinetics is used to assess the hemicellulose and cellulose degradation in CSTR. Hydrolysis kinetics is used to evaluate biodegradation of slowly degradable fractions including hemicellulose and cellulose in CSTR.

$$r_S = \frac{dS}{dt} = -kS \qquad \text{Eq. 8}$$

Where r_s= degradation rate of component (g l^{-1} day^{-1}), k= hydrolysis kinetic constant (day^{-1}) and S= component concentration (g l^{-1}). Using mass balance calculation for hemicellulose and cellulose combined by Eq. 8 leads to Eq. 9 explaining reciprocal S versus hydraulic retention time.

$$\frac{1}{S} = \frac{1}{S_0} + \frac{k}{S_0}\theta \qquad \text{Eq. 9}$$

Where S is hemicellulose or cellulose concentration in the reactor (g l^{-1}), S_0 is the hemicellulose or cellulose concentration in the feed (g l^{-1}).

3. Results and discussion

3.1 Substrates and digestates characteristics and methane production potentials

The physicochemical characteristics and methane potentials of substrates and digestates are demonstrated in table1. The crude lipid content in all digestates are zero meaning it is converted to long chain fatty acids (LCFA) and consequently partly to biogas. Non-fibrous carbohydrates in digestates except the one at 45 days HRT are higher than feed that might be as a result of degradation of lipids and lignocellulosic fraction to LCFA and sugars. Concentration of crude protein is much lower in all digestates compared to feed which shows its transformation to biogas and probably some nitrogen-containing polymers, amino acids and nitrogen in the structure of microorganisms. In case of lignocellulosic content in VS, it is increasing in all digestates in comparison with feed due to the fact that its degradation is slower than proteins and lipids with different hydrolysis rates leading to decline in the biodegradability of digetates along the HRT.

Table 1: Physicochemical analysis and biomethane potentials of substrates and digestates.

Sample name	Corrected VS (g/kg)	Crude proteins (%VS)	Crude lipids (%VS)	VFAs (%VS)	Ethanol (%VS)	Hemicellulose (%VS)	Cellulose (%VS)	Lignin (%VS)	Non-fibrous Carbohydrate (%VS)	TBMP (NL/kg VS)	BMP (NL/kg VS)	Biodegradabality (%)
Pig manure	30.4	21.1	0.0	23.1	0.0	15.5	24.1	16.5	0.0	474	297	62.7
Jaka bov	239.1	50.2	41.8	0.0	0.0	0.0	0.0	0.0	8.0	706	605	85.7
Slaughterhouse waste	314.2	11.9	73.2	0.8	0.0	0.0	0.0	0.0	14.1	862	821	95.2
Brewery waste	6.2	0.0	0.0	4.6	60.0	0.0	0.0	0.0	35.4	602	577	95.8
Feed	58.3	30.6	29.2	9.6	0.0	6.1	9.4	6.5	8.6	631	513	81.4
PgM 100%	17.3	18.1	0.0	0.9	0.0	12.3	22.3	20.9	25.5	494	23	4.7
PgM 75% & OWM 25% HRT=15	33.3	13.1	0.0	23.1	0.0	8.3	12.7	11.5	31.2	452	286	63.4
PgM 75% & OWM 25% HRT=20	27.5	18.4	0.0	10.8	0.0	9.2	14.4	13.4	33.8	467	210	45.0
PgM 75% & OWM 25% HRT=30	22.3	24.5	0.0	2.9	0.0	10.6	16.7	17.0	28.2	487	157	32.1
PgM 75% & OWM 25% HRT=45	19.0	28.1	0.0	0.0	0.0	10.3	19.0	20.9	21.7	503	65	13.0

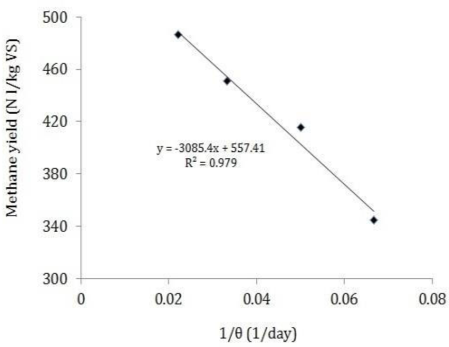
3.2 Biogas yield and CSTR performance

The methane yield for 15, 20, 30 and 45 days of HRT are 345, 416, 451 and 487 (Nl CH_4 kg^{-1} VS). Methane production and methane concentration were steady during the time course showing performance of reactors in all experiments is without inhibition. After 20 days, VFA concentration and rate of methane production are heading to level off and reach stable condition.
TAN in the digestates ranged from 3.77 to 4.02 g/kg, however gas concentration and production was sound and stable.
Using Eq. 6, the methane yield at each HRT, graphed versus 1/θ (Fig. 1, left) at which B_{max} and $B_{max}\theta_c$ are intercept and slope of the linear equation. It was found that methane yield versus reciprocal HRT had a fine inverse linear correlation (R^2 0.979) and intercept was 557 (Nl CH_4 kg^{-1} VS) which is maximum methane yield. Thus, the methane yield and rate of methane production could be expressed as below.

$$B = 557 * \left(1 - \frac{5.5}{\theta}\right) = 557 * \left(1 - \frac{OLR}{10.5}\right) \qquad \text{Eq. 10}$$

$$r_{CH_4} = 32.5 * \left(\frac{1}{\theta} - \frac{5.5}{\theta^2}\right) \qquad \text{Eq. 11}$$

Where B= methane yield (NL CH_4 kg^{-1} VS), θ= HRT, OLR= organic loading rate (g/l day) and r_{CH4}= rate of methane production (NL CH_4 L^{-1} day^{-1}).
Using the Eq. 10 and 11, biogas production was simulated in Fig. 1 right, showing the highest rate of methane production at HRT of 11 days (twice critical HRT) calculated by putting derivative of Eq. 11 equal to zero. In addition, the R^2 of predicted values against actual ones is 0.993 expressing the good predictability of model for gas production.



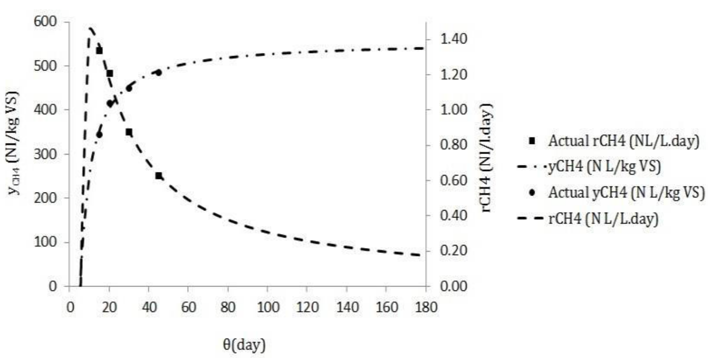


Figure 1: Left: Methane yield plotted versus reciprocal hydraulic retention time. Right: Methane yield and rate of methane production affected by hydraulic retention time from the model.

3.3 Degradation kinetic of lignocellulosic material

Hemicellulose concentrations in feed and digestates are 3.54, 2.78, 2.54, 2.38 and 1.96 and cellulose concentrations are 5.49, 4.23, 3.96, 3.72 and 3.60 respectively. Both fractions are
The 1/S is plotted versus HRT (Fig. 2, left) to estimate degradation kinetic constant k from Eq. 9 which is hydrolysis rate constant of cellulose and hemicellulose.
Using linear regression, *k* is estimated of 0.0204 and 0.0116 (day^{-1}) for hemicellulose and cellulose respectively, showing the degradation of hemicellulose is around twice than that of cellulose. The reason could be the structure of hemicellulose is more amorphous compared to cellulose which is partly crystalline [6].
Hemicellulose and cellulose are relatively easily biodegradedable, however if they are trapped within the lignocellulosic matrix, lignin acts as a protector from their decomposition. Lignocellulosic materials are very slowly degradable fractions if the lignin content is high [5].

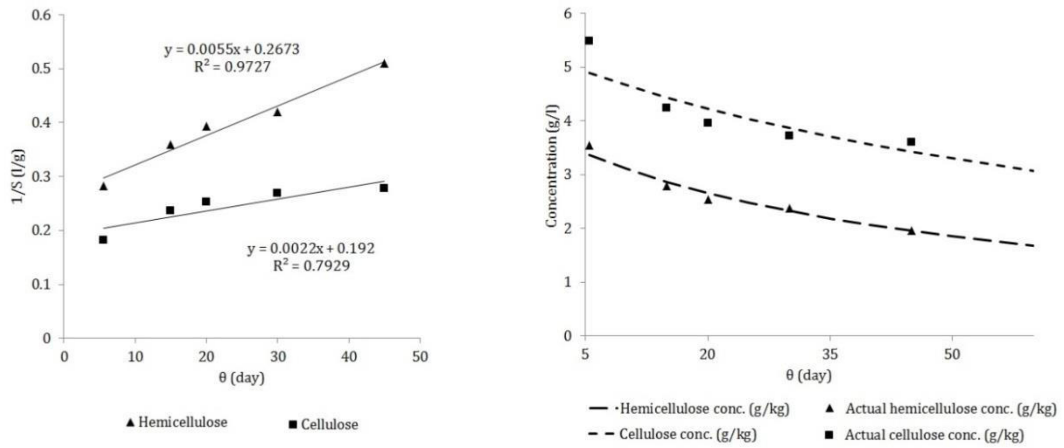


Figure 2: Left: Hemicellulose and cellulose concentration plotted against HRT. Right: Hemicellulose and cellulose concentration predicted by model and actual results.

4. Conclusion and outlook

The R^2 of predicted values against actual ones is 0.993 expressing the good agreement of model for methane production. The results showed the critical HRT at 5.5 days and maximum methane production rate at 11 days. The methane production from the feed mostly containing crude protein and lipids follows the kinetic suggested by Grau et al., while the degradation of hemicellulose and cellulose follow first order kinetic. The k values estimated for hemicellulose is around twice than that of cellulose due to the fact that hemicellulose has amorphous structure and cellulose is more crystalline.

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References

[1] APHA. Standard Methods for the Examination of Water and Wastewater, 21st. ed. American Public Health Association, Washington, DC 2005.
[2] Van Soest PJ. Use of Detergents in the Analysis of Fibrous Feeds. II. A Rapid Method for the Determination of Fiber and Lignin. J Assoc Off Anal Chem 1963;46:829–35.
[3] Triolo JM, Sommer SG, Møller HB, Weisbjerg MR, Jiang XY. A new algorithm to characterize biodegradability of biomass during anaerobic digestion: Influence of lignin concentration on methane production potential. Bioresour Technol 2011;102:9395–402. doi:10.1016/j.biortech.2011.07.026.
[4] Pavlostathis SG, Giraldo-Gomez E. Kinetics of Anaerobic Treatment 2011.
[5] Lübken M, Gehring T, Wichern M. Microbiological fermentation of lignocellulosic biomass: Current state and prospects of mathematical modeling. Appl Microbiol Biotechnol 2010;85:1643–52. doi:10.1007/s00253-009-2365-1.
[6] Zheng Y, Zhao J, Xu F, Li Y. Pretreatment of lignocellulosic biomass for enhanced biogas production. Prog Energy Combust Sci 2014. doi:10.1016/j.pecs.2014.01.001.

TD-O_17 Promoting pig slurry low-cost methanization by understanding the microbiology of anaerobic digestion at low-temperature

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1. Objectives

Psychrophilic methanisation is used all over the world as a simple, robust and cheap way to produce green energy from animal waste. One limitation is the long time required for microbial communities to adapt to strong temperature changes, especially for low temperatures. To better understand microbial community adaptation, we applied several strategies to test the impact of temperature on biogas production and methanogen communities of swine slurry:

- selection of inocula adapted to 13°C (annual mean temperature in temperate climate)
- brutal shift of the communities from 35°C to 5°C

2. Methodology

Three swine slurries (fresh (FSM) and stored for 2 (SSM) or 9 months (LTSSM)), 1 fresh cow manure (BM) and 1 mesophilic swine slurry digestate (DSM) were tested as inoculum for methanisation of swine slurry at 13°C (Table 1). 0.5l from each manure was put in 3 batch reactors made of 2.2l bottles and flushed with CO_2 to facilitate methanisation start-up. Once a week, 75 ml of fresh swine manure (FSM, homogenized and stored frozen) was added to simulate the filling of a storage tank. The initial organic loading rate (OLR) was about 1.2 g COD/l.d. The added load was increased to 1.6 and 2.4g COD on days 104 and 118 respectively. For each bottle, methane production was monitored continuously using an Automatic Methane Potential Test System (Bioprocess Control AB, Lund, Sweden) [1]. Monthly, all bottles were sampled to measure pH (APHA, 1998), total and soluble COD (micro-method, MERCK), VFA concentrations [2] and for microbial community analyses.
Biomass from the best acclimated bottle was then used for BMP-like tests incubated at 35, 25, 15 and 5°C to study the impact of brutal temperature changes on methanogenesis. Comparison was done in the same conditions with a mesophilic biomass from a laboratory anaerobic digester treating swine manure for about 1 year. BMP were done without agitation in 150ml glass bottles filled with 110ml of liquid composed of ¼ of inoculum and ¾ of fresh swine manure supplemented with 3.45g of co-substrate (Sodiva EG Poney), leaving 40ml headspace for gas accumulation.Gas production and methane concentration were measured using a pressure meter device and a gas chromatograph [3]. Biomass samples were taken from each bottle at the beginning, the active phase and the plateau phase of biogas production for microbial community analyses.
Bacterial and archaeal communities' dynamics was monitored by 16S rDNA qPCR and CE-SSCP fingerprinting as described in [4] and [2] respectively.

3. Results and discussion

3.1 Selection of a microbial community adapted to 13°C

Before incubation, the bovine manure (BM), long term stored swine manure (LTSSM) and swine slurry digestate (DSM) inocula were concentrated by centrifugation (4000g for 20 min at 13°C) to adjust their volatile solids content to the one of the fresh swine manure (FSM) and stored swine manure (SSM). It resulted in inocula with relatively similar pH, total solids and total COD (Table 1). However, the FSM and DSM inocula differed from the others by their high concentrations in soluble COD, volatile fatty acids and ammonia and their low proportion of *Archaea* (putative methanogen) in their microbial community. Interestingly, the comparison of values of the 3 swine manure suggests that the proportion of *Archaea* in the microbial community increases with the duration of storage in the farm, FSM < SSM < LTSSM.

Average CH_4 production of bottles inoculated with each of the 4 manure and digestate is shown in Figure 1. Methane production was not detected for the bottle inoculated with the digestate (DSM). For the manure bottles, a slight latency period occurred from days 7 to 13. Then,for the first 50 days, the production increased regularly and was slightly better for the bottles inoculated with the long term stored swine manure (LTSSM) or the bovine manure (BM). Thereafter, the methane production rate of the bottle inoculated with the stored swine manure (SSM) overpassed the one of all the other bottles. At the end of the experiment, the methane production rate of each bottle was (l/day): SSM (22.3) >LTSSM (15) > BM (11.7) > FSM (10). Methane production from the fresh manure inoculum (FSM) was the less important in comparison to other ones. It could be linked to its high VFA and ammonia initial content.

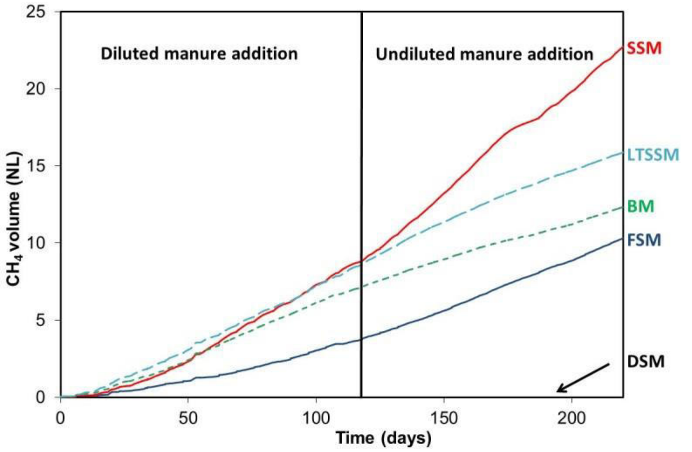

Figure 1: Average methane production (l/day) during anaerobic digestion of fresh swine manure in laboratory reactors incubated at 13°C and inoculated by five different manures: FSM, fresh swine manure; SSM, stored swine manure; BM, bovine manure, LTSSM, long time stored swine manure and DSM, digestate of swine manure. Vertical bar represents the OLR change on day 118.(Variabilitywithin triplicates was ± 7 %).

Table 1: Physico-chemical and microbiological characteristics of the 5 manures tested as inoculum.

Inoculum	pH	TSS	VSS	Total COD	Soluble COD	VFA	N-NH4	Bacteria	Archaea	Archaea/ Total[b]
		g/l			g O2/l		g/l	Gene copies/l[a]		%
FSM	7.4	76.3	58	109	31	18.6	4.9	$3\ 10^{10}$	$3\ 10^{8}$	1.0
SSM	7.4	83.1	55	100	6.6	0	1.5	$4.6\ 10^{11}$	$1.4\ 10^{10}$	2.9
BM	7.3	59	51	102	12.3	1.4	1	$6.2\ 10^{11}$	$2\ 10^{10}$	3.1
LTSSM	7.7	82	58.6	85	4.1	0.1	0.9	$4.9\ 10^{11}$	$3.9\ 10^{10}$	7.4
DSM	7.5	83	53	131	34.6	15.3	7.9	$1.2\ 10^{12}$	$9\ 10^{8}$	0.1

[a] Copies of 16S rRNA gene/l of manure; [b] Calculated as: (copies of archaeal 16S rRNA gene/ (copies of bacterial 16S rRNA gene + copies of archaeal 16S rRNA gene)) x 100

Table2: Physico-chemical and microbiological characteristics of the five bottles after 222 days of acclimation.

Inoculum	pH	TSS	VSS	Total COD	Soluble COD	VFA	CH4	N-NH4	Bacteria	Archaea	Archaea/ Total[b]
		g/l			g O2/l		%	g/l	Gene copies/l[a]		%
FSM	7.9	42.0	30.3	82.0	31.2	9.3	69	4.	$1.7\ 10^{10}$	$5.5\ 10^{7}$	0.3
SSM	7.9	42.8	29.3	71.8	11.7	0.2	69	3.5	$5.1\ 10^{10}$	$3.5\ 10^{9}$	6.4
BM	7.6	41.0	29.2	92.6	18.5	7.7	60	3.5	$2.2\ 10^{11}$	$2.6\ 10^{9}$	1.2
LTSSM	7.7	41.5	29.1	93.3	18.9	5	64	3.5	$2.7\ 10^{11}$	$5.4\ 10^{9}$	1.9
DSM	8.0	36.2	23.9	88.3	29.3	9.5	18	4.5	$3.6\ 10^{11}$	$1.2\ 10^{9}$	0.3

[a,b] As in Table 1.

After 222 days of acclimation at 13°C (Table 2), and despite similar TSS and VSS values, only the SSM bottle did not accumulate VFA which is in agreement with its biogas production. The soluble COD and VFA levels were relatively high for the FSM and DSM bottles as compared to others. Interestingly, the SSM bottle is also the only one which supported an increase of the *Archaea*/total microorganisms ratio. Moreover, except for the DSM bottle, this ratio is in agreement with the level of methane production observed in each bottle (Figure 1).

3.2 Adaptation of microbial communities to temperature change

The SSM methanogen microbial community adapted to 13°C in the above experiment (named now "psychrophilic inoculum") was used to study methane production at 35, 25, 15 and 5°C (Figure 2). One volume of this community was mixed with 3 volumes of fresh swine manure supplemented with a co-substrate to increase methane production. A mesophilic inoculum was managed in the same way as control. At 35°C both inocula demonstrated comparable biogas production kinetics showing that the psychrophilic inoculum did not lose its ability to produce methane at 35°C. At 25°C, the psychrophilic inoculum performed almost as good as at 35°C while the mesophilic inoculum showed a 40 days period of adaptation and lower total biogas production. At 15°C, only the psychrophilic inoculum produced a significant amount of biogas after an adaptation time of about 50 days. At this temperature, biogas production from the mesophilic inoculum broke down after about 20 days. At 5°C both inocula showed very slow kinetics and low overall biogas production. These results show that the psychrophilic inoculum was actually able to produce biogas between 15 and 35°C.

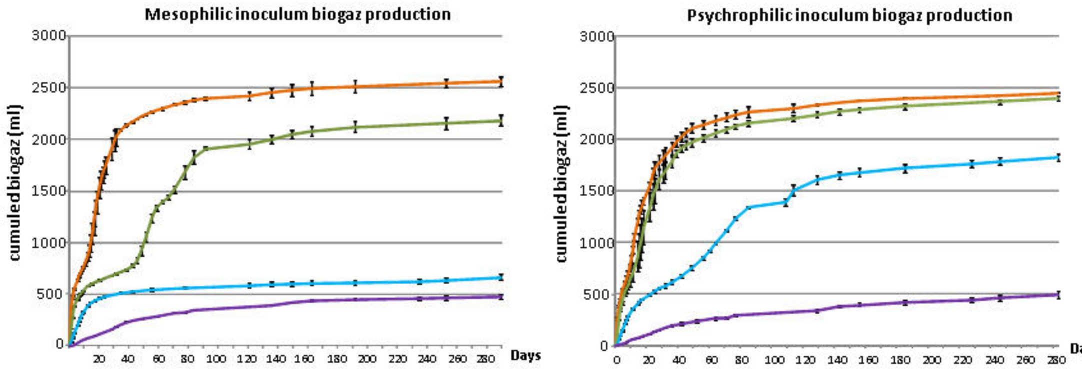

Figure 2: Cumulative biogas production of the mesophilic and psychrophilic inocula incubated at 35 (orange), 25 (green), 15 (blue) or 5 (purple) °C.

The methane content of the biogas reached more than 75% after 20 days of incubation for both inocula at 35°C and for the psychrophilic inoculum incubated at 25°C (not shown). An adaptation period of about 8 and 13 weeks was required for the mesophilic inoculum at 25°C and the psychrophilic inoculum at 15°C, respectively. At this last temperature, the methane content in biogas from the mesophilic inoculum varied between 20% and 30% only, confirming that methanisation broke down. At 5°C, the methane content in biogas from both inocula remained below 25%.

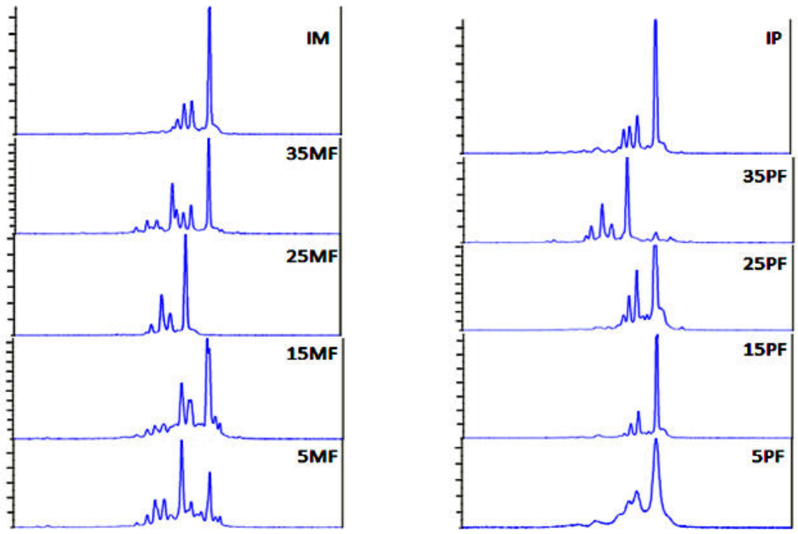

Figure 3: Alignment of the archaeal CE-SSCP fingerprints from the mesophilic and psychrophilic inocula at the beginning of incubation (IM and IP respectively) with those of communities present at the end of the incubations at 35, 25, 15 and 5°C. Nomenclature: the first two numbers are the temperature, M and I are the mesophilic or psychrophilic inoculum respectively and F is the ultimate time sampled.

The 16SrDNA CE-SSCP fingerprints of each inoculum showed different archaeal communities between the initial mesophilic and psychrophilic inocula and the fresh swine manure used as substrate, but they all showed a low diversity with at most 6 dominant peaks (not shown). The fresh swine manure archaeal community was the most diverse with a balanced diversity of 4 dominant peaks representing each between 10 to 20% of the total profile. The archaeal community of the mesophilic and psychrophilic initial inocula were dominated by only one peak (figure 3) that differed in each inoculum but both belonged to the acetotrophic methanogens *Methanosarcinales* group.

Alignment of the archaeal CE-SSCP fingerprints (Figure 3) showed that:

- The *Methanosarcinales* archaeal population that dominated the mesophilic inoculum (IM) did not adapt to lower temperature. It was replaced by another population at 25°C, in agreement with the lag phase observed during biogas production in Figure 2. This population belongs to a hydrogenotrophic methanogen from the *Methanobacteriales* group.
- On the opposite, the *Methanosarcinales* archaeal population that dominated the psychrophilic inoculum (IP) could adapt to higher temperature to some extent. It remained dominant at 25 and 15°C and was replaced by another population at 35°C.
- Communities with different archaeal composition (35MF, 35PF, 25PF Figure 3)can have similar methane productions (Figure 2). In this case, running the process at 25°C with an adapted microbial community and having the same biogas production may represent an energy saving of 10°C.
- At low temperatures that do not allow biogas production (15 and 5°C for IM; and 5°C for IP) the dominant populations remain present but inactive and seem to be slowly overgrown by a higher diversity.

4. Conclusion and outlook

Using different manure inocula and a simple selection procedure we were able to adapt a microbial community to anaerobic digestion of swine manure at low temperature (13°C) within 200 days. The results show that the regular storage time of manure in farms is too short to obtain maximal methane production. It also shows that the choice of the inoculum and of the OLR is important parameters. The best methane producing inoculum was coming from swine manure stored in a farm storage tank for about 2 months. After adaptation, the inoculum produced 125 l CH_4/kg COD added, reaching 68% of the substrate BMP. At the end of the experiment, methane production was at 42 l CH_4/kg $VS_{substrate}$·day after a 9 months period of acclimation.

This adapted microbial community was able to produce methane at 15°C but also (and even more) at 35 and 25°C. In comparison, the mesophilic community used as control could not adapt to produce methane à 15°C. The analysis of the archaeal community dynamics to temperature changes suggests that both microorganism metabolic adaptation and populations shifts within the community where involved in acclimation. It also demonstrates that different archaeal community structures can support similar methane production yield and kinetics. It finally suggests that some archaeal species are better adapted to either warm or cold temperatures.

Because of its ability to produce methane at temperatures above 20°C, our acclimated microbial community is actually a mesophilic community adapted to low temperature and not a true psychrophilic community. It would be interesting to determine how long the community can keep this ability since it will allow a better flexibility in case of inoculation of real processes.

Acknowledgements

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References

[1] Badshah, M.; Lam, D.M.; Liu, J.; Mattiasson B. (2012): Use of an Automatic Methane Potential Test System for evaluating the biomethane potential of sugarcane bagasse after different treatments, Bioresource technology 114, 262-269

[2] Peu, P.; Brugère, H.; Pourcher, A. M.; Kérourédan, M.; Godon, J. J.; Delgenès, J. P.; Dabert, P. (2006): Dynamics of a pig slurry microbial community during anaerobic storage and management, Applied and Environmental Microbiology 72(5), 3578-3585

[3] Vedrenne, F.; Béline, F.; Dabert, P.; Bernet, N. (2008): The effect of incubation conditions on the laboratory measurement of methane producing capacity of livestock wastes, Bioresource Technology 99, 146-155

[4] Marti, R.; Dabert, P.; Pourcher, A.M. (2009): Pig Manure Contamination Marker Selection Based on the Influence of Biological Treatment on the Dominant Fecal Microbial Groups, Applied and Environmental Microbiology 75, 4967-4974

TD-O_18 Interstage treatment for increasing methane production from recalcitrant biomass

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1. Objective

Our objective was to increase the conversion efficiency of recalcitrant biomass to methane during anaerobic treatment through non-chemical inter-stage treatment. This paper will discuss two biological inter-stage treatments we have tested on wastewater sludge:

- Inter-stage thermophilic aerobic digestion (TAD),
- Inter-stage TAD followed by a non-aerated thermophilic incubation.

2. Methodology

Substrate was a mix of primary sludge and secondary dewatered sludge for experiment 1, and was secondary dewatered sludge for experiment 2. Figure 1 summarizes the process. Methodology was as follow:

- First stage mesophilic anaerobic digestion was carried out at 37°C. The reactor was full-scale for experiment 1 (hydraulic retention time (HRT) 28-30 d) and laboratory stirred batch for experiment 2 (inoculum to substrate (I:S) ratio was 0.5 based on wet mass, HRT 26 d).
- Inter-stage TAD was applied to waste water sludge using a heated (55°C) and aerated stirred tank reactor. Air supply was 0.25 L per L reacting mass per minute. Inoculation was done using an inoculum to substrate (I:S) ratio of 0.25 based on wet mass. The batch reactor was run for 1 day. Effluent was sampled at different time points (from 30 min to 1 day) and anaerobically digested afterwards. COD in TAD effluent was measured directly after sampling.
- Non-aerated thermophilic incubation was applied to TAD effluent in sealed tubes incubated in a water bath (55°C) for 1 h to 2 days (for some of the samples).
- Second stage anaerobic digestion was mesophilic in batch mode. Digestion was started at the same time for all samples in each experiment; if needed, samples were kept at 3°C for ≤ 2 days. Gas volume was quantified by syringe and gas composition was determined by gas chromatography. Samples were inoculated with an I:S ratio of 2.5 based on COD. Inoculum, adapted to the substrate, came from full-scale digester. Digestion time was 20 days.

Data analysis was done using the biogas package [1].

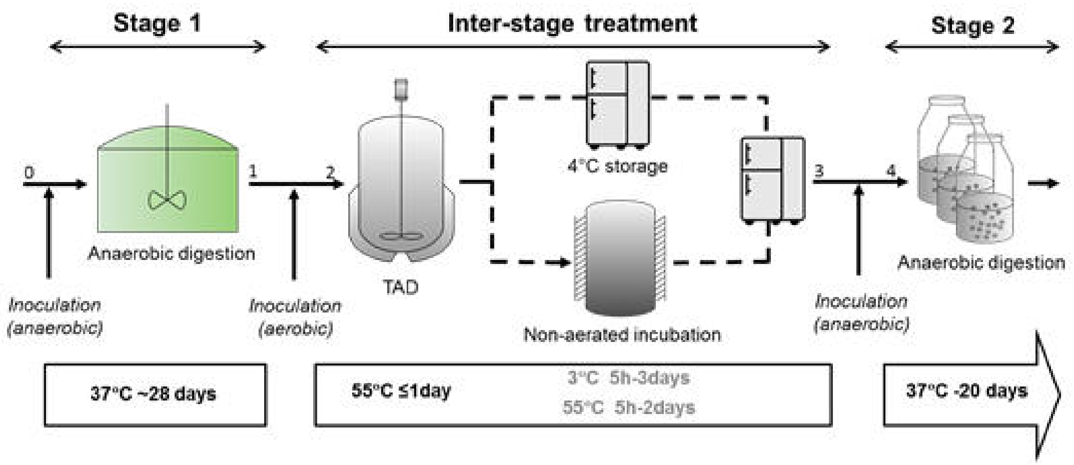

Figure 1: Overview of the studied process. Numbers refer to the effluent at different points of the process: 0, initial substrate; 1, effluent after stage 1 (digestate from initial substrate and inoculum); 2, effluent after stage 1 mixed with TAD inoculum; 3, effluent after inter-stage treatment (effluent after stage 1, mixed with TAD inoculum and treated by inter-stage treatment); 4, effluent after inter-stage treatment mixed with anaerobic inoculum. In experiment 1, data collection started from position 1. In experiment 2, data collection started at position 0 but analysis presented in this paper starts at position 1. Stage 1 was different for the two experiments, see Methodology section for details.

3. Results and discussion

3.1 Thermophilic aerobic digestion as an inter-stage treatment for anaerobic digestion

The sampling times in TAD were based on the dissolved oxygen concentration trend in the reactor, which starts near saturation, drops after adding substrate, then levels off at a lower concentration, and finally increases slowly until reaching saturation again. This pattern in oxygen concentration reflects changes in oxygen consumption rate due to substrate addition and depletion. Samples from each phase of the process and the beginning (0 h) and end (24 h) of the inter-stage treatment were analyzed for COD.
When designing the experiments, we were interested in three questions: (1) Is TAD effective for oxidizing COD remaining after anaerobic digestion? (2) Can an inter-stage TAD treatment increase the methane production of a subsequent anaerobic digestion? (3) Does the duration of TAD influence the results?

COD destruction by TAD: COD (of effluent from position 1 in figure1) was reduced by 18.6 ± 1.6% (mean ± standard error) in experiment 1 and 6.41 ± 0.76% in experiment 2 (Figure 2), this assumes no degradation of TAD inoculum. Initial values were 33.6 ± 0.33 g kg-1 for experiment 1 and 34.3 ± 0.25 g kg-1 for experiment 2. If degradation of TAD inoculum is the same as the effluent, COD was reduced by 17.2 ± 0.9% in experiment 1 and only by 6.82 ± 0.77% in experiment 2 (COD of effluent in position 2 on Figure 1, data not shown). These two approaches give similar results, implying that in the proportion used, TAD inoculum does not have a strong influence on the outcome. The lowest dissolved oxygen concentration was found in experiment 1 where it dropped to 4.22 mg L-1 (at 55.3°C) after 5 h (saturation is 5.07 mg L-1 at 55°C [2]). A minimum of 5.39 mg L-1 (56.1°C) was found for experiment 2 after 1.12 h (data not shown). From the oxygen consumption we can deduce that there was an active population of aerobes and that the digestate from experiment 2 was more recalcitrant than digestate from experiment 1. This may be due to the inclusion of primary sludge (more degradable) in the full scale reactor and/or the variation in polymer quantity used to dewater the secondary sludge, since polymers are easily degradable. This difference was also apparent in the COD destruction from stage 1 digestate (shown in Figure 2). COD loss in TAD is fast; within the two first hours of the process, one-third of the total COD destroyed in TAD was lost in experiment 1, while almost two-thirds were destroyed in experiment 2 (Figure 2).

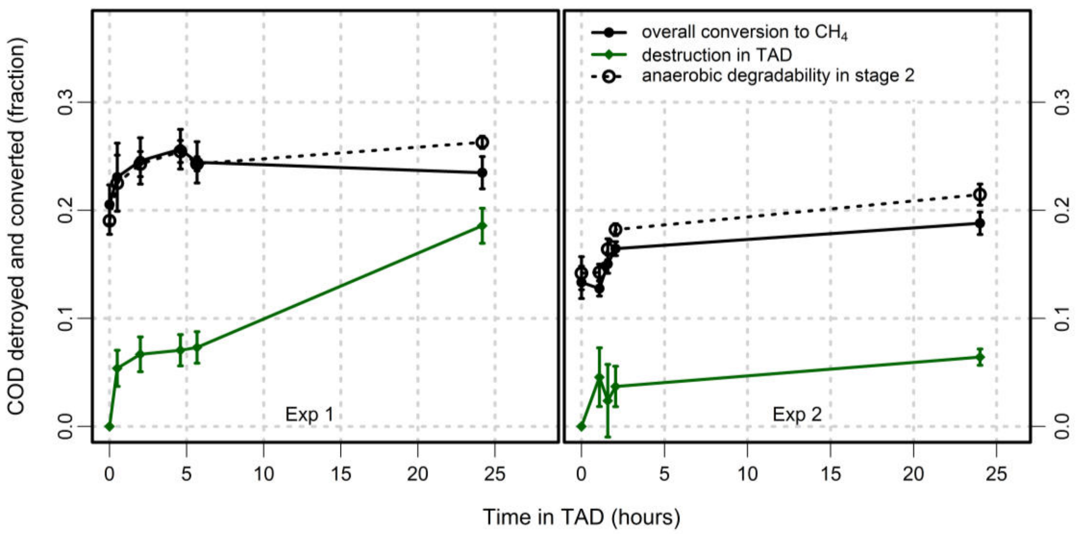

Figure 10: Fate of COD during inter-stage thermophilic aerobic digestion (TAD) and subsequent anaerobic digestion (stage 2) as a function of retention time in TAD. Substrate was digestate from wastewater sludge (effluent at position 1 in Figure 1). Untreated substrates (control conditions) are the points at time = 0. Solid lines present the overall effect of the treatment: results are normalized to the mass of COD of substrate present at the end of stage 1 (be fore aerobic inoculation, position 1 in Figure 1). Dotted lines represent the anaerobic degradability of the effluent in the second anaerobic digestion: results are normalized to the mass of COD present at the beginning of stage (position 3 in Figure 1). These normalizations assumed anaerobic inoculum from stage 1 and effluent from stage 1 are equally degraded during the process. Contribution to the biogas volume of the anaerobic inoculum, used for stage 2, was measured in separate reactors and subtracted. Methane production was evaluated for 20 days. COD lost in TAD was based on COD measurements. COD conversion to methane was calculated from methane production assuming 350 ml of methane corresponds to 1 g COD [3].

TAD to increase methane production: In both experiments, an increase in overall methane production due to TAD was observed for some of the time points in comparison to the control (Figure 2). The maximum increase in conversion of the original substrate (at position 1 in Figure 1) during TAD and second stage anaerobic digestion was around 25% in experiment 1 and, 40% in experiment 2 (25.0 ± 2.2 % after 4.6 hours in TAD for experiment 1 and 41.0 ± 5.7% after 24 hours for experiment 2). In both experiments an increase in anaerobic degradability during stage 2 was found after 24 h in TAD, where methane production was increased by more than one-third in experiment 1 and by a half in experiment 2 as compared to the control (experiment 1: 93.0 ± 3.3 vs 66.6 ± 7.5 mL CH4 g COD-1; experiment 2: 75.1 ± 5.9 vs 49.6 ± 9.2 mL CH4 g COD-1). Anaerobic degradability evaluates the conversion of the substrate as it enters the digester at stage 2 (position 3 in Figure 1). It does not account for the effect of the treatment as opposed to the overall methane production. An increase in overall methane production means that anaerobic degradability at stage 2 has increased more than the losses of carbon occurring during the TAD (Figure 2).

Influence of TAD retention time on methane production: The retention time of TAD strongly influenced methane production. However the pattern is not totally consistent between the experiments. Conversion to methane was improved with the time in TAD for all samples in experiment 1; but it declined (until 2 hours in TAD) and then increased for samples in experiment 2. Optimum time in TAD, to increase COD conversion to methane was found around 4.5 hours for experiment 1 while it was 1 day in experiment 2. The effects of TAD were greater in the early stages of the anaerobic digestion (≤10 days). More experiments will be needed to understand this phenomenon.

3.2 Non-aerated thermophilic incubation combined to TAD as an inter-stage treatment

After the TAD and before the second stage anaerobic digestion, a non-aerated incubation was applied to samples from experiment 1 and 2. The objective of this thermophilic incubation was to provide additional time for the enzymes produced by TAD bacteria to continue breaking the organic matter while avoiding oxidation of the organic matter.
Our work addressed three questions related to this incubation: (1) Does the incubation after TAD have an effect on methane production? (2) Is the incubation effect influenced by the duration of TAD? (3) Does the incubation time influence methane production?

Effect of non-aerated incubation on methane production: We found positive and significant effects of the thermophilic non-aerated incubation following the TAD inter-stage treatment for all time points in both experiments, see Figure 3. Addition of non-aerated incubation improved methane produced in the second stage anaerobic digestion by 47.0 ± 3.9 % in experiment 1 and by a maximum of 60.0 ± 10.2 % in experiment 2 (relative to controls with no TAD treatment, position 2 in Figure1). Paired t-test between the two groups of samples non-incubated and samples incubated indicated that the incubation significantly increase methane production (P = 9.72×10-11). This positive effect could be explained by the presence of extracellular enzymes produced by thermophilic aerobes to break down organic matter. These enzymes would mainly be protease and amylase as found by Hasegawa et al. [4]. As oxygen is not provided, enzymes continue to break down the carbon chains but aerobes are not able to continue the degradation process. In this way the degradability is increased but the complete oxidation is avoided and the reduced substrate can be used for methane production in stage 2.

Influence of TAD retention time on non-aerated incubation effect: The non-aerated incubation effect was independent of the TAD retention time (Figure 3). All our tested samples were inoculated with aerobic inoculum. This suggests that as long as there is TAD inoculum (or possibly extracellular enzymes) in the mix incubated, the anaerobic degradability can be increased by a non-aerated thermophilic incubation. Greater amounts of organic matter can be converted to methane when both TAD and incubation effects are combined (i.e., when the substrate is treated by TAD and not just inoculated). Significant effect of the incubated samples over non treated (0 h in TAD – position 2 in Figure1) was found for a confidence level 0.05 using a two-factor ANOVA with interactions.

Influence of non-aerated incubation time: The influence of the incubation time was determined for one time point in experiment 2 (1.6 h in TAD). Four durations were tested: 1 h, 5 h, 1 day and 2 days. In early stages, until 10 days of anaerobic digestion, results (not shown) indicated that, the longer the incubation, the greater the increase in organic matter conversion to methane. However, this general trend is less marked after 20 days of anaerobic digestion (Figure 3). Incubation time positively influences the early conversion rates of organic matter to methane; the longer the incubation (tested over 2 days), the faster the conversion to methane in the early stages of anaerobic digestion (≤10 days).

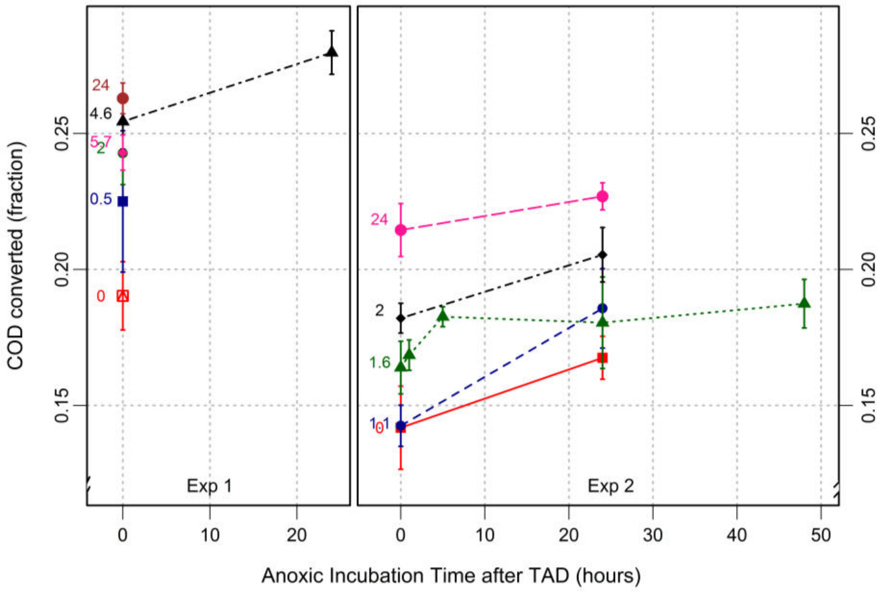

Figure 11: Effect of non-aerated thermophilic incubation of TAD effluent on anaerobic degradability (stage 2). Numbers in the figure refer to the retention time in TAD. COD converted represents the anaerobic degradability during stage 2. COD converted is normalized to the initial substrate present at the start of stage 2 (position 3 in Figure 1). Contribution of the anaerobic inoculum to the biogas volume was measured in separate reactors and subtracted. COD conversion to methane was calculated from methane production assuming 1 g COD corresponds to 350 ml of methane [3].

4. Conclusion and outlook

Inter-stage treatment of municipal waste water sludge by TAD improved methane production of the second stage anaerobic digestion by more than 20% in both experiments. Retention time in TAD had an influence on the methane produced in the second stage digestion. However, as different optimum retention times were found in the two experiments, it is not clear yet which retention time of TAD is the optimal. We suggest that this optimal time depends on the substrate used and its degradability. More work is needed to understand the phenomenon. The best results are obtained when a non-aerated thermophilic incubation follows the TAD treatment. In the best scenario, by coupling the effects, methane produced during the second stage anaerobic digestion could be increased by half. We attribute the increase in methane production after the non-aerated incubation mainly to faster degradation rates during the first 10 days of anaerobic digestion. More work is needed to evaluate the effects of these inter-stage treatments on overall methane production.

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References

[1] S. D. Hafner and C. Rennuit, *biogas: Analyze Biogas Data and Predict Biogas Production*. 2015.
[2] D. L. Parkhurst and C. A. J. Appelo, *Description of input and examples for PHREEQC version 3–A computer program for speciation, batch-reaction, one-dimensional transport, and inverse geochemical calculations*, vol. 6. U.S. Geological Survey, 2013.
[3] B. E. Rittmann and P. L. McCarty, *Environmental Biotechnology: Principles and Applications*. Boston: McGraw-Hill, 2001.
[4] Hasegawa S, Shiota N, Katsura K, and Akashi A, “Solubilization of organic sludge by thermophilic aerobic bacteria as a pretreatment for anaerobic digestion,” *Water Sci. Techology*, vol. 41, no. 3, pp. 163–169, 2000.

TD-O_19 Performance of fully scale anaerobig digestion plant to achieve quality fertilizer for apple cultivation and vine-growing use

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1. Objectives

The combination of anaerobic digestion and composting for the treatment of biowaste in one centralized industrial plant with a high level of technology permits the double exploitation of this renewable biomass: on one side the biogas produced is used for heat and power generation (CHP); on the other one the organic matter can be recovered for agronomical use In this paper the main results deriving from two years of monitoring of the biological processes are exposed with the following aims:

- to define the parameters and related values useful to characterize the process;
- to verify the biological stability of compost before final use;
- to promote the system with an adequate level of information at local level.

2. Case study description

The waste plant considered is located in Trentino province, alpine region of Northern Italy. It started to work in November 2012 and it has a total capacity of 25,750 tons/y of biowaste, which corresponds to about 50% of the total amount produced, deriving from the collection of source selected organic waste. Selective collection of solid waste is mandatory in Trentino since 2003 and the quality of the organic fraction increased gradually in the years due to a constant activity of information spread all over the territory. This determines a low level (< 3%) of undesirable components in the organic waste entering the process.

The dry anaerobic digestion (AD) is carried out in 2 horizontal mixed reactors (D1 and D2) for 25 days at 51°C. The digestate is sent to the subsequent aerobic composting after addition with yard waste to balance the C/N ratio of the starting mixture. The biogas produced feeds 1 MW engine for CHP; 12,000 tons/y of good quality compost (D.Lgs 75/2010) are sold to local farmers.

A plan for the operational management and monitoring of the plant was elaborated with the local authority, focused on the optimization of energy exploitation and the biological stability of the organic matter during the aerobic composting, prerequisite for the agronomical use of the final product.

3. Results and discussion

3.1 Anaerobic digestion and composting

A regular and constant monitoring of the two biological processes involved (anaerobic and aerobic) aims to ensure the proper management and stability through *(a)* the creation of optimal conditions for the growth of microorganisms and *(b)* the identification of inhibitory or toxic phenomena. Moreover the control of physical, chemical and biological parameters allows *(a)* to avoid possible unrecoverable blocks, *(b)* to optimize biogas yields and *(c)* to ensure the quality of the final products, biogas and compost. Therefore some "key" parameters were identified and used [1], by defining the range of values specific for this plant. Table 1 shows some mean values 2014 of the parameters used to monitor the AD process in both the reactors.

The AD under dry conditions is obtained with a feeding mixture of biowaste and green waste in order to have the total solid (TS) about 40.04 %, 73.41 % organic matter (OM). The organic load is about 25,816.95 tons/year; 13,056.71 tons/year to D1 and 12,760.24 tons/year to D2, which, for D1, corresponds to 941 – 1213 tons/month or 35.8 ± 2.46 tons/day (annual mean value).

The mean values of FOS/TAC (Ratio between Organic Acid and Total Inorganic Carbonates) of the substrate were defined as 0.45 in input and 0.34 in output; this range indicates a good feeding of the system, which is confirmed by the specific amount of biogas produced due to the quick answer of the microbial community. The ratio FOS/TAC was measured in input and output of each reactor (Fig 1a), in order to check the proper biodegradation of fresh organic matter. D1 and D2 show the same behaviour. Changes in the range of values of this parameter reflect problems related to imbalance of the organic load and the functionality of microorganisms. The integration of FOS/TAC analysis with the detection of volatile fatty acids, specifically propionic, butyric and acetic, is important to better follow the activity of each specific trophic group (data not shown).

Table 1: Analytical parameters of AD process. The values reported are the mean of all data 2014. Values expressed on fresh matter (FM), dry matter (DM) and organic matter (OM) content.

Parameter	Unit	Mean Value D1	Mean Value D2
		AD process	
Organic load	ton / day	35.08 ± 2.46	34.89 ± 2.59
Ammonia (out)	mg N-NH_4/ L	4642.97 ± 260.42	4722.17 ± 276.42
FOS/TAC (out)	-	0.372 ± 0.07	0.39 ± 0.08
pH	-		
Biogas Production	Nm^3	2,234,721.52 ± 13,675.76	2,210,697.24 ± 13,754.30
Specific Biogas Production	Nm^3 / ton FM	171.35 ± 8.55	173.52 ± 9.17
Methane content	% CH_4	58.20	58.20
Specific Biomethane Production	Nm^3 / ton FM	99.72 ± 4.99	100.99 ± 5.49
Reduction of OM	% OM	44.69	44.92
Digestate			
Water content	WC / % DM	75.37	
pH	pH	8.5	
CES	µS / cm	4020	
Organic matter	OM / % DM	58.43	

The specific production of biogas was calculated by the total amount of biogas produced and the organic loading (Fig. 1b). The mean value is about 170 Nm^3/ton of FM loaded. The annual trend depends of several factors and reflects the good balance of different populations of Bacteria and Archaea in response to the organic matter addition. It is possible yet to note the impact of seasonal variations in the biowaste compositions, mainly summer and winter season, on the plant performance (Fig 1a-b).

The methane content in the biogas is quite stable throughout the year (58.2 % as mean annual value). Therefore in 2014 the plant produced about 1,300,959.99 (mean value) Nm^3 CH_4 from D1 and 1,287,002.43 Nm^3 from D2; the biomethane amount is equal to 100 Nm^3/ton of fresh matter (FM).

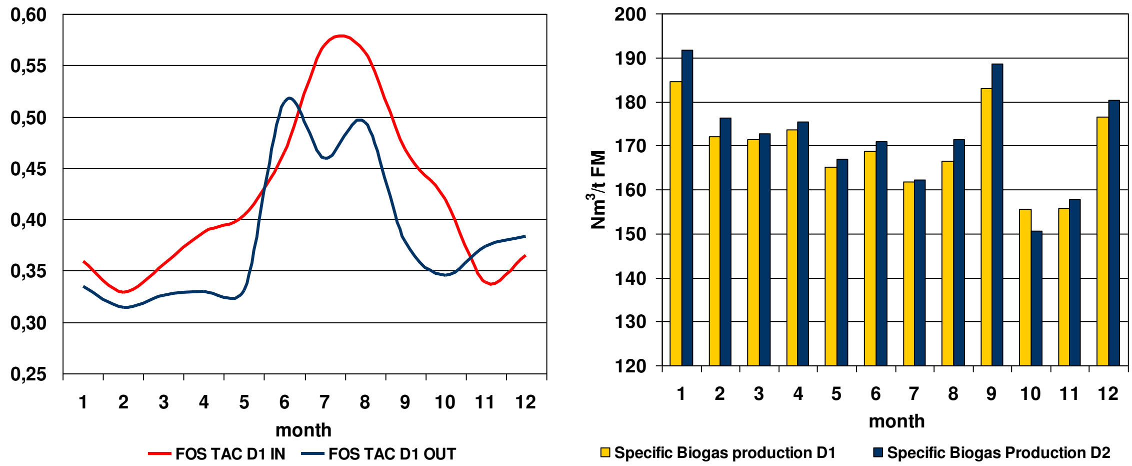

Figure 1: Annual trend (2014) of FOS/TAC (a) and Specific biogas production (b) of the two digesters. The input and output values of FOS/TAC of reactor one (D1) are reported; the trend of D2 is very similar. For the specific production of biogas the trend of both the digesters (D1 and D2) is reported.

The biodegradation of organic matter is good. The final reduction of VS is about 44 % both in D1 and D2 and reflects a good transformation of biowaste. The digestate obtained has a higher water content and a lower content of organic carbon (Table 1). The residual biomethane potential (BMP) of digestate was calculated through a laboratory test. The result, 20 Nm^3/ton FM of biogas, corre-

sponding to the 11.8 % of the overall BMP of the starting material, shows that the OM of digestate is very-well transformed.
The subsequent aerobic process on digestate needs another addition of green waste in order to balance the C/N to favour the microbial activity [2]. The composting process divides in a first bioxidation phase in biocell followed by a curing phase in static pile (forced ventilation). Each phase lasts about 12-14 days. After the final sieving (10 mm) the compost is ready for pickup upon purchase by farmers. The evolution of the aerobic process is checked at the end of each phase through measurement of pH, water content and organic matter content and respiration intensity. A dedicated software at plant level permits to control temperature, moisture and air fluxes in each biocell as well as in the curing section.

Compost quality: Table 2 shows the parameters stated by the Italian law for soil conditioners and the mean values of the compost produced in the first quarter 2015. Agronomical indicators as humic and fulvic carbon, macro and micro-nutrients certify the good quality of the product. On the other side the low content in heavy metals and undesirable particles (plastics and inerts) demonstrate the origin of the biomass and the effectiveness of source selection. Finally the microbiological indicators of pathogens reassure the users about the hygienization gained during the process. Furthermore the high level of biological stabilization of the compost produced is supported by other parameters as the respiration index which is determined at the end of the first bioxidation phase and the end of maturation (mean value 432 $mgO_2\ kgVS^{-1}\ h^{-1}$). Moreover the phytotoxicity tests, particularly the germination test, are used to assess compost maturity. When germination index > 70% the compost is considered free of phytotoxic effects on plants.

Table 2: Analytical quality of the compost produced with respect to the Italian law (D.lgs 75/2010).

Parameter	Acronym / unit	Guideline	Mean Value*	Limits
Water content	WC / % DM	UNI 10780:1998	43.1	50
pH	pH	ANPA Man 3:2001	8.6	6 – 8.8
Salinity	meq / 100g	UNI 10780:1998	31	
Organic Carbon	OC / % DM	UNI 10780:1998	23.6	20
Organic Nitrogen	ON / % DM	ANPA Man 3:2001	90.5	80
Total N	TN / % DM	UNI 10780:1998	2.0	
Carbon / Nitrogen	C / N	ANPA Man 3:2001	11.8	25
Humic + fulvic carbon	% DM	ANPA Man 3:2001	9.2	7
Sodium	% DM	UNI 10780:1998	0.5	
Phosphorus	P2O5 / % DM	D.M. 13/09/99 GU n.248	1.8	
Potassium	K2O /% DM	UNI 10780:1998	1.3	
Cadmium	Cd / mg/kg DM	UNI 10780:1998 + EPA 6020 A 2007	0.7	1.5
Copper	Cu / mg/kg DM	"	70.0	230
Lead	Pb / mg/kg DM	"	35.3	140
Mercury	Hg / mg/kg DM	"	0.2	1,5
Chromium VI	Cr / mg/kg DM	"	< 0.1	0.5
Nichel	Ni / mg/kg DM	"	9.0	100
Zinc	Zn / mg/kg DM	"	132.7	500
E.coli	CFU / g	CEN/TR 15214-1:2006	< 10	
Salmonella spp	P / A in 50 g	APAT 3 Man 20:2003	0	0
Germination index	IG %	UNI 10780:1998	90.5	> 60
Respiration index	mg O_2 / kg VS *h	UNI / TS 11184:2006	432 (1)	700**

* values are mean of the analyses of first quarter 2015
** limit given to the considered plant by local authority
(1) mean value of all the samples 2014

3.2 Compost application in orchards and vine growing

Fruit growing in Trentino is characterized by apple cultivation and wine-growing, both for the production of high quality fruit for fresh consumption and wine making. For a long time the inorganic fertilization was the only way to maintain soil fertility in fruit production systems, often due to a lack of manure too. Today the model of a more sustainable agriculture is asking for a better use of natural resources, a high level of environmental protection and the recovery of by-products residuing from agriculture and urban areas. The replacement of organic matter losses

by application of stable manure and compost is receiving even more attention in the last years, as possible solution to complex phenomena which are interesting the intensively cultivated soils. The efforts exerted at local level in realizing this “best practice” by involving at different level all the stakeholders, are giving good results up to the last ring of the chain. The local farmers are, in fact, the main users of the compost produced in the considered plant, as result of an intensive campaign of information and sensitization carried out at demonstration level in different farms and fields.
In the past years FEM carried out many field experiments to test the effects of compost in apple cultivation and vine-growing [3]. The prevailing use is as mulching, especially because of the high salinity due to food waste. The suggested dose is 20-30 ton/ha distributed along the row of fruit trees, 5 cm thick and 50 cm wide. Previous works demonstrated the positive effect of mulches in providing weed control, maintaining a better moisture level under the soil surface, reducing erosion by increase of water infiltration, improving physical properties and reducing the temperature variations (between night and day) of the soil under the row.
Recent experimental tests carried out using new advanced analytical methods [4] based on the detection of enzymatic activity and the functionality of microbial community by ds-DNA detection can improve our skills in the evaluation of soil biological activity, which is strictly connected to soil fertility. Currently the work in progress aims to understand the behaviour of different soil improvers (stable cow manure and compost) and organic fertilizers (slurry and digestate from AD of manure) on different typologies of soils: arable (corn) and vineyard. The purpose of the method applied is to measure at the same time the soil enzyme activity, microbial biomass, nitrogen, carbon and phosphorus available after organic fertilization [5] in different kind of soil and at different doses. The first results (data are under elaboration) indicate the rate of assimilation into the soil, the similar behaviour between compost and composted manure, which differ from the behaviour of digestate and slurry and finally the impact due to the type of soil.

4. Conclusion and outlook

The energy exploitation of biowaste by means of AD is a high efficiency process but the full closure of the chain can be gained only when the organic matter transformed and recovered by means of composting can find proper use in agriculture. The monitoring of both the biological processes is basic to guarantee good and constant production of biomethane, but also high quality compost. The modern agriculture is moving towards more sustainable production systems, according to a better use of natural resources and the recycling of residues produced by agriculture itself at by urban society. The increased attention to the soil in general as ecosystem necessary to life, and to the maintenance of soil fertility as far as concerns agriculture opens new possibilities for renewable sources as biomass. Future work will be addressed to better understand the role of compost or organic fertilizer on soil microbial ecology and to assess the impact on soil fertility restoration over time, also thanks to the application of new analysis tool. The market and use of the compost is possible even thanks to a correct and constant information of farmers and citizens on how and when it has to be applied.

References

[1] Drogs B., 2013. Process monitoring in biogas plants. Technical brochure. IEA Bioenergy. ISBN 978-1-910154-03-8 (electronic version).

[2] Silvestri S., Bona D., Cristoforetti A., Forlin L., Papurello D. 2014. Combination of dry anaerobic digestion, composting and energy exploitation of biogas for innovative utilization in a pilot plant. Proceedings of the 4th Central European Biomass Conference 2014, 15th – 18th January, Graz – Austria.

[3] Pinamonti F., Sicher L. 2001. Compost utilization in fruit production system. In Compost utilization in horticultural cropping systems. Ed. Peter J. Stoffella and Brian a. Kahn. Lewis Publishers. ISBN 1-56670-460-X. pp.177-200.

[4] Fornasier, F.; Asher, J., Ceccherini, M.T.; Tomat, E.; Pietralamellara, G.; (2014). A simplified rapid, low-cost and versatile DNA-based assessment of soil microbial biomass. Ecological Indicators, 45, 75 – 82.

[5] Fornasier, F.; Sinicco, T.; Vida, E.; Picco, D.; Vecchiet, A.; Silvestri, S.; Bona, D.; Mondini, C. (2013). Short term effects of bioenergy by-products on soil C and N mineralization and biochemical properties. In: 15th RAMIRAN International Conference June 3-5 2013, Versailles, France: 1-4. url: http://www.ramiran.net/doc13/Proceeding_2013/documents/S1.32..pdf http://hdl.handle.net/10449/23611

TD-O_20 Biogas for the future – Trade-offs between economy and climate

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1. Objectives

This study is part of a broader endeavour to determine, at the regional level, the possible role of biogas, in the perspective of current and future energy systems (2035 horizon).Through a case study for the Funen region (Denmark), it aims to identify the economic and environmental consequences of various possibilities for utilizing biogas and the dependency of these upon the substrate used for producing the biogas.

2. Methodology

Five possibilities for utilizing biogas where compared: Combined heat & power production (CHP) and two types of upgrading, namely scrubber and methanation. Furthermore, the option for utilizing the flexibility of the two types of upgrading in combination with CHP was investigated. In order to identify the framework conditions characterizing the current and future biogas production in the case study area (Funen) a 5-steps methodology has been developed, as explained below.

2.1 Mapping of Biomass Resources, Biogas plants, District Heating Plants and Gas Grid

Scenarios have been made based on an extensive mapping of the current characteristics of the island of Funen. The location, capacity and manure suppliers for existing and planned biogas plants on Funen have been mapped based on data supplied directly by the biogas plants. The location and type of manure has been mapped according to the methodology developed in [1], i.e. based on the annual fertilization accounts from 2011/2012 in which all livestock units are reported. The amount and composition of manure has been estimated based on the Danish Manure Standards [2, 3] and adjusted to correspond to the values registered by biogas plants. The location, type and amount of straw have also been mapped according to the methodology developed in [1], i.e. based on the latest figures reported under the EU "Single Payment Scheme" [4] combined with estimates for crop yield on different soil types. The location of district heating areas has been retrieved from [5] and the amount of supplied heat is based on a comprehensive mapping of energy consumption in the Region of Southern Denmark [6]. The location, capacity, fuel type & consumption, and heat production & delivery of each heat producing plant where kindly supplied by the Danish Energy Agency. The location and characteristics of the natural gas grid was supplied by the natural gas suppliers of the studied area; NGF Nature Energy and Dong.

2.2 Electricity Forecast for 2035

Based on the latest energy agreement [7], the Danish wind power capacity is expected to increase in the coming years. Accordingly, the share of fluctuating electricity production will increase. The electricity consumption, however, is not expected to change much, implying periods of time where the electricity production from wind will exceed the classic consumption. Based on forecasts supplied by the Danish Transmission System Operator (TSO), Energinet.dk, such excess will, by 2035, occur ca. 49 % of the time.

2.3 Fuel Price Inventory

Table 1 presents the prices considered for electricity, district heating, synthetic natural gas (SNG), taxes and subsidies. SNG has more or less the same composition as natural gas and can substitute natural gas. However, it is produced from biomass and is therefore not of fossil origin as conventional natural gas. The price of SNG is therefore considered to match the price of natural gas but SNG can also receive subsides due to its biomass origin.

Table 1: Fuel price inventory for 2035.

Parameter	Price	Reference
Biomass, straw	40 € ton^{-1}	Own assumption based on the current Danish straw market and expectations for the future
Electricity		
- Average	16 € GJ^{-1}	Supplied by Energinet.dk
- Surplus	13 € GJ^{-1}	
- Deficit	19 € GJ^{-1}	
District heating	13 € GJ^{-1}	Own calculation based on estimates for future heat pump and biomass boiler prices
Natural gas	9 € GJ^{-1}	Supplied by Energinet.dk
Taxes		
- Electricity (industry)	0.15 € GJ^{-1}	Current taxes
- Methane	0.15 € GJ fuel $^{-1}$	[8]
- NOx	0.68 € GJ fuel $^{-1}$	
Subsidies		
- Electricity	16 € GJ^{-1}	Current subsidies
- SNG	11 € GJ SNG $^{-1}$	[9]

2.4 Biogas Model

The data collection focused on five main parameters for biogas production and five main parameters for biogas utilization as shown in Tables 2 and 3.

Table 2: Parameters used for analyzing the production of biogas in 2035.

Parameter	Comments	Reference
	Biogas plant	
Digester	Data on type, efficiency, size and price	[10, 11]
Pre-treatment, for straw and deep litter	Data on type, efficiency, size and price	[12]
	Biomass	
Transport - Slurry - Solid biomass - Digestate	Data on capacity, mileage, and price of different trucks available for biomass transport	[12-15]
Biomass composition and gas potential	Averages have been estimated	[3, 11, 16, 17]

Table 3: Parameters used for analyzing the utilization of biogas in 2035.

Parameter	Comments	Reference
	Common	
Piping	Price data	[18]
Compressing	Price and efficiency data	[18]
	Technology specific	
Engine	Data on type, efficiency, size and price	[19]
Scrubber	Data on type, efficiency, size and price	[19]
Hydrogen production and methanation	Data on type, efficiency, size and price	[20, 21]

2.5 Life Cycle Assessment – Global Warming Potential

The life cycle assessment (LCA) was made as a carbon footprint analysis following the principles of consequential LCA, following the principles in [22]. The only differences are the marginal electricity for continuous, surplus, and deficit supply or production which in this study are based on [23]. Furthermore, the formation of ammonium in the digestate was here modelled as an increased crop yield, which translates into avoided indirect land use changes.

3. Results and Discussion

3.1 Characteristics of the Island of Funen

Figure 1 presents the distribution of dry matter (DM) in slurry, solid manure and deep litter from pigs and cattle as well as the distribution of DM in wheat straw. Other types of straw were not included in the study. The location of existing energy infrastructure is also shown.

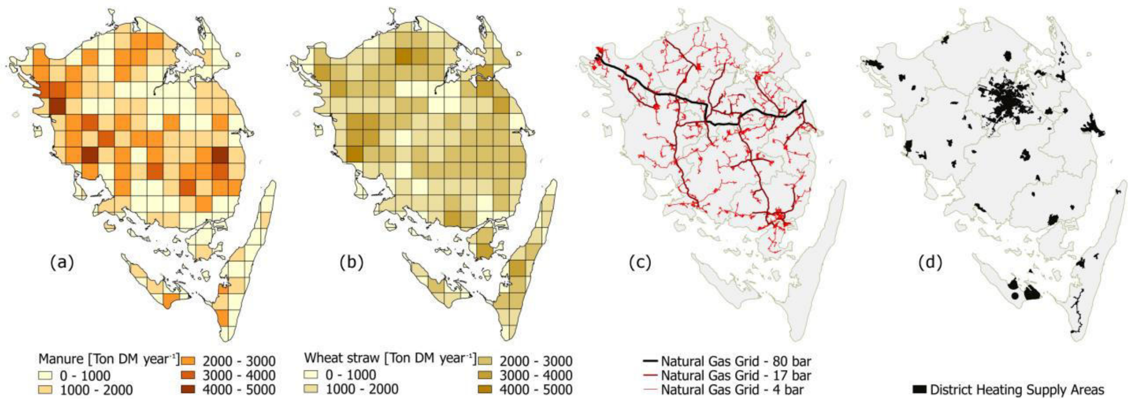

Figure 1: Distribution of (a) slurry, solid manure and deep litter from cattle and pigs (including sows); (b) wheat straw; (c) natural gas grid and (d) district heating grid on the island of Funen (Denmark)

As shown in Figure 1, some areas have more biomass available for biogas production than others. As transport of especially low energy dense biomasses such as slurry has to be kept to a minimum, biogas plants should be close to these biomasses. In order to minimize cost and energy losses, the utilization of the biogas should be situated close to either district heating areas or a natural gas grid.

3.2 Production and Utilization of Biogas

Tables 4 and 5 show the overall results of the analyses for energy production as well as the economic and environmental implications of the different setups.

Table 4: Results for biogas production: Energy, economy and GWP.

	Slurry Cattle	Slurry Pig	Slurry Sow	Solid manure Cattle	Solid manure Pig	Deep litter Cattle	Deep litter Pig	Straw	Mix used for utilization analyses
Energy [GJ net produced ton^{-1}]									
Biogas for sale	0.40	0.43	0.24	1.4	1.5	2.0	2.2	6.0	1.1
Economy [€ GJ net produced^{-1}]									
Incl. taxes & subsidies	28	24	41	12	11	12	11	10	15
Excl. taxes & subsidies	29	25	42	12	11	12	11	10	16
Climate [kg CO_2-eq. GJ net produced^{-1}]									
GWP	-210	-160	-190	-150	-160	-140	-150	-140	-150

The price for producing biogas is very dependent upon the type of biomass that is used and wet slurry is around 3 times as expensive to use compared to straw and deep litter. Taxes and subsidies do not affect the price much. There is a climate benefit from using all investigated types of agricultural wastes for biogas production. The climate impact is almost the same for all types of biomass considered but the study shows that the climate benefit is overall higher for the more wet types of biomasses due to the avoided emissions in connection with storage.

Table 5: Results for biogas utilization: Energy, economy and GWP.

	CHP	Scrubber	Methanation	Flexible, scrubber	Flexible, methanation
			Energy [GJ output GJ biogas input^{-1}]		
Electricity	0.49	-	-	0.25	0.25
District heating	0.19	-	0.048	0.10	0.10
SNG	-	0.98	1.5	0.48	0.75
			Economy [€ GJ biogas input^{-1}]		
Incl. taxes & subsidies	0	1	-9	0	-7
Excl. taxes & subsidies	-7	-9	-25	-9	-18
			Climate [kg CO_2-eq. GJ biogas input^{-1}]		
GWP	-59	-130	-170	-77	-93

All biogas uses investigated in this study result in avoided emissions of greenhouse gases. The climate benefit is much higher when the biogas is upgraded to substitute natural gas compared to when the biogas is used directly for producing heat and electricity. This is explained by the fact that biogas used in CHP, substitutes heat and electricity originating from mainly wind power production which is associated with very low emissions of greenhouse gases. Upgrading biogas with addition of hydrogen results in the largest reduction in greenhouse gases, as more fossil natural gas can be avoided when also the carbon dioxide in the biogas is converted to SNG. However, this technology also results in the largest economic deficit, mainly due to the high electricity prices.

4. Conclusion

The main findings of this work can be summarized as follows:

- The use of all types of agricultural wastes investigated in this study for biogas production result in negative greenhouse gas emissions.
- The use of biogas for both CHP and upgrade results in negative greenhouse gas emissions. The largest climate benefit is obtained by upgrading the biogas through addition of hydrogen.
- The biogas price is very dependent upon the type of biomass used for the production; from 10 to 42 € per GJ excl. taxes and subsidies. Wet biomasses are the most expensive to use.
- The utilization of biogas requires subsidies for being economically feasible. Upgrade with hydrogen results in deficit mainly due to the high electricity prices.
- Utilizing biogas flexibly for electricity generation when there is need of electricity production and for upgrade when there is a high electricity production from wind and solar power is not a benefit when considering climate and economy.

Acknowledgements

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References

[1] Birkmose T (2014): Kortlægning af biomasse til energiproduktion på Fyn, Langeland og Ærø – Husdyrgødning, halm og biomasse fra naturarealer, AgroTech
[2] Poulsen HD (2011): Normtal 2011, University of Aarhus – Department of Animal Science
[3] Poulsen HD (2011): Normtal 2013, University of Aarhus – Department of Animal Science
[4] European Commission (2009): COUNCIL REGULATION (EC) No 73/2009 of 19 January 2009
[5] Danish Ministry of the Environment (2014): PlansystemDK Geodatastyrelsen. Available: http://kort.plansystem.dk/ Accessed 05-05-2014
[6] Iversen F (2014): Energi- og CO2 balancer for 22 kommuner i Region Syddanmark, Region of Southern Denmark
[7] Danish Government (2012): Aftale om den danske energipolitik 2012-2020
[8] SKAT (2014): Den juridiske vejledning
[9] Danish Ministry of Climate, Energy and Buildings (2012): Forslag til Lov om ændring af lov om fremme af vedvarende energi, lov om elforsyning, lov om naturgasforsyning og lov om Energinet.dk
[10] Hamelin L, Wesnaes M, Wenzel H, Petersen BM (2011): Environmental Consequences of Future Biogas Technologies Based on Separated Slurry, Environ Sci Technol 45, 5869-77
[11] Madsen AS, Hansen HH (2012): Faktaark for biogas, Energinet.dk
[12] Jørgensen PJ (2013): Praktisk anvendelse af dybstrøelse som substrat på biogasanlæg - kommende som eksisterende, Danish Energy Agency
[13] DTU Transport and COWI (2014): Transportøkonomiske Enhedspriser, Transportministeriet
[14] Damsgaard M (2013): Effektiv gylletransport med friløbspumpe. Maskinbladet
[15] HV-transport A/S (2014): Gylletransport. Available: http://hv-transport.dk/index.php/gylletransport Accessed 05-05-2014

[16] Birkmose T,Hjort-Gregersen K, Stefanek K (2013):Biomasse til biogasanlæg i Danmark-på kort og langt sigt, Agro-Tech
[17] Heidman T, Nielsen J, Olesen SE, Christensen BT, Østergaard HS (2001): DJF rapport - Ændringer i indhold af kulstof og kvælstof i dyrket jord: Resultater fra Kvadratnettet 1987-1998, Markbrug
[18] Christensen AS. 2014. Personal communication with Aage Siig Christensen, Ringkøbing-Skjern Forsyning, May 2014
[19] Danish Energy Agency (2014): Technology Data for Energy Plants
[20] Haldor Topsøe (2011): Main Final Report Part 1, Energinet.dk
[21] Hansen JB, Fock F, Lindboe HH (2013): Biogas Upgrading: By Steam Electrolysis or Co-electrolysis of Biogas and Steam?, The Electrochemical Society 3089-97
[22] Hamelin L, Naroznova I, Wenzel H (2014): Environmental consequences of different carbon alternatives for in-creased manure-based biogas, Applied Energy 114,774-82
[23] Wenzel H, Høibye L, Grandal RD, Hamelin L, Bird DN, Olesen AS(2014): Carbon footprint of bioenergy pathways for the future Danish energy system, Univeristy of Southern Denmark and COWI

Thematic Area TD – The bioresource challenge (Poster presentations)

Ina Körner: Macroalgae, processed photograph

TD-P_01 Environmental assessment of the agronomical recovery of post-treated digestates

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1. Objectives

The environmental assessment of digestate (residue from anaerobic digestion) post-treatment pathways is seen from a new perspective: it is no longer a residue management but an agronomic pathway which is environmentally assessed. Furthermore, a general analysis of the post-treatment pathways is targeted. Life Cycle Assessment (LCA) is used to compare the post-treatment pathways of four types of digestates: two issued from farm digestion plants treating mainly agricultural waste, one from a collective biogas plant treating residues from various origins, and one issued from biowaste digestion.

2. Methodology

Post-treatment pathways are compared in terms of the agronomic value of the produced and spread products. In order to realize this comparison, a LCA has been performed. LCA is an environmental assessment tool which allows the quantification of potential environmental impacts of the studied system through its whole life cycle [1]. To compare and assess different systems, the LCA practitioner has to build a functional unit translating the common function between the different systems. For this study, the function is based on the fertilizing and soil improving value of the spread product (raw or post-treated digestate). The fertilizing value is based on the amount of available nitrogen for plants and the soil improving value is based on the amount of organic matter that remains to the soil. This amount is quantified as residual carbon. A functional unit based on "to provide X mass of available nitrogen and Y mass of residual carbon to the soil, from an annual production of Z mass of raw digestate" has been deployed to the various pathways studied in the project. Those are 1/ simple pathways such as raw digestate direct spreading; spreading of the solid and liquid phases and 2/ advanced pathways such as composting or drying of the solid phase, and membrane filtration of the liquid phase. To compare two pathways which do not provide the same quantities of available nitrogen and organic matter to the soil, the boundary expansion rule was applied. When comparing two systems, it was then chosen to complete the fertilizing value of the post-treatment pathway presenting a nitrogen deficit by nitrogen mineral fertilizer addition. In the same way, the soil improving value of pathway with organic matter deficit was completed by peat addition.

Most of data concerning digestate characteristics were supplied by analysis and experiments performed along the project ANR DIVA. They are relative to the matter characterization of the products, the gaseous emissions (CO_2, NH_3 and N_2O) during the composting and drying steps or following the spreading on soil, the determination of the fertilizing and soil improving values of the soil-provided products. Matter and energy data issued from four typical sites of the studied pathways in France. Literature data and databases were also used.

GaBi6 software was used for the LCA. CML-IA (version 2013) was the characterization method mainly used in this study for the quantification of the majority of the environmental impacts (CML2002 (2011 updating) was used for the resource depletion category).

3. Results and discussion

3.1 Comparison of the post-treatment pathways

Results showed that the environmental impacts of the raw digestate direct spreading pathway and the phase separation followed by solid and liquid spreading pathway (two pathways which are not so differing) are generally close. This was not true for the resource depletion and the acidification impact categories, for which the phase separation followed by solid and liquid spreading pathway was more impacting, because of the boundary expansion (addition of mineral fertilizer and peat).

Pathways with an advanced post-treatment (composting, drying and/or membrane filtration) were more impacting than pathways with a limited post-treatment regarding resource depletion, smog, toxicity and ecotoxicity. They presented the same order of magnitude regarding three concerns of anaerobic digestion: acidification, eutrophication and climate change.

3.2 Impact of the background activities

Background activities are support activities required by the post-treatment pathways, but which are not directly controllable within the pathway. Those activities, such as the production of electricity (used for phase separation, membrane filtration, air treatment) or the production of chemicals (sulfuric acid for drying and membrane filtration, polymer for centrifugation…), were responsible for the impacts on resource depletion, and explained a part of the impacts on smog, ecotoxicity and acidification.

3.3 Impact of the foreground activities

Foreground activities are leeway activities, that is to say activities which are controllable within the post-treatment pathway. Regarding smog, the impact of the foreground activities was due to the transport emissions. Regarding climate change, eutrophication, acidification, toxicity and ecotoxicity, the impacts of the foreground activities were mainly related to the subsequent spreading emissions and fate of the spread product.

Impacts of two advanced post-treatment pathways, studied in the project, are further detailed below: 1/ solid phase drying and membrane filtration of the liquid phase applied to digestate from a collective biogas plant; 2/ composting of the solid phase applied to the digestate from biowaste.

3.3.1 Example number 1 of a detailed pathway

Solid phase drying and membrane filtration of the liquid phase applied to a digestate from a collective biogas plant.
In this advanced post-treatment pathway applied to a digestate from a collective biogas plant, raw digestate is first centrifuged. Then the solid phase is dried and spread 200 km from the installation site. The liquid phase undergoes ultrafiltration, followed by reverse osmosis. The retentate and the concentrate respectively issued from those two steps are spread next to the installation site. Figure 1 shows the environmental impacts calculated for the foreground activities of this pathway.

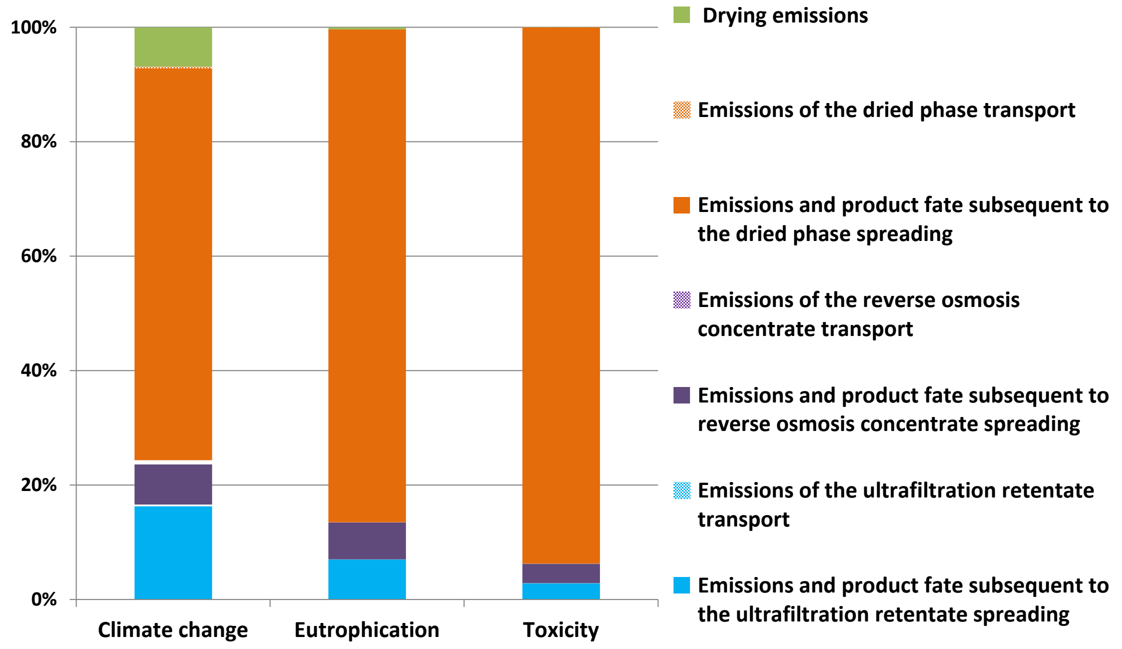

Figure 1: Relative contribution of the foreground activities to the evaluated impacts for the solid phase drying + membrane filtration of the liquid phase pathway applied to a digestate from a collective biogas plant.

As shown on Figure 1, emissions and product fate subsequent to the dried phase spreading explain most of the impacts of the assessed impacts.

Climate change is due to long term biogenic CO_2 emissions from the dried product. Those emissions are linked to the high carbon content of the dried product (437 g of C/ kg of dried matter). Even when not considering the biogenic CO_2 emissions, the conclusion is the same (i.e. preponderance of the dried product) because the N_2O volatilization potential of the dried product is high in comparison to those of the retentate and the concentrate products.

Eutrophication is mainly due to the phosphorus content of the dried phase. This is explained by the centrifugation step, where 85% of the phosphorus goes to the solid phase, while only 15% goes to the liquid phase.

Selectivity at the separation step is also true for toxicity. At this step, most of the trace metals are recovered in the solid phase, resulting in a major contribution of the dried phase spreading to toxicity.

3.3.2 Example number 2 of a detailed pathway

Solid phase composting applied to digestate from biowaste.
In the solid phase composting pathway applied to digestate from biowaste, raw digestate is first filtered, sieved and centrifuged. Then the solid phase is composted and spread 30 km from the digestion plant. The liquid phase is treated in a wastewater treatment plant, near the digestion plant. Table 1 indicates the main flows responsible for the environmental impacts of this pathway foreground activities.

Table 1: Identification of the main flows responsible for the impacts of the foreground activities for the composting pathway of digestate from biowaste.

Impact category	Contributing flows	Flow contribution to the impact	Flow contribution to the main steps of the pathway
Climate change	Biogenic CO_2	41%	Compost spreading
	Biogenic CO_2	32%	Composting
	N_2O	20%	Composting
Acidification	NH_3	83%	Composting
	NH_3	5%	Compost spreading
Eutrophication	P	85%	Compost spreading
	NH_3	8%	Composting
Toxicity	Trace metals	100%	Compost spreading
Ecotoxicity	Trace metals	100%	Compost spreading

As presented in Table 1, emissions during the composting step and emissions subsequent to the compost spreading are responsible for the impacts of the foreground activities.

Biogenic CO_2 emissions during composting and after compost spreading, such as N_2O emissions after compost spreading explain the major part of the climate change impact.

NH_3 emissions during composting cause the potential acidification emissions. This point could be improved by capturing exhaust air from the composting process and by treating it via a sulfuric acid washing, which could drastically reduce ammonia emissions. On the other hand, this would require the production of sulfuric acid.

Eutrophication is mainly due to the phosphorus content in the spread compost. The NH_3 emissions contribution during composting is limited to this impact, and could be reduced, as suggested above.

Trace metals in the spread compost explain the impacts on toxicity and ecotoxicity.

3.4 Results about the methodological implementation

3.4.1 About the functional unit implementation

Using LCA requires a rigorous implementation, which is put to the test here by being applied to digestate post-treatment management. It has been identified via these assessments that the identification of the functional unit, basis of a LCA, relies on the identification of the post-treatment function, which does not mark a consensus between stakeholders. Indeed, the post-treatment interest depends on local conditions. It could be as various as producing a fertilizer, producing a storable or spreadable product, exporting a product, producing a product easy to handle, or producing a product which is spreadable at different periods of the year. Thus, in order to apply LCA for digestate post-treatment assessment, the objective of the planned post-treatment should first be clearly discussed. This implies that results could strongly differ between the post-treatment of two digestates which have similar technical characteristics and are managed by similar technologies, but for which the context differs. A first conclusion here is that, based on the assessments of the DIVA project, the environmental evaluation of post-treatments via LCA cannot be generalized.

3.4.2 Influence of the boundary expansion rule

The study showed a very limited influence of the boundary expansion in the comparison of the post-treatment pathways (addition of mineral fertilizer or peat when it is necessary according to the functional unit). This influence is only noticeable for resource depletion and acidification in the case of agricultural digestate, and for acidification and eutrophication in the case of a digestate from a collective biogas plant. This is explained by the fact that expanding the boundaries requires the production and application of a consumable, a nitrogen mineral fertilizer for which the subsequent NH_3 and N_2O emissions after application have been considered.

4. Conclusion and outlook

Results showed that the environmental impacts of a simple post-treatment (phase separation followed by spreading of the solid and liquid phases) are close to those of a direct spreading pathway. Nevertheless, if the choice of an advanced post-treatment cannot be motivated by environmental defenses regarding resource depletion, smog or ecotoxicity, it can be argued about climate change, eutrophication and acidification.

More investigations should be done to better reflect the environmental benefits of post-treatment, both from the application point of view and from the research point of view in LCA. More reflection on the post-treatment interest should change the results and change the system boundary definition. Improving the consideration of the agronomic value of the spread products should also improve the LCA results. More site-specific assessments could permit to improve the assessments, by considering the specific needs of the cultures to grow, such as the characteristics of the spreading sites. On the LCA point of view, it is expected that the development of spatial and temporal differentiation will improve further assessments, by taking into account the local specificities of the spreading.

Acknowledgements

This study has been managed through a research project funded by the ANR (French Research National Agency) within its “Bioenergy 2010” program. Thanks are also due to digestion plants partners and colleagues who provided samples and data along the project.

References

[1] ISO 14040, 2006, Environmental management - Life cycle assessment - Principles and framework

TD-P_02 High-tech water-and nutrient-recycling – The blackwater-loop

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Objectives

This paper presents the results of the operation of a full-scale pilot plant of the black water loop process that is operated at the main railway station in Hamburg since 2013.
The system is maximally conserving our renewable water resources. Potential recyclables are almost fully recovered: by-products of the process are compost, two fertilizer raw materials and optionally biogas.

Methodology

The operation of the pilot-plant is monitored by several online sensors (liquid-levels, water-flows, air-flows, pH, electric conductivity, pressures, temperatures, energy consumption). Also the quality of the water in the different stages is measured (TS, VS, TN, TOC). The data are analyzed and discussed. Investment and maintenance costs for operation are calculated and compared.

Results

The black water loop process can be operated stable and safely and thus presents a robust technology for wastewater-treatment. The quality of the processed water for flushing is clear and not to distinguish visually from tap-water, so the toilet-users do not realize a difference to normal flush-toilets.
With the integration of a new solid-liquid-separator (patent pending) the safe running of the plant is further enhanced.
Specially adapted microorganisms in the high-performance biological fixed-bed reactor reach efficiencies for nitrification at extremely low ph-values below 5 not reported in literature before.
High energy-demand is a draw-back, but here optimization is possible. In certain conditions where water-prices are high the black water loop is economically viable.

Conclusion

The Black water loop is a technology that enables the reuse of the water and nutrients. It is one step towards the complete water-autarky of a building. With the integration of the “grey water loop process” (loop processing of greywater via a groundwater passage), safe water self-sufficient buildings become possible. Domestic wastewater can be physically eliminated – and with that all the potential threats for public health and the environment caused by fecal contamination and micro-pollutants.

TD-P_03 Process simulation of biological degradation processes in waste management

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1. Objectives

The aerobic treatment of biogenic waste can take place in composting systems, but also in mechanical-biological pretreatment plants (MBP) prior to landfilling. It is partially coupled with anaerobic pretreatments. The anaerobic treatment of biowaste takes place in digestion systems. The utilisation of energy crops through fermentation was pushed by law in Germany (Renewable Energy Law - Erneuerbare Energien Gesetz, EEG, 2012). Predominantly anaerobic degradation processes can also be found in landfills, in which biogenic materials were deposited. However, when air enters, aerobic degradation processes also take place in these landfills.
The aim of the presented work was to simulate the microbiological and biochemical processes of aerobic and anaerobic degradation processes on the computer to such an extent where statements regarding process sequences or process improvements could be made with little effort. This also includes the qualitative and quantitative description of the emissions during the different degradation phases. Furthermore, standard process values such as the gas formation rate or the oxygen consumption are simulated. For validation purposes, simulated and empirically obtained values were compared. In this way, a software called SimuCF has been created that is able to simulate the complex systems of aerobic, anaerobic, and combined treatment or utilisation methods for waste with biogenic shares [1].

2. Methodology

The simulation program was written with LabVIEW™ 7.0 from National Instruments. For the simulation of aerobic and anaerobic biological degradation processes, almost exclusively mathematical and physical correlations and reaction equations from the basic fields of chemistry, biochemistry, microbiology, and physics were used. Only a few empirically obtained mathematical equations were used for the description of correlations. The programming environment of LabVIEW which is, above all, employed in the instrumentation, control, and automation technology, contains graphic programming elements which are composed of virtual instruments (VIs). The VIs consists of a front panel and a block diagram each. The front panel with the front view is the user interface via which input and output values can be displayed in different ways. The block diagram contains the graphical program code and processes the input values from the front panel. The verification of the program was accomplished permanently during programming. The program code is compiled, so the achievement is comparable with other programming high-level languages.
The numeric simulation model can be designated as a forecast, optimisation, explanation, or configuration model, depending on the application purpose. The model can be assigned to the immaterial, formal, and mathematical simulation models.

3. Results and discussion

The simulation model shows the complex relationships of the microbiologically treated materials that contains carbohydrates, proteins and/or fats. The developed software SimuCF help to find process optimisation steps for both the aerobic treatment (e. g. composting) and the anaerobic biogas production (fermentation) or conditions for changing milieu parameters (e. g. ventilated landfill and aerobic/anaerobic process).
The question about the status of the nutrients carbon, nitrogen and sulfur in the different phases can be shown with this process simulation software for the most different attitudes and modes.
A comparison of the simulation values with a variety of empirical values showed that the results are proportional and the development and value range are very similar.
The simulation software (Figure 1) with the simulation model, lying held in English language, can be characterised in summary as follows:

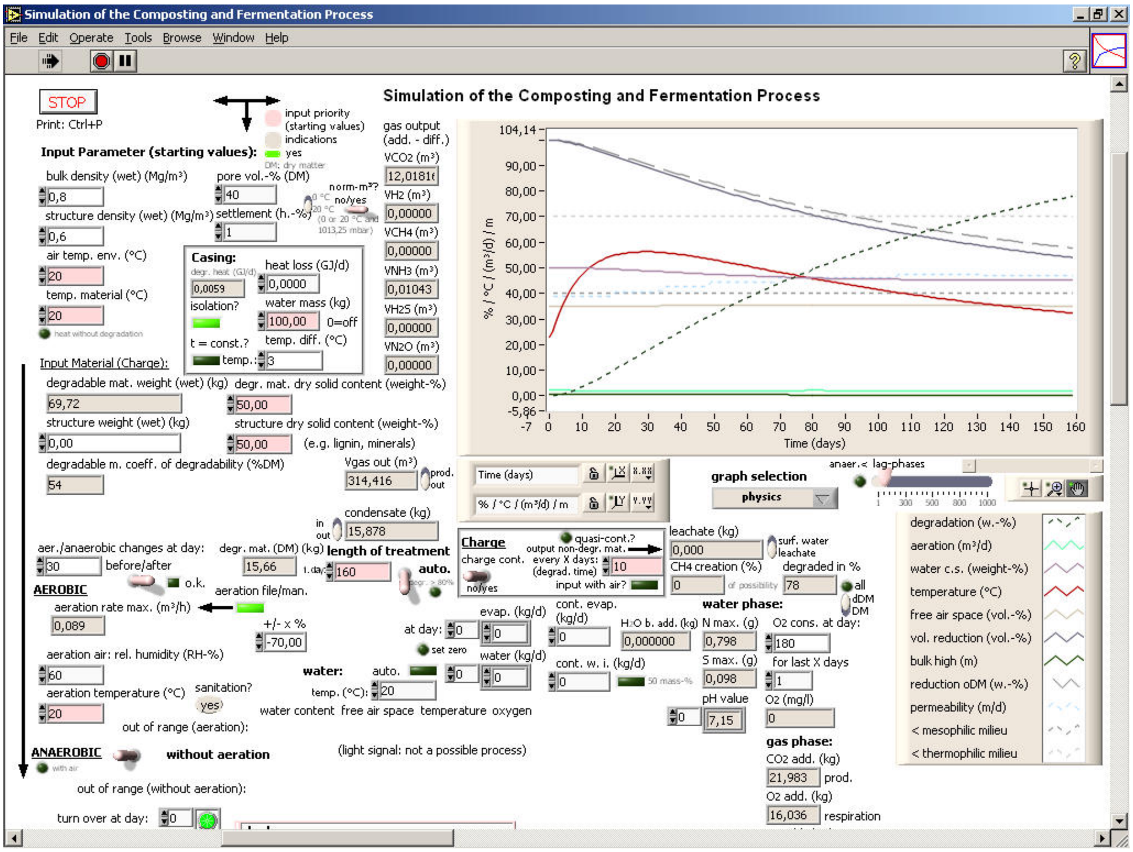

Figure 1: Part of the front view of the SimuCF software for the process simulation of biological degradation processes in the field of waste management with input and output boxes, and diagrams for the presentation of the results.

- The simulation software simulates aerobic and anaerobic microbiological, physical and chemical and/or biochemical processes in solids with a water content < 80 weight % (dry fermentation), in addition, the wet fermentation without solid-phase values.
- The gas flow through a single-layer material with different mixes, quantities and forms. The thermodynamics (heat conduction, thermal radiation, and convection) and the water balance are taken into account. The spatial distribution of the state parameters remains unconsidered (continuity of space).
- Five substance groups are taken into account: carbohydrates, proteins, fats, non-degradable organic substances and inorganic substances (minerals). The basic constituents that are contained in the special substrate, assigned to these groups. The organic part of the material is further subdivided into carbohydrates (general), starch, amino acids, hemicelluloses, fats, waxes, proteins, cellulose, and lignin, depending on the degradability.
- For carbohydrates, proteins, and fats different microbiological degradation curves are given, following approximately Monod kinetics, whereas also degradation delays (lag-phases) can be entered. The amount of material is also taken into account as a time delay factor, in order to include solution, transport, and diffusion processes.
- Considering specified limit values, the program automatically searches the maximum degradation rate at a total given degradation time or the degradation time is searched by the program with given minimum degradation rate.
- For selected organic fractions (e.g. straw, wood, bark, leaves, grass, apples, potatoes, turnips, wheat, peas, meat, and fish), the compositions are specified, and the quantities can thus be entered directly as a dry or wet weight, or as a percentage in the program.
- Exemplary compositions are also specified for some waste materials (e.g. sewage sludge, green organic waste, household waste, organic fraction household waste, biowaste, newsprint paper, and paper fibres).

- Modes of operation comprise aerobic and anaerobic milieu conditions (also on alternating basis), and different types of aeration are selectable. Also the following process operations can be simulated: batches, batches with selectable time intervals or semi-continuous batches.
- On the basis of the entry of approximately 30 initialisation parameters (e.g. ambient conditions, bulk or reactor form, material composition and initial operation conditions), many variations and influences can be simulated.

Example validation: A model validation was implemented by comparing simulated values with experimental data. Here exemplarily the composting is shown on the basis of a single experiment. The results of the simulated composting process with values from the laboratory experiment were compared with measured values of the laboratory experiment in a system with 100-L reactors, documented in [2][3]. Figure 2 shows results for the changes of water content, carbon dioxide and oxygen content, pH-values and substrate temperature during composting from the experiment and the simulation of the experiment.

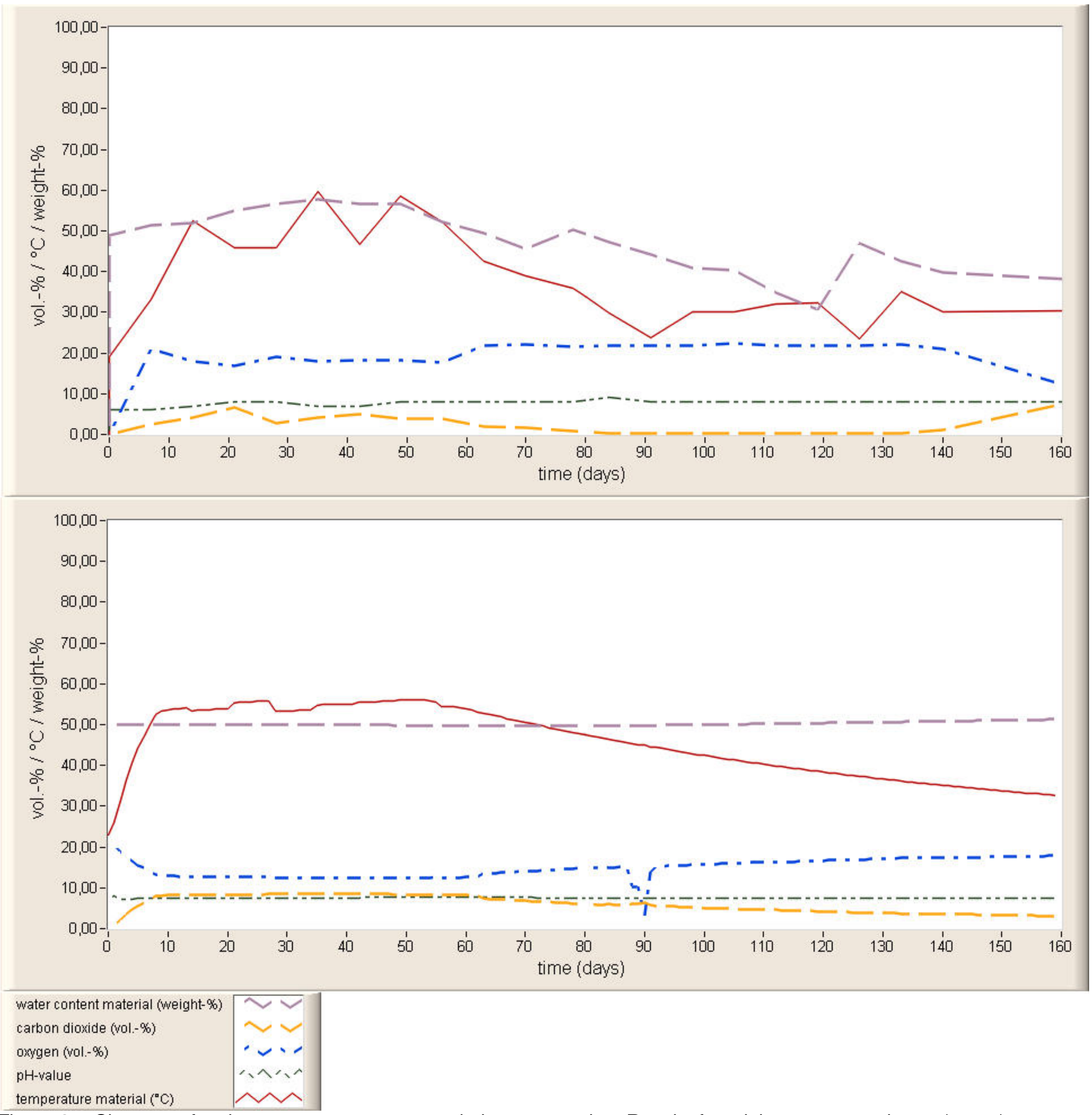

Figure 2: Changes of various process parameters during composting: Results from laboratory experiment (upper); corresponding results from simulation (lower).

The results from the laboratory and simulation experiment are comparable by trend. Differences could mainly be attributed to the fact that inhomogeneities formed during the experiment and that in the simulation, a homogeneous mixture is assumed. In addition, aeration variations were ascertained in the laboratory experiment. Missing data for this needed to be interpolated as an input for the simulation.

Results in the summary:

- On principle, the carbon dioxide and oxygen shares in the exhaust gas are comparable. The differences can be attributed to the variations of the air flow through the material due to changes of porosity during the dismantling e.g. compacting, and aeration irregularities. Aeration irregularities can be considered in the simulation through exact day-by-day recording of the air flow. This would lead to further approximation of the curve courses to each other.
- In contrast to the experiment, the water content did not decrease during the simulation and leachate did not accumulate. The reason is that the water-holding capacity was not exceeded in the simulated homogeneous mixture. In the laboratory experiment, leachate developed as a result of humid zones in the inhomogeneous mixture. The aeration also exerted significant influence on the water content. Desiccation and/or the water content in the simulation would be adapted to the experiment by the enhancement of the uneven aeration flow.
- The laboratory experiment included lime addition which was taken into account in the simulation as well. Inhomogeneities, and with it an uneven lime distribution, were taken into account in the simulation by only including 90 % of the lime in the calculations. In both cases, laboratory experiment and simulation, slightly lower pH values developed initially as a result of the initial acidification by biological processes.
- The simulated substrate temperature is comparable with the laboratory experiment. At the beginning, it increases up to almost 60 °C and drops down to approximately 30 °C at the end. Here, further approximations would also be possible, if the aeration entries were adapted.

4. Conclusion and outlook

The essential aerobic and anaerobic microbiological, chemical and biochemical processes but also physical effects are summarised in a simulation model.
The comparison of tendencies and orders of magnitude of the simulated values with experimentally determined results also showed in further validations that the simulation model provides sufficiently exact values that are similar to reality. The simulation model can be applied in practice, for example for the emission optimisation or plant planning, and for training purposes. Thanks to many setting alternatives, almost all of the standard process operating modes for the biological treatment of solid waste can be simulated. In addition to the parameters that are presented here, further values can be determined. That includes e.g. developments of the energy and explosion-relevant gases hydrogen and methane in anaerobic digestion. Furthermore, the fate of carbon, nitrogen, and sulphur in the different phases of composting and digestion can be ascertained.
In summary, it can be stated that substances that occur during microbiological, aerobic, and anaerobic degradation processes, can be examined and assessed under different milieu conditions that normally are, analogously to reality, not optimal. The results are represented graphically, and as array and single-value representations. Modifications of inhibiting influences can be simulated and processes can be optimised in such a manner. The heat and the water balance are also particularly important for this, and they are both taken into account in the simulation model.

The documentation regarding the program can be found in German in [1]; the SimuCF simulation software in English is available upon request at *deipser@t-online.de.*

References

[1] Deipser, A. (2014):Prozesssimulation biologischer Abbauprozesse im Bereich der Abfallwirtschaft, Dissertation, TU Hamburg-Harburg, Hamburger Berichte zur Siedlungswasserwirtschaft, Vol. 88, Gesellschaft zur Förderung und Entwicklung der Umwelttechnologien an der Technischen Universität Hamburg-Harburg e.V. (GFEU) (Hrsg.), ISBN 978-3-942768-13-9, urn:nbn:de:gbv:830-tubdok-12814, Eigenverlag, Hamburg

[2] Körner, I., Ritzkowski, M., Stegmann, R. (1999): Nährstofffreisetzung bei der Kompostierung und bei der Vergärung (Abschlussbericht) im Verbundvorhaben „Neue Techniken der Kompostierung“, Teil 1 des Teilvorhabens 10: „Stickstoffdynamik in Komposten und bei der Kompostanwendung“, Förderkennzeichen: 1450 638 C8, Umweltbundesamt (Hrsg.), Eigenverlag, Berlin

[3] Körner, I. (2008): Stickstoffhaushalt bei der Kompostierung – Bilanzen, Gehalte, Umsetzungs- und Austragsprozesse, Hamburger Berichte, Vol. 33, ISBN 978-3-9812867-0-0, Verlag Abfall Aktuell, Stuttgart

TD-P_04 Biogas and fertilizer from lawn silage and lawn silage juice

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1. Background and objectives

Lawn cuttings are generated as residues in large amounts in public areas and private gardens. Actually, they are often inefficiently used or disposed. One utilization option for lawn cuttings is the biogas production. In Germany, lawn cuttings only accrue between March and October, but biogas facilities need the feedstock for the whole year. To ensure feedstock availability lawn cuttings have to be stored. One possibility is preparing silage: grass cuttings are compacted and stored in encased bales or covered heaps. In such silos anaerobic conditions occur and lactic acid fermentation takes place. Due to acid production pH value decrease and lead to inhibition of organic degrading microorganisms. As result a conservation of the material is achieved [1].

The influence of ensilage on the biogas formation was investigated. This article focuses on experiments using lawn silage based substrates - lawn silage suspensions and lawn press juices - for anaerobic wet fermentation. The results were combined with earlier investigation with fresh lawn cuttings and with results from literature. The lawn cuttings were transformed into a suspensuin or a juice since the purpose was to use them wet fermentation. The work was in-cooperated into investigation around the Jenfelder-Au-project. In Hamburg's neighbourhood Jenfelder Au a new combination of renewable energy provision with innovative wastewater treatment (Hamburg Water Cycle®", HWC) shall be implemented. The construction of water systems started in 2013, the biogas facility will be erected in 2016, commercialisation of houses is planned to be finished in 2018. HWC includes a separate collection of rainwater, greywater and blackwater from vacuum toilets. Biogas will be produced from the blackwater in an anaerobic digestion process together with co-substrates. Besides greasy water, lawn cuttings are a potential co-substrate option [2].

2. Methodology

Grass from urban lawn was prepared into silage in laboratory scale and compared with agricultural grass silage. Two treatment options for the biogas processes – into lawn silage suspensions and lawn silage press juice - were investigated. The various silage preparations were characterized and the biogas potentials determined.

2.1 Origins of grass silage

Three grass-based silages were investigated. They differed in their origin, time of cutting, type of ensiling, and storing condition (1).

Table 1: Overview on the three silages from lawn cuttings used in the experiments.

Property	Silage 1	Silage 2a	Silage 2b
Origin and grass type	Extensively produced around an agricultural biogas facility	Lawn cuttings generated during park management at University campus	
Common cutting times	Once-twice a year	All 2-3 weeks in growing season	
Time of sampling and silage preparation	April-Mail 2014	22.07.2014	28.08.2014
Type of ensilage process	Heaps (practical scale)	Barrels (laboratory scale)	
Duration of ensilage	5-6 month	14-15 weeks	11-12 weeks

Silage 1 from a real scale heap silo (feedstock of Biogas plant Herling Hamburg-Reitbrook): The extensively cultivated grass is commonly used as feedstock in the Herling-biogas facility. For the biogas potential experiments approximately half-year-old silage was sampled and stored in a cooling chamber (8°C) until utilization.

Silage 2 from barrel experiment (lawn of the University campus): The lawn at the campus is usually cut every 2-3 weeks. It was sampled twice. The ensiling was carried out in 30-L-plastic barrels. Lawn cuttings were firstly compacted in the barrel to remove the air and then closed air tightly. A temperature recorder was positioned within the compacted material. In first eight weeks silage temperature was slightly higher than ambient temperature and later equalization followed until a constant temperature between 10 to 15 °C.

2.2 Silage preparations for biogas experiments

Silage suspensions: They were prepared in a laboratory macerator (Grindomix GM 300). The mixing ratio of silage cuttings with water was a compromise between an anticipated high organic matter content and the pumpability of the suspension. A further influencing factor was the water demand necessary for a good shredding result. The suspensions consisted of 17 mass % fresh silage cuttings and 83 mass % water (mixing ratio 1:5). The particle size was below 4mm (around 30 % of the solids were smaller than 0.1 mm). The suspensions were stored in a deep freezer (-20°C) until utilization.

Silage juice: The juices were prepared using a hydraulic manual press (Model WPT 10 HC). The operating pressure was up to 2 Mg. About 20% of the original amount of silage was collected as juice. The pressing was carried out 3 days before usage and the juice was stored till usage under cooled conditions (8°C).

2.3 Determination of biogas potential

Six samples were tested as a substrate for anaerobic digestion under mesophilic conditions (37°C +-°C) in a batch test system following the method described in [3]:

- Suspension of Silage 1: SS1 (Herling silage)
- Suspension of Silage 2a: SS2a (TUHH July silage)
- Suspension of Silage 2b: SS2b (TUHH August silage)
- Juice of Silage 1: J1 (Herling juice)
- Juice of Silage 2a: J2a (TUHH July juice)
- Juice of Silage 2b: J2b (TUHH August juice)

The feedstocks were mixed with inoculum, which was collected from the wastewater treatment plant Seevetal (mesophilic digested sewage sludge). The 1-L-reactors were placed in a temperature controlled water bath. Biogas volumes were measured by displacement method using Eudiometers. The tests were carried out as double determination for 21 days.

2.4 Methods for substrate analyses

The methods used for characteristics of the three silage samples as well as the suspensions and juices are summarized in Table 2. Samples were taken and analysed before and after the anaerobic batch test.

Table 2: Methods for substrate analysis.

Parameter		Guideline /Equipment
Dry matter content	DM in % FM	DIN EN 12880 (DIN EN 12880) [4]
Organic dry matter content	oDM in % DM	DIN EN 12879 (DIN EN 12879) [5]
Chemical oxygen demand	COD in mg/kgFM	Cuvette test by Hach Lange, Germany
Total organic carbon content	TOC in mg/kgFM	TOC/TN Analyzer multi N/C 3000 by Analytika Jena, Germany
Total nitrogen content	TN in mg/kgFM	TOC/TN Analyzer multi N/C 3000 by Analytika Jena, Germany
Ammoniacal nitrogen content	NH_4^+/NH_3-N in mg/kgFM	Cuvette test by Hach Lange, Germany Destillation unit K350 by Büchi, Germany
Total phosphorus content	TP in mg/kgFM	Cuvette test by Hach Lange, Germany
pH-value	pH	pH meter, model 323 by WTW, Germany
Biogas potential	in nL/kgoDM in nL/kgFS	VDI 4630 [3]

FM - Fresh matter; nL - norm liter (273K, 1013 hpa); FS – Fresh silage

3. Results and discussion

3.1 Silage preparation from lawn cuttings

In all experiments a decrease of pH occurred during the ensilage process. After opening the silage barrels a slightly acidic, but not unpleasant smell was recognized. The DM content of the substrate did increase slightly in most cases, oDM decreased slightly. Table 3 shows the results of the three silages as well as of lawn juices.

Table 3: Results from chemical analyses of lawn silage and lawn juice.

Parameter	Unit	TUHH July (S2a)		TUHH August (S2b)		Herling (S1)	
		Silage	Silage juice	Silage	Silage juice	Silage	Silage juice
DM	% FM	28.1	5.3	19.3	3.5	33.1	8.1
oDM	% DM	81.2	62.4	81.6	60.3	83.6	66.8
COD	mg/kgFM	233600	59400	244400	59600	323300	60600
TOC	mg/kgFM	109830	12409	107310	11248	101894	12276
TN	mg/kgFM	6102	5247	5543	3722	6929	4311
NH_4^+/NH_3-N	mg/kgFM	1288	347	896	252	1109	171
TP	mg/kgFM	1360	1280	840	1060	1070	1690
pH	-	5.2	5.0	5.4	5.0	4.4	4.2

FM - Fresh matter; DM – Dry matter

3.2 Biogas potential

Figure shows the specific biogas production (related to oDM) for the three silage juices and for the three suspensions. The curves of the suspensions are almost identical. The curves show that biogas formation started without delay in all juices and with minor lag-phases in the suspensions.

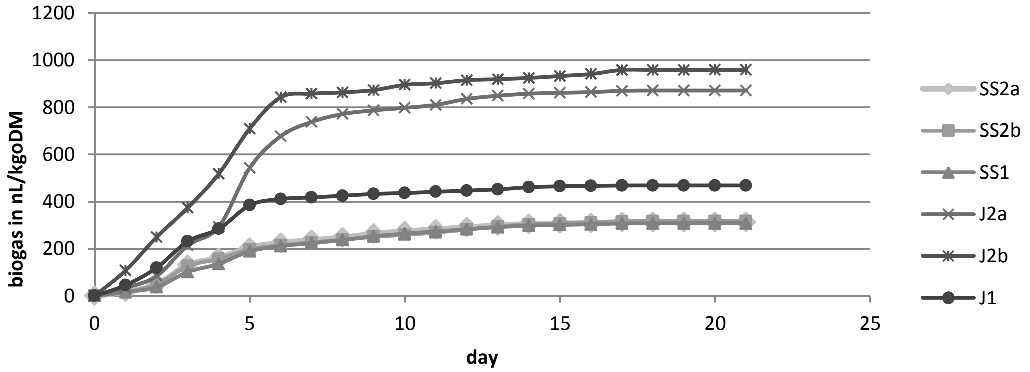

Figure 1: Biogas production for three different lawn juices (J1, J2a, J2b) and lawn suspensions (SS1, SS2a, SS2b).

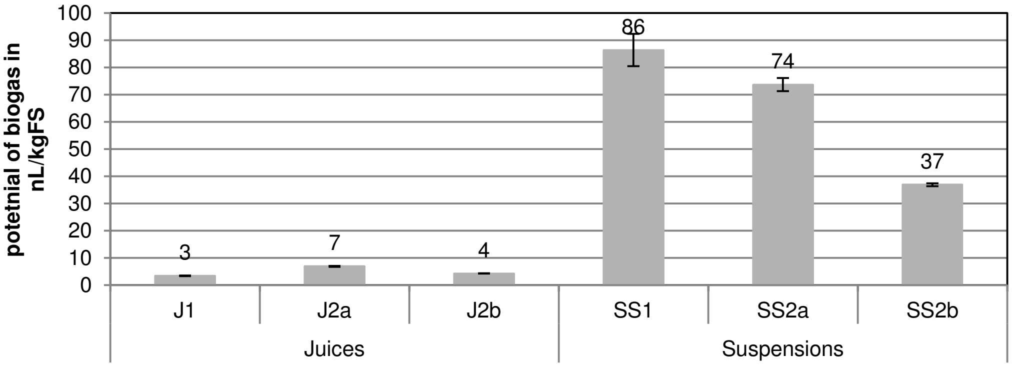

Figure 2: Biogas potential related to fresh organic matter of the originally used lawn silage (FS) for silage suspensions and silage juices.

The specific biogas yields (related to oDM) from the juices was clearly higher compared to the silage suspensions. The juices contain primarily the soluble and therefore more easily available organics from the silage cuttings. But also some small particulate matter was contained. It can be assumed that only easy fermentable organics is pressed out; hardly digestible lignocellulosic shares remain in the press cake. The solid particles in the juice are small in size and therefore have a large surface for microbial attack which is presumably from further advantage compared to the macerated fractions with bigger particle sizes.

Figure 2 shows the biogas potential for silage suspensions and silage juice related to the originally used fresh silage. The suspensions have a clearly higher yield with 84 ± 10 nL/kg FS. The juices had only a yield of 5 ± 1 nL/kg FS. The reason is the higher total organic matter content in the suspensions (Table 3).

The results for specific biogas production were compared with earlier investigation with other lawn silage juices and suspensions [6]. Weiler (2014) found specific biogas yields of 633 ± 66 nL/ kg oDM for lawn silage juices prepared from lawns from public areas [6]. The specific biogas yields for the corresponding suspensions ranged from 329 ± 11.64 nL/kg oDM for lawn from public areas up to 441 ± 8 nL/kg oDM for a suspension made with grass silage for agricultural purposes. The results of [6] were in the same ranges as in the experiments described here.

Table 4 shows an overview of biogas potentials from various lawn substrates with and without ensilage and with different preparation methods.

Table 4: Dry matter, organic dry matter, biogas potential, biogases methan content and the organic dry matter reduction durin anaerobic fermentation for various lawn-based feedstocks (own investigation and literature results).

Substrate	No. samples	DM %	oDM % DM	Specific Biogas nL/kgoDM	nL/kgFM	Methan %	oDM reduction %	Literature
lawn cuttings								
Fresh cuttings	2	38	52	393 ±4	78			[7], [9], [8], [10]
lawn silage	12	27	79	366 ± 49	75	62	45	[9]
grass silage	8	30	86	375 ± 51	99	61	45	[9]
lawn juice from								
fresh lawn	6	6	68	500 ± 7	20	71	60	[11], [12]
lawn silage	4	4	61	909 ± 44	25	70	75	
grass silage	4	6	66	531 ± 73	19	71	75	

The specific biogas potential of fresh lawns ranges very widely from 300-700 nL/ kg oDM. [7] gives 300 nL/ kg oDM, [8] 300-600 nL/ kg oDM, [9] 550-680 nL/ kg oDM and [10] 700 nL/ kg oDM. This variations have several reasons: 1) dry matter content may vary strongly with harvesting time and weather conditions; 2) chemical composition may change over the growing period with more ligneous shares in autumn compared to spring; 3) organic impurities such as leaf (poorly anaerobically degradable) are contained more often in high shares in autumn.

For lawn silages and lawn juices less literature information were available. The few refer also more to grass from extensive productions. [9] gives a specific biogas potential for grass silage from 600 nL/ kg oDM from extensively produced grass. Some results for fresh lawn juices are available from literature focusing ob biorefinery processes, where the juice is used for higher valued applications. [11] gives 617 nL/ kg oDM and [12] 550 nL / kg oDM (calculated from 350-400 l methan/kg oDM with an assumed methane concentration of 70%).

In summary from from own investigation it can be concluded, that the biogas potential of the material after the ensilage process was in the same range like the fresh material, but more homogeneous. Also [13] showed, that ensilage of lawn cuttings had no mentionable influence on biogas potential. On the one side, lactic acid fermentation process of ensilage leads to degradation of organics, but on the other a cell disruption takes place and can even improve the biogas potential [14]. Similar to fresh cuttings also in the ensilaged variants the DM content and degree of lignification of the lawn had main influence of the specific biogas potential. It decreases with increasing water content and lignification, but since during ensilaging some mixing of fractions and homogenization takes place, the variations of silage are generally lower compared to fresh lawn cuttings.

3.3 Overall evaluation of lawn usage for biogas processes

The preparation of silage from agricultural grassland is frequently applied, but it is actually not the case for lawn cuttings from public areas. Urban lawn cuttings differ in many aspects, e.g. with respect to harvested amounts and times as well as chemical composition from extensively produced grass. It could be shown, that it is also possible to prepare silage from public lawn cuttings successfully. The process can be carried out also in small scale with the advantage that low harvested amounts can be used. For calculations in biogas process plannings following values for specific biogas potential for lawns from urban areas, and the respective lawn silages and lawn juices can be used:

- Fresh lawn: 300 – 700 nL/ kg oDM
- Lawn silage / lawn silage suspension: 300 – 400 nL / kg oDM
- Lawn juice: 600 – 950 nL / kg oDM

Also the nutrient potential of the substrates was investigated. This becomes important, when thinking about utilization of digestates. E.g. the juices contained between 1000 and 1700 mg/l (kgFM) phosphorous (P), for the silage P values ranged from 800 to 1400 mg/kgFM. The nitrogen content of the silage varies between 5000 to 7000 mg/kgFM and 3500 to 5000 mg/l (kgFM) for the juice (Table 3). After digestion the nutrients are still in the digestate. It is attractive as fertilizer.

4. Conclusion

The work shows that lawn cuttings from public area can be stored as silage. Both, silage and silage juice provide good biogas yields and have a considerable nutrient potential. This makes lawn cuttings from public areas an interesting bioresource for biogas and fertilizer generation, which is available the whole year. Biogas production, based on organic matter, is higher for pressed juice compared to suspended silage; based on fresh matter, the silage biogas potential is much higher. Which variant is suitable is case-specific and depends on fermentation plant and feeding system.

References

[1] Oude, Elferink; Stefanie J. W. H.; Driehuis, Frank; Gottschal, Jan C.; Spoelastra, Sierk F. (1999): Silage fermentation processes and their manipulation. In Food and Agriculture Organization of the United Nations (FAO) (Ed.): FAO Electronic Conference on Tropical Silage, 01.-15.12.1999. The Grassland and Pasture Group and Feed Resource Group of FAO. Available online at http://www.fao.org/WAICENT/FAOINFO/AGRICULT/AGP/AGPC/gp/SILAGE/PDF/Paper2.pdf, checked on 1/9/2015.

[2] KREIS (2012): Versorgung durch Entsorgung. Kopplung von regenerativer Energiegewinnung mit innovativer Stadtentwässerung. Edited by Hamburg Wasser. Available online at www.kreis-jenfeld.de.

[3] VDI 4630, 2014: Vergärung organischer Stoffe - Substratcharakterisierung, Probenahme, Stoffdatenerhebung, Gärversuche.

[4] DIN EN 12880, 2000: Charakterisierung von Schlämmen - Bestimmung des Trockenrückstandes und des Wassergehalts.

[5] DIN EN 12879, 2001: Charakterisierung von Schlämmen - Bestimmung des Glühverlustes der Trockenmasse.

[6] Weiler, K. (2014): Analyse des Biogaspotentials von unterschiedlich gelagerten und aufbereiteten Rasenschnitt; Bachlor Thesis; Hamburg University of Technology, Institute of Wastewater Management and Water Protection.

[7] FNR (2006): Fachagentur Nachwachsende Rohstoffe e.V. (Ed.): Handreichung Biogasgewinnung und -nutzung (2006). Available online at http://www.big-east.eu/downloads/FNR_HR_Biogas.pdf, checked on 2/25/2014.

[8] Westphal, L. (2010): Biogaspotenzial von Gras- und Rasenschnitt. Energetische Verwertung von organischen Reststoffen am Beispiel der grasartigen Bioressourcen im Bezirk Bergedorf. Diplomarbeit. Technische Universität Hamburg-Harburg, Hamburg. Institut für Umwelttechnik und Energiewirtschaft.

[9] KTBL (2014): Kuratorium für Technik und Bauwesen in der Landwirtschaft; Wirtschaftlichkeitsrechner Biogas. Edited by KTBL. Available online at http://daten.ktbl.de/biogas/startseite.do.

[10] Kruse, A. (2011): Einfluss der verschiedenen Lagerungsverfahren auf das Biogaspotenzial von Rasenschnitt. Bachelorarbeit. Technische Universität Hamburg-Harburg, Hamburg. Institut für Umwelttechnik und Energiewirtschaft.

[11] Wachendorf, M., Hensgen, F., Bühle, L., Möller, D.; Blumenstein, B. (2012): Aus Grass Pellets und Biogas erzeugen. In Top Agrar -Energiemagazin (2). Available online at http://www.prograss.eu/fileadmin/PROGRASS/documentation/german_press/topagrar_2_2012.pdf, checked on 10/2/2014.

[12] Florafuel AG: Das Florafuel-Verfahren: Bioenergie aus Biomasse. Available online at http://www.florafuel.de/de/florafuel-verfahren/das-florafuel-verfahren/, checked on 10/2/2014.

[13] Schidler, Susanne; Adensam, Heidi; Hofman, Roswitha; Kromus, Stefan; Will, Markus (2003): Technikfolgen-Abschätzung der grünen Bioraffinerie. Band II Materialsammlung. Edited by iTA. Institut für Technikfolgen-Abschätzung der Österreichischen Akademie der Wissenschaften. Wien. Available online at http://epub.oeaw.ac.at:8000/ita/ita-projektberichte/d2-2d21-2.pdf, checked on 11/24/2014.

[14] Rösch, C.; Raab, K.; Skarka, J.; Stelzer, V. (2007): Energie aus dem Grünland – eine nachhaltige Entwicklung? Edited by Forschungszentrum Karlsruhe GmbH. Forschungszentrum Karlsruhe in der Helmoltz-Gemeinschaft; Institut für Technikfolgenab. Karlsruhe. Available online at http://www.itas.kit.edu/pub/v/2007/roua07b.pdf, checked on 1/15/2015.

TD-P_05 Multi-composting as a tool to produce compost from olive mill waste as a substitute for growing strawberries in the United Kingdom

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1. Objectives

The objectives of this study as part of a wider project were to assess the effects of using compost produced from olive mill wastewater (OMW) and animal manure as compost for growing soft fruits. These objectives will test whether:

1. Different ratios (v/v) of the OMW compost have an effect on the growth of soft fruits
2. Does the addition of biochar affect yield of fruit and nutrient retention in the compost

The process of olive oil extraction is a worldwide industry, predominantly based in Mediterranean countries, but recently intensively cultivated in Argentina, Australia and South Africa. The olive tree (*Olea europea*) is one of the main Mediterranean crops with a cultivated area of approximately 8.2Mha [1]. This crop produces an annual volume of 10 million m^3 of olive mill wastewater and 6 million m^3 of solid olive mill by products; consisting of olive stones, leaves, and pomace [2]. The OMW is high in nutrients, especially potassium, but can also have phytotoxic effects on plants due to the high levels of phenols present.[3]

2. Methodology

The OMW product used in these trials has been developed using repeated composting, there are two different composts being tested. The composts have been composted two and three times respectively with chicken manure as the base component. The product was initially composted in windrows with green garden waste as a bulking agent with OMW sprayed onto them to correct the moisture. The chemical characteristics of the raw OMW are given in Table 1.

Table 2: The chemical characteristics of the raw OMW used to produce the compost.

Parameters	Value
pH	4.9
EC (mS/cm)	2.3
N (mg/l)	331.95
K (mg/l)	2345.1
P (mg/l)	152.4
TOC (%)	23.0
NO3- (mg/l)	< detection limits
NH4+ (mg/l)	95.8
Mg (mg/l)	143,2
Ca (mg/l)	307.6
Phenols (g gallic acid/l)	3.4
Heavy metals	
Cu (µg/l)	229.4
Zn (µg/l)	5.441.1
Ni (µg/l)	253.3
Cr (µg/l)	5.3
Pb (µg/l)	< detection limit

During the first composting procedure these windrows were turned 14 times. For the second composting process fresh manure was introduced and the compost from the first stage was used as the bulking agent. The windrows were turned 8 times with 2.8l of OMW added at each turning. A portion of the final product from the second composting was used as a bulking agent for the third round of composting, with fresh chicken manure added. The final composting took place in composters with 8 turnings over 96 days with 70l of OMW added in that time. The compost was then dried by spreading in a thin (15-20cm) layer in a greenhouse and then turned and 84l of OMW added over a course of 55 days. The compost was then transferred into tanks for the final drying for 115 days being turned daily with 260l of OMW added.

Biochar added to crops can reduce the bulk density of the substrate, reduce nutrient leaching and can also help control plant pathogens [4-6]. The addition of biochar along with the OMW will allow for possible trends using these treatments to be assessed.

The Elsanta (*Fragaria x ananassa*) variety of strawberry was the soft fruit chosen for these trials given its use by commercial growers and popularity with consumers in the UK market. Two strawberry plants were used in each pot with 5 replicates for each treatment, giving 10 plants for each treatment. For objective 1 three different ratios of compost were used in 10, 25 and 50% v/v ratios for each compost along with a control. The treatments for objective 2 were v/v ratios of 10 and 25% OMW along with an addition of biochar (BC) at a rate of 10Mkg/ha. The OMW compost was combined with commercial peat free compost (PFC) to provide a soilless substrate base. Treatments for the trials in 2014 are shown in Table 2.

Table 2: The treatments prescribed for the strawberries in 2014.

Treatment	Compost combination
Control 1 (C1)	100% PFC
Control 2 (C2)	100 PFC
1	10% OMW2 + 90% PFC
2	10% OMW2 + 90% PFC + BC
3	25% OMW2 + 75% PFC
4	25% OMW2 + 75% PFC +BC
5	50% OMW2 + 50% PFC
6	10% OMW3 + 90% PFC
7	10% OMW3 + 90% PFC + BC
8	25% OMW and 75% PFC
9	25% OMW and 75% PFC + BC
10	50% OMW3 + 50% PFC

The strawberries were supplied from cold storage by commercial grower and ready to grow. They were then transplanted following defrosting into 2l pots and placed in a randomised arrangement in the polytunnel. Fertiliser was added to all the pots on day 20 when flowers had appeared on each of the plants. This was completed using a standard fertiliser for strawberry production, high in potassium for fruit development. The addition of fertiliser and management of the strawberry plants were done in accordance with guidance from MAFF (now DEFRA) [7].

Yield of strawberries in number per replicate and the average weight of 10 marketable fruits was recorded. Assessing marketability of fruits was completed in accordance with the Class I standard as detailed in Council Regulation (EC) No 1234/2007. This is the standard used for high quality strawberries in the UK. Flesh firmness and sugar content were measured on these 10 marketable fruits using a penetrometer and a refractometer respectively. Total yield from each plant from all harvests was calculated with intact fruits with a diameter greater than 18mm and a weight more than 4g representing marketable yield. Smaller fruits, malformed and rotten fruits were counted, weighed and discarded.

3. Results and discussion

3.1 Quantitative results analysis

Fruits were harvested on 10 dates; the length of time from transplanting to harvesting the first fruits was 55 days. The total number of marketable strawberries produced from each replicate, and those that were discarded are shown in Figure 1. There is a trend in the control treatments (C1 and C2) and in the treatments with the 25% rate of three times composted OMW (treatments 8 and 9) that with an addition of biochar there is an increase in the number of strawberries produced. In all the other treatments with biochar added there is a trend of decreasing the number of strawberries produced per treatment.
The results from the statistical analysis show that none of the different treatments from either OMW product are significantly different from each other or the control for weight. All results for the Tukey's test were greater than $p<0.05$.

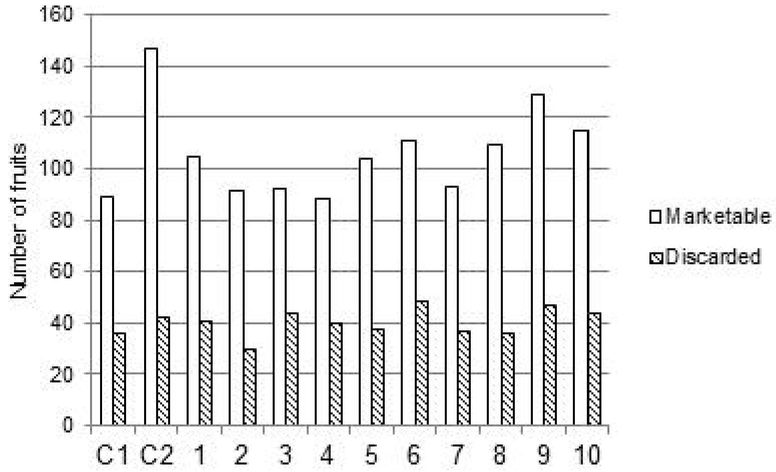

Figure 1: The total number of fruits that were marketable and discarded from the different treatments.

3.2 Qualitative results analysis

The comparison between the 10 marketable strawberries that were analysed in detail from each replicate can be subjected to a more critical analysis. The results from the statistical analysis show that none of the different treatments from either OMW product are significantly different from each other or the control for width of fruit. All results for the Tukey's test were greater than $p<0.05$.
The results of the penetrometer test for the 10 marketable fruits from each replicate are shown as a mean average in Newtons (N) in Figure 2. The results of the penetrometer method in the Tukey's test showed that C1 was significantly higher than treatment 5 and 8. Treatment 10 (50% OMW3) was also significantly higher than treatment 8. The 50% OMW3 treatment (10) was significantly higher than all treatments bar C1 and treatments 2 and 4. The average for treatment 1 was significantly higher than that for treatment 5.

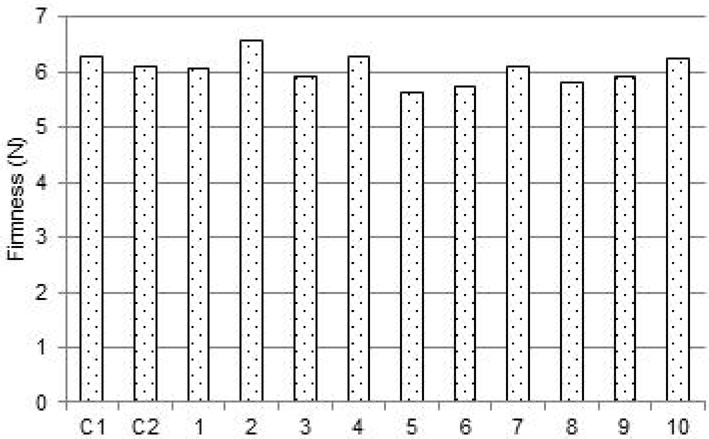

Figure 2: The penetrometer value in Newtons for each treatment.

The results from the refractometer test on the fruits for sugar content were subject to analysis using Tukey's test. In this assessment, no treatments were significantly different from each other or the control with a p value of <0.05.
The number of strawberries infected with *Botrytis cinerea* as part of the discarded cohort were counted and assessed for each treatment and expressed as a percentage of the total number of discarded strawberries. The results of this are shown in Figure 3. Biochar has been shown to have positive effects with pathogen control in soft fruit, and some evidence of this is seen between C1 and C2 and treatments 8 and 9. The same trend is not seen between treatments 1-2, 3-4, or 6-7.

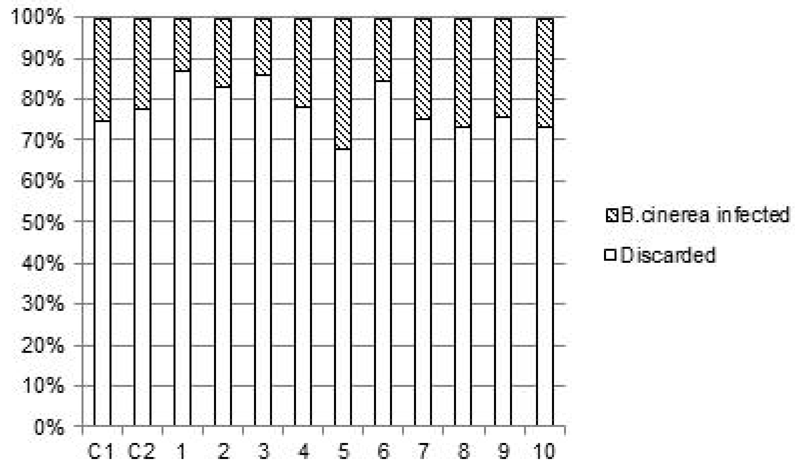

Figure 3: The number of *B. cinerea* infected strawberries expressed as a percentage of total discarded strawberries.

4. Conclusion and outlook

There is no statistical difference between any of the compost treatments and the control for weight or width of fruit, or numbers of strawberries produced. These trials demonstrate that under these conditions compost produced from chicken manure and OMW can compare similarly when assessed against fruits grown in traditional peat-free compost.

Further trials with a greater number of replicates and with a more gradual increase in the v/v percentage of compost used may show a change in the growth of strawberries. The fertiliser added to the treatments may have masked any phytotoxic effects from the OMW compost by making nutrients more readily available for uptake into the plants. A comparison between plants grown in OMW compost using fertiliser and without using fertiliser may highlight potential phytotoxic effects of the phenols present in the compost. Different biochar application rates may have a more significant effect on pathogen control in strawberries.

Acknowledgements

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References

[1] López-Piñeiro, A., et al., Short and medium-term effects of two-phase olive mill waste application on olive grove production and soil properties under semiarid mediterranean conditions. Bioresource Technology, 2008. **99**(17): p. 7982-7987.

[2] Nektarios, P.A., et al., Effect of olive mill compost on native soil characteristics and tall fescue turfgrass development. Agronomy Journal, 2011. **103**(5): p. 1524-1531.

[3] Di Bene, C., et al., Short- and long-term effects of olive mill wastewater land spreading on soil chemical and biological properties. Soil Biology and Biochemistry, 2013. **56**(0): p. 21-30.

[4] Atkinson, C., J. Fitzgerald, and N. Hipps, Potential mechanisms for achieving agricultural benefits from biochar application to temperate soils: a review. Plant and Soil, 2010. **337**(1-2): p. 1-18.

[5] Lehmann, J., et al., Biochar effects on soil biota – A review. Soil Biology and Biochemistry, 2011. **43**(9): p. 1812-1836.

[6] Meller Harel, Y., et al., Biochar mediates systemic response of strawberry to foliar fungal pathogens. Plant and Soil, 2012. **357**(1-2): p. 245-257.

[7] Ministry of Agriculture, F.a.F.M., Strawberries. 1970.

TD-P_06 Wastewater treatment plant of the future – Energy storage in interaction with technical infrastructure between the poles of energy generation and consumption (“ESiTI”)

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1. Background

Municipal wastewater treatment requires a considerable input of energy. However, at the same time, wastewater contains energy especially in the form of heat and chemically bound energy, both utilizable within the wastewater treatment process. In municipal wastewater treatment plants, energy consumption and generation are usually subject to separately optimized processes and are separated in space and time, as well. Corresponding to operating conditions at a wastewater treatment plant (WWTP), the energy demand shows daily, weekly and annual variations (cf. Fig. 1). Though, today’s energetic optimization is typically based on average values, impacts of daily variations are very rarely considered [1].

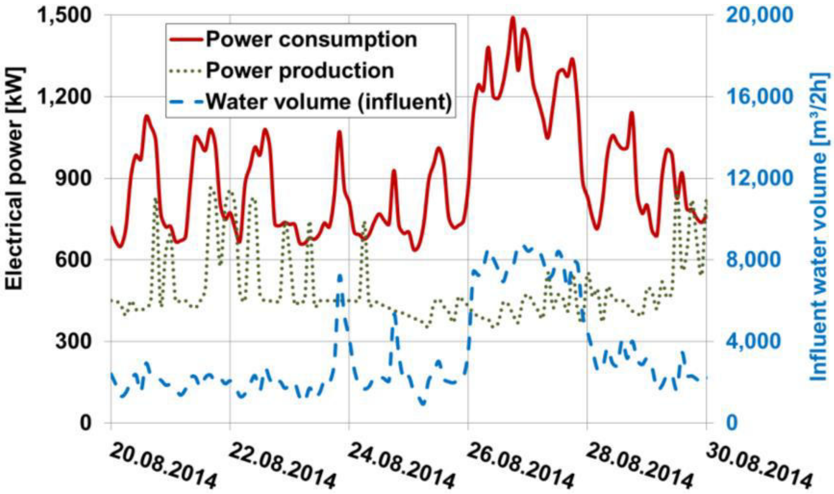

Figure 1: Power consumption and production of a WWTP with a size of 240,000 PE according to the influent water flow [1], [2].

In the context of structural changes of the power supply based on grid connections of supplier using renewable energy sources, developments of consumer working as power storage are necessary. WWTP can become an efficient energy storage by crosslinking energy consumption and generation. Furthermore, by interacting with technical infrastructure facilities, e.g. power providing companies or large-scale energy consuming resp. generating enterprises (industry, waste incineration plants, etc.) new potentials of efficient energy utilization and generation can be developed. Due to its potential regarding energy storage and generation, sewage sludge treatment is an important component of this system.

One technical option is to use the energetic potential of sewage sludge/co-substrate by anaerobic digestion. Figure 2 shows the digester gas production of a municipal WWTP with a capacity of 32,500 PE (PE = Population Equivalent). Sludge is digested in two simultaneously charged digesters, whereat raw sludge is continuously charged into both of the digesters to the same volume. Digester 1 is additionally fed with co-substrates (food residues). The additional charging of digester 1 with co-substrates leads to a quick increase of digester gas production that reaches the level of digester gas production in the reference digestion fed only by sewage sludge after 5 hours. This can be used for a flexible operation of the digestion with the objective to meet energy needs under consideration of maintaining a stable process/operation.

Sewage sludge as well as the co-substrate therefore can serve as energy storage through the chemically bound energy.

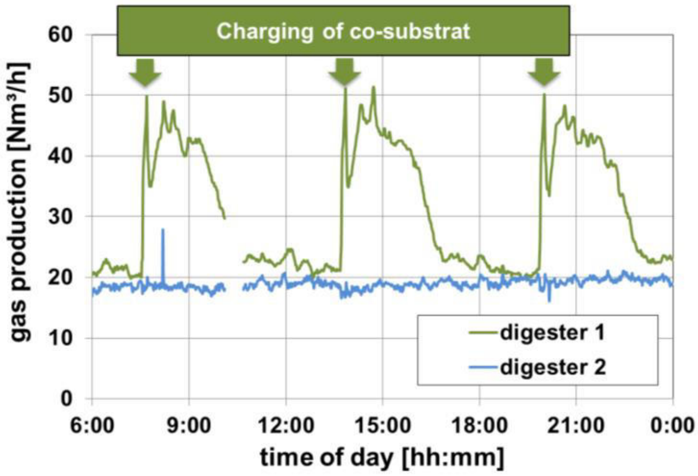

Figure 2: Digester gas production of a municipal WWTP, capacity 32,500 PE, digester 1: input of raw sludge (50% of total amount); digester 2: input of raw sludge (50% of total amount) and co-substrate (food residues) [3].

2. Objectives

Within the scope of the research project "Wastewater treatment plant of the future: Energy storage in interaction with technical infrastructure between the poles of energy generation and consumption ("ESiTI" - www.esiti.de)" the connections between energy consumption and energy generation are assessed. Therefore, the WWTP is evaluated as to its potential to serve as energy storage. The linkages to peripheric infrastructural actors such as energy service providers, energy consumers and producers, e.g. industrial companies or waste incineration plants, are analyzed with regard to new potentials for energy generation and consumption.
Through exemplary investigations for the Wissenschaftsstadt Darmstadt, a high user relevance is established. Darmstadt lies in the southern part of the metropolitan area Rhine-Main with about 145,000 inhabitants and therefore may serve as an example for numerous cities. The research project set up by 11 joint partners from commune, industrial and academic partners and funded with 2.7 mio. Euro for a duration of three years, is one from 12 joint projects from the research initiative "Future-oriented Technologies and Concepts for an Energy-efficient and Resource-saving Water Management (ERWAS)" funded by the German Federal Ministry of Education and Research.

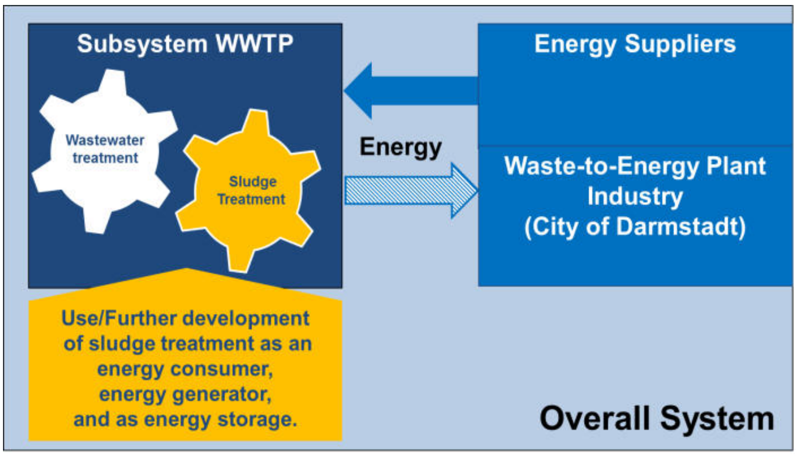

Figure 3: Wastewater treatment plant as energy consumer, producer and storage

The objective is to develop a manual-based planning tool as well as tools for real-life application, under consideration of technical, ecological, economic and social aspects for operating future wastewater treatment plants in interaction with infrastructure facilities. Target is the identification of the sewage sludge treatment plant as energy service provider, with focus on sewage sludge digestion as energy consumer, storage and producer by providing maximum flexibility of energy flows.

3. Methodology

The joint project ESiTI includes five major fields of activity: The system analysis evaluates the potential of the wastewater treatment plant to become an energy service provider through the identification and visualization of dynamic energy flows (electricity, heat, cooling energy as well as potential storage media such as sewage sludge, substrates, digester gas). The technology work package strives for the development of a sewage sludge treatment process as functional module of a flexible energy system under consideration of digestion and its optimization potentials, limitation of usage of co-substrates, thermal pressure hydrolysis, and thermal treatment processes. Different strategies, processes, and combinations will be investigated and compared by using key figures and decision trees. All technical aspects are framed by an ecological assessment of the process variants as well as a multi-criteria evaluation. Later, economic, ecological and social costs and benefits are incorporated. These four activities serve as a basis for the transfer work package which contains the development of a planning tool for the integration of the wastewater treatment plant into an energy network system.

3.1 Life Cycle Assessment in ESiTI

Life Cycle Assessment in ESiTI is carried out according to ISO 14040/14044 [4,5]. The functional unit for the project is set to 1 Mg of sewage sludge (dry matter/total solids; 1 Mg = 1,000 kg) to be charged into the digester. Figure 4 shows the system boundaries of the wastewater treatment plant and surrounding infrastructure facilities.

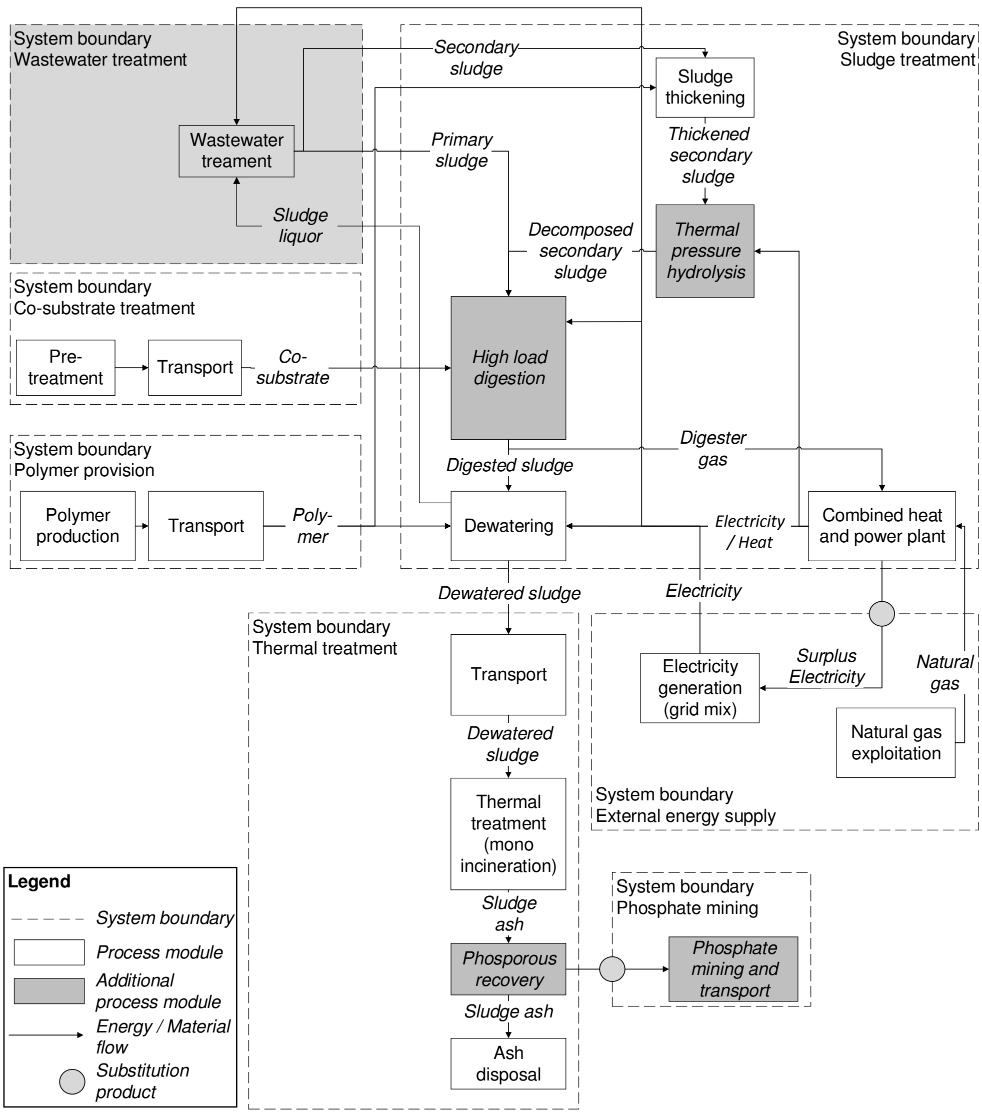

Figure 4: System boundaries of the Life Cycle Assessment in ESiTI (upstream chains only shown for processes on the WWTP).

The system boundaries include the on-site sludge treatment processes of sludge thickening, digestion, and dewatering. The thermal sludge treatment covers the energetic use of sewage sludge in a mono incineration plant. Pre-treatment processes for co-substrates and production of raw materials and supplies are also implemented. The wastewater treatment is included with a simplified approach that assesses the increased electricity demand of wastewater treatment for aeration induced by a higher nitrogen load in the sludge liquor.

In the further course of the research work, alternative processes will be implemented in the model such as thermal pressure hydrolysis, high load digestion and recovery of Phosphorous from mono-incineration ash. Input data for the adapted or newly implemented process modules come mainly from experiments in bench or pilot scale.

Two major research questions are the environmental assessment of the share of co-substrates to be fed into the digesters and the potential of the WWTP to serve as a flexible energy provider as to the supply of balancing energy.

A higher amount of co-substrate digestion rises energy self-sufficiency on the wastewater treatment plant but comes with higher environmental impacts for pre-treatment of co-substrates. Additionally, transport of food residues from the pre-treatment facilities to the WWTP may also conflict with the less carbon dioxide intensive energy generation on the WWTP.

The potential of the WWTP to serve as energy storage also enables the potential to provide balancing energy to the electricity grid that is assumed to be needed when fluctuating energy sources like wind power or photovoltaics emerge further. Balancing energy can be provided by every energy generation/consumption facility that is able to change its loads flexibly. The environmental assessment of the provision of balancing energy therefore has to cover the impacts that are caused by energy generation as well as the substituted amount of electricity coming from other sources. To assess this interlinkage to the power system, an approach that covers both sides has to be developed.

Acknowledgements

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References

[1] Schaum, C.; Lensch, D.; Zhukova, E.; Cornel, P. (2015): Kläranlagen als ein Baustein einer zukünftigen Energieinfrastruktur, 48. Essener Tagung, Gewässerschutz - Wasser - Abwasser (GWA), 236

[2] Schaum, C.; Lensch, D.; Lutze, R.; Cornel, P. (2014): Klärschlammbehandlung der Zukunft, wwt - wasserwirtschaft-wassertechnik, Nr. 10/2014, Berlin

[3] Lensch, D.; Schaum, C.; Cornel, P. (2014): Target-oriented Co-digestion - Possibilities to manage the peak in energy demand, IWA Specialist Conference „Global Challenges: Sustainable Wastewater Treatment and Resource Recovery“, 26.-30. Oktober 2014, Kathmandu, Nepal

[4] DIN Deutsches Institut für Normung (2006): DIN EN ISO 14040: Environmental Management – Life cycle assessment – Principles and framework

[5] DIN Deutsches Institut für Normung (2006): DIN EN ISO 14044: Environmental Management – Life cycle assessment – Requirements and guidelines

TD-P_07 Nutrients removal capacity of the green microalgae isolated from wastewater treatment plant in Hamburg

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Objectives

This research aims to inspect the nutrients removal capacity and the potentials of the microalgae to produce biomass in the wastewater for treatment and bioenergy production. The main objectives are to isolate and identify the green microalgae from wastewater and furthermore, to grow the isolated algae in the wastewater and monitor the growth rate, assimilation of the nutrients to determine the most productive strain.

Methodology

Wastewater samples (including sludge) were collected from Seevetal wastewater treatment plant (WWTP) to inoculate the artificial medium BG11 [1]. The algae plaques were transferred using methods of dilution-spread, dilution-spray and micromanipulation (with capillary) to isolate the algae and separate each distinct strain from other strain and other biological contaminations.
To eliminate the influence of the bacterial activity, municipal wastewater (from Seevetal) was autoclaved and filtered through glass microfiber filters. The nutrients in the wastewater were measured before inoculation and wastewater was introduced to airlift bioreactors (300 ml) in triplicates in light intensity of 720 µmol m^{-2} s^{-1}. The reactors were inoculated with the isolated algae and the batch cultivation continued for 12 days. Total nitrogen (TN) and non-purgeable organic carbon (NPOC) were measured using Shimadzu TOC/V auto-analyzer. Chemical oxygen demand (COD), Total phosphorus (TP), Phosphate (PO_4-P) and ammonium (NH_4-N) were measured using Hach-Lange standard cuvettes and DR3900 spectrophotometer. Biomass growth, pH were investigated throughout the experiment on days 0,1,3,5,8 and 12.

Results

Based the on microscopic photos, 7 distinct algae strains are distinguished. The genera of the isolated algae are Chlorella, Desmodesmus, Tribonema and Stigeoclonium. Due to the very small cell sizes (<10 µm), identification of the algae species using DNA sequencing is carrying out. Growth of 6 most productive strains is investigated in the wastewater. The nutrients removal also was impressive when we achieved up to 99% phosphorus removal on Day 3 and up to 96% ammonium removal on day 5 of the experiment with both Desmodesmus sp.(1) and Chlorella sp.(1). The highest attained biomass concentration was about 1.6 g/l in Desmodesmus sp.(1) and Chlorella sp.(1) as well. There was no significant correlation was found between COD and NPOC removal and the biomass growth. The nutrients removal and biomass growth in Tribonema sp. was delayed because of the morphology of this specie which is very sensitive to shear stresses.
Our results coincide with those of Cabanelas et al. (2013) and Ji et al. (2013) [2,3].

Conclusion

This experiment has shown that the native microalgae in the wastewater have a high nutrients removal potential. Two species which are recognized as Desmodesmus sp.(1) and Chlorella sp.(1) in our experiment have demonstrated a high biomass productivity and nutrients removal.

References

[1] Rippka, R., Deruelles, J., Waterbury, J. B., Herdman, M. & Stanier, R. Y. (1979). Generic assignments, strain histories and properties of pure cultures of cyanobacteria. Journal of General Microbiology 111, 1-61.

[2] Cabanelas, I. T. D., Ruiz, J., Arbib, Z., Chinalia, F. A., Garrido-Pérez, C., Rogalla, F., … Perales, J. a. (2013). Comparing the use of different domestic wastewaters for coupling microalgal production and nutrient removal. Bioresource Technology, 131, 429–436.

[3] Ji, M. K., Abou-Shanab, R. a I., Kim, S. H., Salama, E. S., Lee, S. H., Kabra, A. N., … Jeon, B. H. (2013). Cultivation of microalgae species in tertiary municipal wastewater supplemented with CO2 for nutrient removal and biomass production. Ecological Engineering, 58, 142–148.

TD-P_08 LIFE + MANEV – Evaluation of manure management systems and treatment technologies in Europe

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1. Objectives

The LIFE+ MANEV project aims at unifying the know-how on manure treatment options and technologies available in the market in all Europe for closing the gap between technological-scientific development and its application in daily practice in the sector.

The final objective is to help the stakeholders to determine the manure management system that fits best into a specific situation from the environmental, economic, energetic, agronomic, legal, biosecurity and social points of view.

2. Methodology

The environmental impact of high density intensive pig farming zones includes qualitative deterioration of air (ammonia, nitrous oxide and methane), water (by nitrates, phosphates and organic matter) and soil (phosphates and heavy metals) and the odour nuisance which is sometimes a major factor owing to the pressure of public complaints. Furthermore, there are also the related disease risks especially when pig slurry is incorrectly handled [1].

In Europe there are a wide number of technologies aiming at the improvement of the manure management. Although there is not one universal solution for manure management, each treatment can represent a good strategy in some areas, especially in nitrate vulnerable zone (NZV) and nutrient surplus areas. The application of one or other technology will depend on the farm size, local geography, land type, climate and production method that give rise to farms with highly individual features [2].

The use of an appropriate decision support system (DSS) could assist livestock operators in identifying manure management systems or components that have a positive impact on the reduction of GHG emissions. It has been observed that most computer-based decision support systems for manure management are mainly focusing on nutrient management. Very few have considered the whole farm manure management options. Thus, an overall evaluation of a manure management system based on defined criteria is required for the benefit of livestock producers as well as society in general [3].

2.1 Common Evaluation and Monitoring Protocol (CEMP):

The first action carried out in the project was to define a protocol for the evaluation and monitoring of the treatment plants in Europe. The objective was to unify the criteria, parameters and indicators of monitoring and evaluation of the different treatment technologies and manure management systems with the aim of obtaining comparable data and results.

Seven criteria were born in mind: environment, energy, economy, agronomy, social, biosecurity and legislation (table 1).

The boundaries of the systems assessed ranged from the farm storage tank and the final destination of the end products: cropland fertilization, exportation to other areas or discharge in watercourses (if possible).

The functional unit of the assessment was one m^3 of slurry or one t of manure.

Table 1: Criteria, indicators and parameters established in the CEMP.

Criteria	Indicators	Parameters
1. ENVIRONMENT	Water: Eutrophication Air: Acidification Air: Global warming Soil: Salinity Soil: Metals	N balance, P balance NH_3, SO_2 emissions CO_2, CH_4, N_2O, NO_x emissions Electrical conductivity Cu, Zn
2. ENERGY	Production Consumption	Electricity production, heat production Energy consumption
3. ECONOMY	Incomes Expenses	Energy production, end-products Depreciation, energy consumption, chemicals, maintenance, manpower
4. AGRONOMY	Fertilizing units NPK	NPK balance
5. SOCIAL	Odour Noise Visual impact Social activity impact	Reference values Reference values Height, distances, population,… Jobs created
6. BIOSECURITY	E. Coli Salmonella	Reduction / no reduction Reduction / no reduction
7. LEGISLATION	European legislation National legislation Local legislation	Compliance Compliance Compliance

NPK-Nitrogen, Phosphorus, Potassium

2.2 Monitoring and evaluation of treatment technologies at full scale

Based on the CEMP, 13 treatment plants (formed by several treatment technologies) currently working in different regions of Europe (Spain (Aragón, Catalonia, Murcia, Castilla y León), Italy (Lombardy and Emilia Romagna) and Denmark (Midjytland)) have been monitored (table 2). The aim was to get a better understanding of their performance and impact, bearing in mind all the conditions that affect their daily functioning.

Table 2: Technologies monitored in the project.

Number of treatment technologies monitored	Treatment technologies	Type
9	Anaerobic digestion	Mesophilic Thermophilic
2	Composting	
8	Separation	Centrifugate Screw press Decantation Flotation Coagulation / flocculation
6	Aerobic biological treatment	Nitrification / Denitrification (N/DN) Sequential batch reactor (SBR) Sharon process
2	Ammonia stripping	
1	Pasteurization	

2.3 Scientific review

A thorough review of published knowledge has been carried out in order to define the state of the art of the different treatment technologies working in Europe, gathering information from the main scientific papers, official data bases (IPCC, EEA, Eurostat, FAO…) and experimental data.

3. Results and discussion

All the work developed within the project has been materialized in a software tool available on line on the website of the project (www.lifemanev.eu) named MANEV tool. This tool provides support for decision-making of the stakeholders helping to identify which manure management systems fulfil the necessities of their agricultural scenario.

This tool simulates and assesses the effects of the eventual implementation of a slurry management system in a specific context: one or various livestock holdings and related agricultural plots taking into account their specific characteristics (climate, regulation, economy ...). The comparison of different simulations in the same scenario allows the user to make a decision on his/her best choice (figure 1).

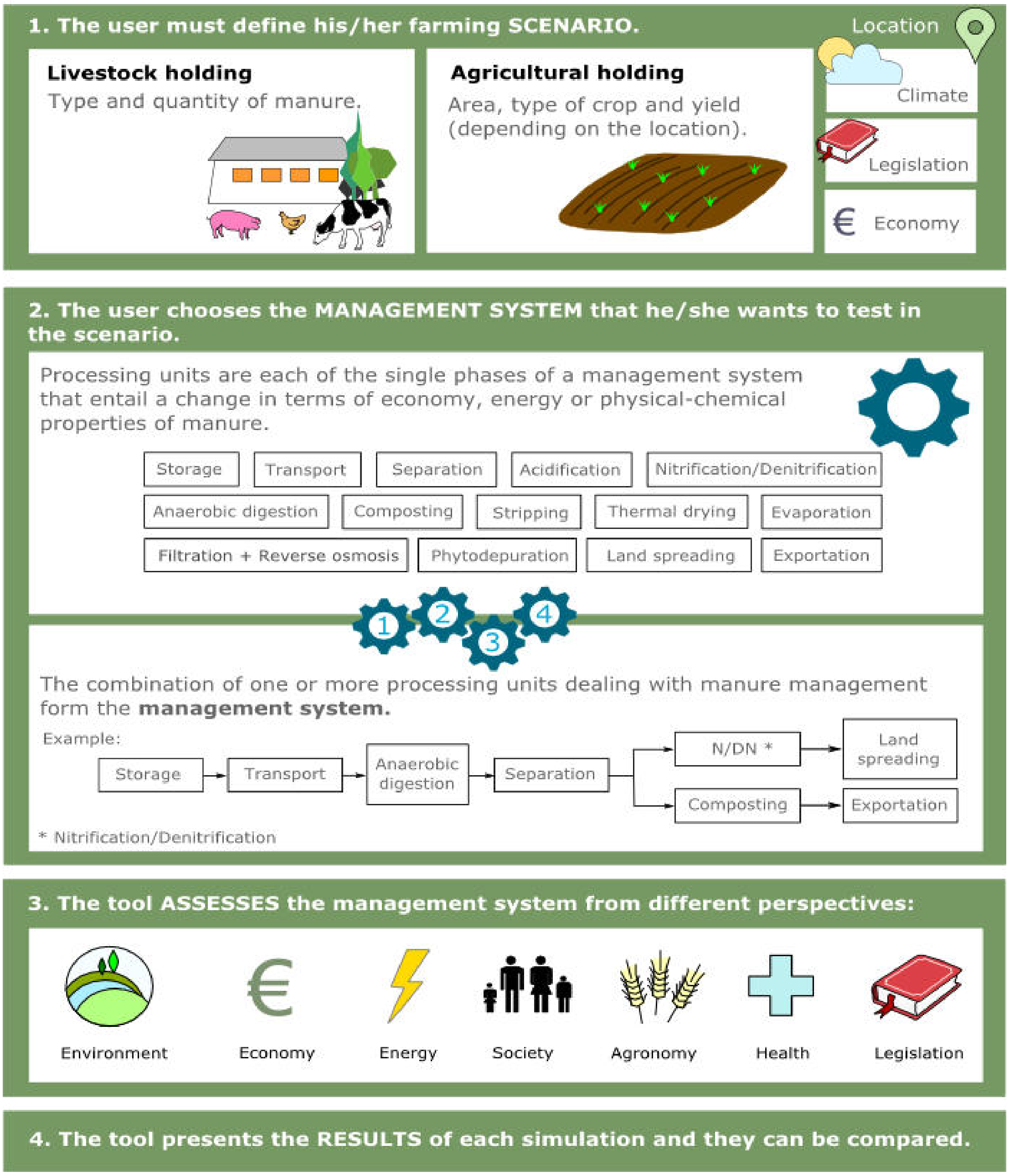

Figure 1: Scheme of the works carried out by the Manev tool.

The assessment is based on algorithms that were developed by project partners from information found in bibliography, experimental data and recognized international organizations. The results have been validated with the data obtained in the monitoring of the different treatment technologies.

The fact that the evaluation carried out by the MANEV tool addresses the main features involved in the manure management makes it interesting for all kind of stakeholders: the livestock holders worried about complying with the legislation minimising the management cost, the agricultural holders interested in a good and cost-effective fertilizer, the local and regional administrations ensuring the environmental protection, the technological companies concerned with the development of new technologies increasing the performances.

4. Conclusion and outlook

Although there are different technological options to treat manure, the selection of one or other will depend on the characteristics of the agro-farming scenario.

A deep knowledge of the technologies leads into a better identification of a correct manure management. Consequently, the implementation of these management alternatives will improve the environment and enhance the sustainability of the sector fulfilling with the legal requirements.

The MANEV project unifies the know-how on the main technologies available for the manure management currently working at full scale by means of homogeneous assessment based on a common protocol (CEMP). The MANEV tool gathers the state of the art of the different technologies and manure treatment systems, putting all this knowledge at the disposal of every stakeholder for their profit with the aim of minimising the environmental impacts and strengthening the livestock sector in Europe.

Acknowledgements

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References

[1] Burton, C.H., Turner, C. 2003. Manure Management. Treatment strategies for sustainable agriculture. 2nd Edition. Silsoe Research Institute, Wrest Park, Silsoe, Bedford, UK. 451 pages.

[2] García-González, M.C., Riaño, B., Teresa, M., Herrero, E., Ward, A., Provolo, G., Moscatelli, G., Piccinini, S., Bonmatí, A., Bernal, M.P., Wisniewska, H., Proniewicz, M., (2015). Treatment of swine manure: case studies in European's N surplus Areas (Scientia Agricola (under review)).

[3] Karmakar, S., Lagüe, C., Agnew, J., Landry, H. (2007). Integrated decision support system (DSS) for manure management: A review and perspective. Computers and Electronics in Agriculture 57, 190–201.

Thematic Area TE – Sustainable regions
(Oral presentations)

Christiane Lüdtke: Art from Tetrapak

TE-K New paradigms on how to achieve zero food waste in future cities – Optimizing food use by waste reduction and valorization

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1. Context

Cities currently manage uneaten food and other food system based biowaste quite inefficiently. The organic compound, despite its high nutriment value, is only to a small extent recycled and returned to farm soil and therefore, does not contribute to closing ecological nutrient cycles and to supporting sustainable food production [1]. In the US, over 97% of food waste is estimated to be buried in landfills [2]. Forkes has shown for Toronto that only 4.7% at most of food waste nitrogen (including sewage waste) was recovered and/or recycled [3]. For Paris and its suburbs, a similar estimate has been obtained, and this share of nitrogen food waste recycling has been in steep decline in the course of two centuries, from 40% to close to 5% estimated for today [4]. One study has analyzed the nutrient balance (N, P) for Bangkok Province [5]. These studies mainly focus on food waste and sewage waste management from a nutrient recycling point of view.

Furthermore, food waste related resource use and environmental pollution are highlighted as no longer acceptable in the context of global warming and increasing pressure on the planet's limited boundaries [6], [7]. According to an analysis from the Waste & Resources Action Program (WRAP), prevention of 1 ton of food waste can yield in carbon equivalent savings of 3090 kg when food from manufacture or retail is redistributed to people. But savings are much lower when food from manufacture is redistributed as animal feed (220 kg eq CO_2/ ton food) or used for anaerobic digestion (162 kg eq CO_2/ ton food). This analysis illustrates from a climate point of view priority for food waste prevention over food waste valorization.

The problem of food waste is crucial: the FAO estimates that one third of world food production is lost or wasted. In industrialized countries, food waste amounts to close to 300 kg/cap/year in North America or Europe – and more than two third of it occurring at distribution, catering and in-home consumption [8]. A "preparatory study on food waste across the EU 27 Member States" estimates annual food waste generation in the EU27 at approximately 89 million tons, or 179 kg per capita (without agriculture) [9]. Households (42%) and manufacturing (39%) have been identified as the most important food waste producers, followed behind by food services/caterers (14%) and retail/wholesale (5%). The high share of food waste occurrence close to consumption, cities' dense population and the accumulation of waste in periurban areas, together with the numerous socio-technical initiatives coming from both urban citizens and stakeholders are all factors that place cities as important players. Although food waste in cities in Asia, Africa and South America is relatively lower at the downstream stages of supply chains, the fast growing population and changing habits towards urban diets nevertheless raise the question also for these sets on how to optimize food use in cities. The world population is going to become more and more urban, being expected to make up 66% of the world population by 2050 compared to 30% in 1950. Ongoing population growth together with urbanization is expected to increase the urban population predominantly in Asia and in Africa. Today, the most urbanized regions include Northern America (82%), Latin America and the Caribbean (80%) and Europe (73%), but all regions in the world are projected to urbanize further [10].

Our study analyses the specific link between food waste and cities in a zero waste perspective in the future. By using a foresight approach we suggest to identify and discuss key prevention and valorization measures, to pinpoint knowledge gaps on the specific character of food waste in cities and to bring up relevant questions for research.

2. Objectives

Objectives of our work are twofold: i) identify high potential socio-technological innovations in food waste prevention and valorization and ii) extract research questions contributing to fostering and accompanying cities' breakthrough strategies towards zero waste sustainable food systems, specific to different urban settings worldwide (covering both industrialized and unindustrialized areas). Data collection on urban food waste flows was expected to support the analysis, but could not be accomplished in the course of this study as this kind of data has not been available at urban scale so far, a point that literature on urban metabolism applied to case studies confirms [11], [1]. Our analysis is based on examples most covered by media of food waste prevention and valorization initiatives from European, American or industrialized Asian cities. The lack of literature and of media attention makes us not consider the situation of cities in less advanced countries, although we have been reminded by experts that there exist similar initiatives as well.

3. Methodology

We define food waste as "any food with is edible and inedible parts which leaves the supply chain meant for human consumption". Included are uneaten food (edible and inedible parts), by-products, organic solid waste and effluents from food processing, preparation and distribution (food industry, caterers, restaurants, shops and artisanal food producers). Sewage waste however has not been considered in this study.

We used a definition of urban scale close to the concept of functional urban regions . Not political or administrative indicators determine "urban scale", but the influence on activities and prescription of these (for example on farming practices or on waste management) coming from the city. Urban scale according to our definition can therefore include non-urban areas on which the city bears influence.

Twenty experts related to disciplines and fields of interest like industrial ecology, urban metabolism, urban farming, aquaculture systems, waste recovery and processing techniques, law, ethics, system innovation and foresight studies were organized as a working group following a foresight study approach. A literature review, stakeholder interviews (food sector companies, waste management companies, NGOs, local governments) and 5 workshops with the group of experts were being conducted from September 2014 to June 2015.

The 5 workshops allowed:

- The definition of precise objectives and methods of the study (definition of food waste, boundaries of urban food systems, foresight study approach),
- The validation of the method and discussion of a state-of-the-art working paper on initiatives in food waste prevention and valorization,
- The organization of initiatives into broader categories of prevention and valorization measures; combination of measures into coherent food waste reduction scenarios within one business-as-usual and two alternative (green economy, "alter") food system contexts;
- Set-up of these scenarios according to a typology of different urbanization dynamics (retrieved from the Agrimonde Terra foresight study) [12];
- Identification of key measures expected to have high potential in food waste reduction; multi-criteria analysis of these key measures and identification of issues for research.

4. Results and discussion

4.1 Characterization of food waste prevention and valorization measures

Stakeholder interviews and literature review have shown that innovative approaches in food waste management are abundantly experimented worldwide (for example in Canada, the USA, UK, France and other European countries). Table 1 provides an inventory of the different food waste prevention and valorization initiatives collected in the course of this study.
Food system actors involved are as different as business and catering companies, civil society, NGOs and municipalities.
However, the specific link between these initiatives and urban systems is difficult to identify. Indeed, some of them have been mentioned in relation to cities (for example in Seattle, in New York) which have been known for elaborating an urban food system approach. However it has not become clear from the analysis whether the initiatives' occurrence is specific to a city's context or whether they could be occurring as well in rural areas. Furthermore, literature on the topic of food waste prevention has not been found specifically applied to urban settings.

Table 1: Inventory of food waste prevention and valorization initiatives, classified by supply chain stage.

Supply chain stage \ aim	FOOD WASTE PREVENTION	FOOD WASTE VALORIZATION
Farming	- Better matching of supply and demand due to proximity of consumers with farming (including business and community-based urban farming) - Genetic selection of raw material - Information availability and access - Field gleaning - Urban garden gleaning - Connection between food donors and receivers - Marketing of surplus or of food not according to standards	- Industrial synergies - On-farm composting and anaerobic digestion
Processing	- Optimization tools for stocks - More resistant packaging - Donation to charities - New products from discarded food	- Animal feed
Logistics	- Better transport packaging	- Inverse logistics to return food waste to farmers for composting or soil amendment
Distribution	- Better planning and shelf management - Collaborative management of stock and orders to match both better - Adjustment of sales promotions and sales campaigns - Adjustment/deletion of expiry dates - Stock clearance operators - Tools for matching demand and supply - Processing and delivery at order - Reduced transport duration - Donation to charities - New products from discarded food	- Animal feed - Conversion to energy
Catering, restaurants	- Incentives to leftover reduction - Price reductions close to shop closing hour - Management tools - Adjustment of plate sizes to clients' appetite - Removal of lunch tray at self-service (canteen and restaurant) - Meals based on "inedible" parts of food - Doggy bags	- Frying oil recycling - Collection for composting and anaerobic digestion
Households	- Changes in food related habits (e.g. use of a shopping list) - Changes in use of preservation techniques (cupboard organization, freezing, use of canned food, etc...) - Donation - Distribution of food amongst other households	- Use of hens for recycling kitchen waste - Individual or community composting or drying
Technologies	- Higher yielding technologies at processing - Longer shelf-life - Remaining shelf-life indicator - Analysis and monitoring of consumption data (big data) - For food use (new products, extraction of valuable compounds for food)	- For feed use - Biorefinery - Drying of organic waste - Composting - Anaerobic digestion of food waste
	Overarching tools	
Politics & regulation	- Landfill ban for organic waste - Waste taxation - Obligation in handling of organic waste (separate collection, valorization, …) - Changes in regulation of product standards - Obligations for food donation to charities - Reduced tax payment schemes due to food donation to charities, recovery of VAT - Certification, labelling - Support to the use of hens as food waste recyclers	
Awareness raising, education, training to professionals	- Awareness raising campaigns - Recommendations for food waste reduction (households, catering, …) - Education and training - Networking tool and exchange of experience	

4.2 Key measures for food waste prevention and valorization in urban settings

Food waste prevention strategies mainly use communication and awareness raising tools, whereas valorization of food waste mainly aims at energy recovery so far. However, coherent concepts and strategies linking different initiatives are currently missing in most case studies, while single initiatives on food waste prevention and valorization are abundant and using manifold tools (regulation, technology, social innovation).

Based on the initiatives collected for this study, experts have extracted key measures expected to be particularly efficient in food waste reduction and interesting to copy in many different settings.

High potential key measures

- Education of public and training of professionals
- More flexible supply chain specifications,
- Collaborative use of data, flow monitoring and smart sensors,
- Regulation, taxation and financial tools,
- Gradual withdrawal of food from market, selling off, stock clearance, on-site processing and donations,
- Breakthrough manufacturing and packaging technologies,
- Urban practices and urban planning,
- Biomass valorization and biorefinery,
- Fair distribution of responsibility between stakeholders

Key measures identified in the previous stage have been discussed according to a multi-criteria approach both with the experts and during stakeholder interviews. For each measure, analysis key points have been made available as much as possible to show performances, forces and drawbacks. For instance, for the key measure « collaborative use of data, flow monitoring and smart sensors » it has been discussed that a kind of ecosystem of data sharing (production capacity, stock and sales capacities) between stakeholders would be feasible which would enable to produce just-in-time and to diminish food waste. Technologically, a data exchange platform would be possible as well. However, even if the technology is available, its implementation is far from reality. Stakeholders argue that the data is confidential and falls within the scope of data secret; it is therefore difficult to obtain. Furthermore, the economics and profitability of data sharing remains to be proven. Data protection and data ownership are discussed. These obstacles as to the extension of collaborative use of data, flow monitoring and smart sensors are considered highly relevant by stakeholders and experts.

4.3 Perspectives for research on the link between cities and food waste prevention and valorization

Based on the multi-criteria analysis of food waste reduction measures (prevention and valorization), some generic research perspectives specific to urban contexts have been identified:

- **Innovative Logistics** needs to be adapted to urban requirements and limits. The logistics of waste collection and recycling translates into small-scale organized flows. For food waste reduction, close monitoring of supply/demand could entail more frequent delivery of smaller amounts to shops leading to overall higher environmental impacts and congestion. Therefore, innovative logistical networks including supply and food waste collection need to be developed. Furthermore, reversed logistics needs to be considered, for example in urban planning, in order to allow for increased efficiency in the collection. One reason is that recycling infrastructure needs time to become profitable and depends on stable input from waste.
- The **question of scale** is relevant not only for logistics but as well for processing. To what extent should processing and logistics be down-scaled to urban small-scale?
- **Robustness of the supporting system: C**ities services are expected to be increasingly supported by digital information and communication technologies. Smart cities may provide opportunities for improved management of the supply and demand matching, for example by the means of collaborative tools. Yet, the supply of abundant energy may no longer be the unique reference scenario in any case. How can current strongly technology

and energy reliant food systems become adjustable to a scenario of instability? What role for alternative preservation options to the cold chain?

- **Urban planning:** How can urban planning back or prompt food waste prevention and valorization measures? What action can be taken by the municipality? Municipal decisions on urban planning can for example include requirements on the nature and share of city infrastructure or surface dedicated to the purpose of food waste reduction.
- **Waste for agriculture**: what is the stakeholders' perception on the "waste character" of organic matter from food waste, an obstacle or an opportunity? Contamination potential for the soil and the water?
- **Social acceptability** of innovation in food waste prevention and valorization in cities. For example social acceptability of livestock in cities, of new packaging and processing technologies for food items, for digital technology based city management?
- What **public health issues** can be relevant (for example infectious diseases from animals and allergies)?

5. Conclusion and outlook

Cities today are acting as laboratories for socio-technological innovations in food waste prevention and valorization, yet coherent concepts and strategies involving the different actors are missing. Technological and cultural challenges remain to be overcome, for example the analysis of "big data" to support alignment of supply and demand, the mutual share of information and joint planning of food supply, and societal acceptance of new technologies. Overall, data on food supply, consumption and food waste flows in cities in order to conduct material flow analysis are challenging to obtain.
In a next step we are going to run fieldwork on food waste flows in four cities (Dakar, Chicago, Antananarivo and Montpellier) to contribute to closing this data gap and to progressing on the urban metabolism approach applied to food systems.

The inventory of food waste prevention and valorization measures doesn't really take into account the diversity of initiatives in cities in the world's less advanced countries. A next step would be to repeat the analysis in developing countries, where population growth rate is projected to be highest in the future. This complementary analysis could be interesting because of the wide diversity of practices observed and claimed by the experts. New tools should be developed in order to access information in those countries despite the lack of literature, such as interviews with local industrials for example.

Key measures for food waste prevention and valorization should be analyzed more deeply and challenged by further stakeholder discussions.

References

[1] Barles, S. (2009): Urban metabolism of Paris and its region, Journal of industrial ecology 13, 898–913
[2] Levis, J.W.; Barlaz, M.A.; Themelis, N.J.; Ulloa, P. (2010): Assessment of the state of food waste treatment in the United States and Canada, Waste management, 30, 1486–1494
[3] Forkes, J. (2007): Nitrogen balance for the urban food metabolism of Toronto, Canada, Resources, Conversation and Recyling 52, 74-94
[4] Barles, S. (2007): Feeding the city: food consumption and flow of nitrogen, Paris, 1801-1914, Science of The Total Environment 5, 375(1-3), 48-58.
[5] Farge, J.; Magid, J.; Vries, F.W.T.P. de (2001): Urban nutrient balance for Bangkok, Ecological Modelling 139 (1), 63-74
[6] WRAP (2015): Strategies to achieve economic and environmental gains by reducing food waste, Banbury, 61 pages
[7] Kummu, M. ; de Moelb, H. ; Porkka, M. ; Siebert, S. ; Varis, O. ; Ward, P.J. (2012): Lost food, wasted resources: Global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use, Science of the total environment 438, 477-489
[8] Gustavsson, J.; Cederberg, C.; Sonesson, U.; van Otterdijk (2011): Global food losses and food waste: Extent, causes and prevention, Rome, FAO, 29 pages
[9] Monier, V.; Shailendra, M.; Escalon, V.; O'Connor, C.; Gibon, T.; Anderson, G.; Hortense, M.; Reisinger, H. (2010): Preparatory study on food waste across the EU, Technical Report, 54, 213 pages
[10] United Nations (2014): World Urbanization Prospects, The 2014 Revision Highlights. New York
[11] Goldstein, B.; Birkved, M.; Quitzau, M.-B.; Hauschild, M. (2013): Quantification of urban metabolism through coupling with the life cycle assessment framework: concept development and case study, Environmental Research Letters 8, 035024
[12] Lançon, F.; Mora, O., Aubert, F. (2014): L'extension urbaine à travers le monde : enjeux pour les villes et les campagnes. Cahier Demeter « Agriculture et foncier », number 15, 43-58

TE-O_01 Appropriateness and potential of large-scale composting initiatives in developing and transitional municipalities

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1. Objectives

Composting is a popular option with municipalities in developing and transitional countries to upgrade collect-to-landfill waste management schemes [1]. Intensive urbanization trends and the related growth in organic waste production are the main drivers for the introduction of this method [2]. The current critical review describes relevance and potential of composting in asset-poor municipalities to propose key indicators for the assessment of its local resilience.

2. Methodology

A multidisciplinary review assessed appropriateness of large composting initiatives in waste management and nutrient recovery in developing and transitional municipalities. The regions (i) Sub-Saharan Africa, (ii) India and South-East Asia, (iii) the Middle-East, (iv) Latin America and (v) China were studied. Key factors indicating appropriateness and resilience were identified by viewing (i) resource recovery potential, (ii) operational & technological resilience, (iii) institutional framework & management, (iv) economic performance and (v) public perception. Indicators were chosen based on objectiveness, direct relation to resilient operations & ease of acquisition in practice.

3. Results and discussion

3.1 Resource recovery considerations

The role of composting in resource recovery is limited by the amount of available organic wastes. The identified per capita production appears to be comparable in the viewed regions. In Asia, 0.6-1.0 kg/capita/day is estimated for daily municipal solid waste (MSW); for Kampala and other Ugandan municipalities, the per capita value is in the range of 0.4-0.9 kg/capita/day [3]. Comparable ranges were found for other African municipalities [4]. The largest fraction of this waste is organic. In Nairobi (Kenya) and Dar es Salaam (Tanzania), the organic fraction of solid waste (OFMSW) is shown to exceed 70 % [4,5]. OFMSW data from Chinese cities is somewhat lower (~50%), but in larger cities this value was shown to be higher, with OFMSW in Beijing being as high as 70% [6]. The high organics content in most developing and some transitional cities is wealth-related as low-income levels ensure most consumption to be food related. But limitations in (financial) assets also imply that municipalities may lack funds to collect all their MSW. This limits the potential of OFMSW composting. In West Africa, 10% (Benin) to 76% (Guinea) of the urban MSW is collected; other countries in the region feature rates around or below 40% [1]. In South-East Asia, 71% of all MSW was reported for Vietnam and 70-80% for the Philippines, respectively [7,8]. Transitional cities (in Latin America and China) exhibit higher (>95%) collection rates. However, data reliability (worldwide) is limited and illegal disposal further reduces availability organics for composting. [1,9].

Composting efficiently concentrates and stabilizes organic wastes [10,11]. In developing regions, a 40-70% volume reduction was possible to achieve [5]. The final product is expected to offer a valuable nutrient source, but its fertilizer value is debated. OFMSW composition estimates indicate N:P:K fractions of 1.3:0.8:0.9 % [10], but sources agree that waste-derived composts are to be seen more as soil conditioners than as fertilizers [5,11]. Despite their limited N supply, OFMSW composts offer emission-poor organic matter and a wide range of other nutrients. With their additional capacity to lower water need in soils (through the increasing of cation exchange capacity), they may be seen as potential contributors to food sovereignty in LDCs (least developed countries).

3.2 Operational capacity and technical performance

Limited data availability was apparent even at the level of large-scale installations in the viewed regions. The CDM (Clean Development Mechanism) project listing is likely to offer the most significant database [8]. According to this portal, CDM-supported large-scale plants are mostly implemented in Asia. Of the 16 initiatives, India and Vietnam are listed with 3-3 installations, China with 2 and Bangladesh, Pakistan, Singapore and Indonesia with 1-1 plant, respectively. Two initiatives from Latin America (Columbia) and two from Africa (Egypt and Nigeria) complete the registry. Typically, these initiatives process between 350-700 tpd (ton per day) OFMSW [8].
Outside the CDM database, the information on large-scale plants is very fragmented. A new composting plant in Mexico City is mentioned as the only large-scale initiative of the country. With an estimated capacity of 2500 tpd OFMSW (that treats 80% of Mexico City's OFMSW), it is possibly the largest plant in Latin America [12]. Composting is a common method in other Latin-American countries, but the only data found indicated 74 Brazilian initiatives with a processing range of 150-1,200 tpd. The most detailed report on a national composting sector originates from Egypt. This work summarizes 43 composting facilities in 27 governorates [13]. In the NEMA region, Saudi Arabian information mentions the existence of at least 30 composting facilities [14]. Other mentions of composting plants in this region include a 260 tpd installation in Tripoli (Libya) and a high-tech, commercial, agro-industrial waste processing plant in Israel [15]. In Sub-Saharan Africa, Ghana appears to possess one large plant (Zoomlion). A national-level program for Uganda cites a growing number of installations with a processing capacity of 50-200 tpd, designed to treat about 40% of the OFMSW in their respective municipalities [8].
Difficulty in obtaining more information may partly be the result of the significant rate of failed or sub-optimally operating plants. As an example, 9 out of the 11 compost-processing plants had to close down in India in the period of 1974-1996. In a more recent example, the OFMSW composting capacity of ~6000 tpd by four plants (Delhi, India) shrunk to just 500 tpd from a single plant between 2004-2006 [8]. Brazilian studies indicate that several major cities engage (d) in OFMSW composting in the past. In Brazil, 36 out of an initial 54 operations closed in the '90s [16]. Many surviving plants appear to produce compost at a fraction of their original design load. In Egypt, a study revealed an average plant output of 37% when compared to the initial design output [13].
The key reasons of failure or limited utilization appear to be similar for the different regions. The main obstacle is confirmed to be the application of high-tech, developed designs, often in combination with too short processing times [13, 16, 17, 18]. Within just a few years, such high-tech designs were replaced with simplified, local protocols in e.g. Egypt and Brazil. The Egyptian unified protocol also emphasized land provision by the recipient governorates. In their design, compost production followed a mixed waste collection process and pre-sorting at the plant. No forced aeration, but (daily) turned windrows were the technology of choice. Operations also had a ripening phase and a final sieving with a rotating drum or similar device. Other factors reducing resilient operations were the (i) high dependence on the quality of waste sorting (with a key emphasis on heavy metal contaminations), (ii) lack of a sufficient uptake market, (iii) difficulties in obtaining skilled personnel, (iv) inappropriate plant locations and (v) the limitations to electricity and water supply.

3.3 Institutional framework and management

Lack of detailed institutional framework and type of ownership was mentioned as main bottlenecks to the resilience of technology-intensive facilities in Brazil [16, 17]. Lack of specifications on the role of composting and its potential for nutrient recovery are apparent in most other countries as well. The focus of legislation on waste reduction implies that composting is not seriously regarded for its role in nutrient recovery. As a result, policy frameworks neglect defining protective measures (financial or legislative) to safeguard operations and compost distribution. Studies imply that privately-owned installations have a better chance to survive in this environment, likely because their operations had a superior focus on matching characteristics of market demand [13, 16].
Existence of waste treatment alternatives, especially in the form of incineration plants, appears to limit composting appropriateness. Until the mid-90's, China had a growing composting sector for OFMSW treatment. At its peak, it handled 20% of the produced MSW against a <2% of incineration treatment. Since then, this trend is reversed and composting is indicated to treat not more than 1-2 % of all MSW in Chinese municipalities [6, 19]. Sources point at the lack of political measures to stop the regression in number of plants. Despite the general support in national-level 5-year plans, no specific program seems to exist to safeguard composting. As a result, incineration and to some extent landfilling have replaced this method in Chinese OFMSW management [6].

Systemic integration of composting in policy frameworks through e.g. a simplified licensing procedure [20] could support an increased role and appropriateness of large-scale composting. Just as in developed countries, it is expected that adequate legislation (and its efficient enforcement), innovative waste separation solutions and financial drivers (e.g. subsidies, landfill taxes or even a landfill ban) may be necessary to make composting initiatives more resilient.

3.4 Economic considerations

Affordability of technology-intensive, large-scale operations depend on their profitability, which relies largely on product sales. This is a weak feature of operational sustainability in practically all composting projects. In developing regions, the primary reason of low profitability lies in the quality of the produced composts. OFMSW-derived compost is perceived to be inferior both to animal manures and artificial fertilizers. This has to do with the lower N concentration of OFMSW compost, the sometimes considerable contaminations (plastic, glass, heavy metals) and widespread subsidizing of artificial fertilizers [18]. A seasonally changing demand may also affect compost producers, as composts are in highest demand prior to the growing periods. Where proper, long-term storage is not possible, the rapidly reducing N content results in a quick depreciation in compost price and demand. These limitations can be reduced by improving process management so that the end-product features little impurities and a constant quality in time.
A second feature of developing composting initiatives considers the apparent lack of additional financing. In developed regions, gate fee-type arrangements ensure that composting plants get double financing for the production of OFMSW compost. Such a regulation is intended to acknowledge the role of plants in reducing organic waste streams. In practice, the gate fee income is indicated to be the main reason of profitability in developed countries [21,22]. In its absence, developing region initiatives need to rely on compost sales and in some cases collection and transport fees for their income. It is proposed that financial resilience in composting plants requires a specific legislative support – possibly in the form of a gate fee for the financing of their treatment. A less crucial, but frequently mentioned aspect of the financial viability of plants is indicated to be the high-cost leasing of their sites and the transport expenses of the sold product [23]. Such costs may be a significant burden on the low margin of profit achieved.

3.5 Public perception

Public perception is key in maintaining operations as contaminated products and odor nuisance were proven to reduce market demand in e.g. China and Brazil [6,16,17]. Strong smell from composting plants has even led to plant closures [16]. Next to proper measures to reduce odor from the composting process, the siting of the installation is crucial. Because siting depends on the smallest distance to both waste suppliers and potential markets, a sufficient distance from inhabited areas needs to be considered [23]. This phenomenon was not only confirmed for developing locations, it was reportedly the key factor in the numerous plant closures in e.g. the United States [11].

3.6 Proposed objective criteria for a resilience-based assessment

Eight objective factors are proposed that are easy to access and may have important implications on local appropriateness. They are to serve as indicators in areas with seriously limited data availability. Their interpretation should consider local context and may therefore vary per scenario.
Feedstock composition: An objective factor that is either directly available or through a small survey. Seasonal changes in compositions should be considered. This factor carries indicates product quality and potential applications. This factor is most relevant for countries where process and product quality requirements are not defined in detail.
Collected waste amount: An objective factor that is easy to obtain either directly or through known waste collection rates and per capita waste production volumes. It is a key parameter that suggests the optimal design size of a facility. Care should be given to the potential future improvement in waste collection as large amounts of OFMSW may become available for treatment.
Existence of (at least wet-dry) waste separation: Objective factor that implies potential product quality and to some extent the product demand. Quality of separation is should be considered.
Existence of alternative treatment methods: The existence of incineration, anaerobic digestion or similar methods in the area of implementation is easy to identify. This factor is most relevant for transitional areas as Latin America, China and possible the Middle East as implementation of (sophisticated) waste processing alternatives are more likely than in asset-poor, developing areas.

Specific legislation for composting: A less objective factor, but very important in assessing resilient operations for new or existing plants. As sub-factors, (i) existence of a gate fee or similar mechanism, (ii) a landfill ban or tax and (iii) subsidy of artificial fertilizers should be considered.
Existence of a significant compost uptake market: This is a less objective factor that may also require a detailed assessment study. It is included, because the potential uptake market for the compost is perhaps the strongest indicator to nutrient recovery and profitability of the installations.
Appropriate technical sophistication: A less objective indicator that can be best assessed based on the available information of past initiatives in the project area. Appropriate technical sophistication will differ per region and change in time with economic and technical progress.
Risk of odor nuisance: An objective factor, sometimes already integrated in the specific composting-related legislation. The review indicates a good relevance for public acceptance.

4. Conclusion and outlook

Appropriateness and potential of composting initiatives in developing and transitional municipalities was reviewed. A multidisciplinary, international approach was chosen to mitigate effect of restricted data availability. Comparable experiences from different regions confirm that municipal composting is a relevant option for reduction of OFMSW, but their current role in nutrient recovery is very low. Despite the obvious need to recycle the increasing flux of OFMSW to agricultural soils, the reviewed initiatives often cope (d) with multiple challenges. Limited profitability, mismatch between local infrastructure and technical design, underdeveloped policy frameworks and restricted compost demand are shown to be the key areas of improvement. The proposed objective indicators can help identifying resilient designs, but their interpretation needs to consider local context. Addressing of the listed indicators is expected to reduce the global nutrient recovery challenge through the supporting resilient implementation of locally appropriate designs.

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References

[1] Hoornweg, D., Bhada-Tata, P. (2012) What a waste - A Global Review of Solid Waste Management. Urban Development Series, Knowledge Papers, No.15. World Bank, Washington D.C., USA.
[2] Hargreaves, J.C., Adl, M.S., Warman, P.R. (2008) A Review of the Use of Composted Municipal Solid Waste in Agriculture. Agriculture, Ecosystems and Environment (123) p1-14.
[3] CDM (2014) CDM project cycle search registry. http://cdm.unfccc.int/Projects/projsearch.html
[4] Palczynski, R.J. (2002) Study on Solid Waste Management Options for Africa. African Development Bank, Sustainable Development & Poverty Reduction Unit. On: http://www.afdb.org/, 08-22-2008.
[5] Mbuligwe, S.E. et al. (2002) Potential and Constraints of Composting Domestic Solid Waste in Developing Countries: Findings from a Pilot Study in Dar es Salaam, Tanzania. Resources, Conservation & Recycling (36) p45-59.
[6] Balkan, E. (2011) Composting in China: RIP? On: needigest.com/2011/10/12/composting-in-china-rip/; 14-02-2014.
[7] Nguyen, T. (2006) Solid Waste Management in Vietnam. On: http://www.seas.columbia.edu/ ; 30-12-2013.
[8] CDM (2014) CDM project cycle search registry. On: http://cdm.unfccc.int/Projects/projsearch.html
[9] Karanja, A. (2005) Solid Waste Management in Nairobi. Actors, institutional arrangements and contributions to sustainable development. PhD Thesis, Institute of Social Studies, the Hague, the Netherlands.
[10] Lardinois, I., Klundert, A. van de (1994) Recycling Urban Organics in Asia and Africa. Biocycle, 06-1994.
[11] Haug, R.T. (1993) The Practical Handbook of Compost Engineering, Lewis Publishers, Boca Raton, Florida, USA.
[12] Villagran, L. (2012) Mexico City launches massive composting project. Smart Planet, issue 11. On:
http://www.smartplanet.com/blog/global-observer/mexico-city-launches-massive-composting-project/; 14-02-2014.
[13] Abt Associates (2003) Evaluation of composting facilities in Egypt. On: pdf.usaid.gov/pdf_docs/Pdaca202.pdf.
[14] Al-Turki, A., Al-Romian, F., El-Hadidi, Y. (2013) Evaluation of compost quality produced in Saudi Arabia. Lambert Academic Publishing.
[15] BioCycle (2010) Composting Biosolids Into High Quality Agricultural Product (Israel). BioCycle Vol 51 (10), p39.
[16] Schueler, A.S., Mahler, C.F. (2003) Composting in Brazil. On: http://www.getres.ufrj.br/ ; 14-02-2014.
[17] Neto, J.T.P. (1996) Composting: Experiences and Perspectives in Brazil. In: In: M. De Bertoldi, P. Sequi, B. Lemmes and T. Papi, Editors, The science of composting, Blackie Academic and Professional, Glasgow, Great Britain.
[18] UNEP/IETC (2014) International sourcebook on environmentally sound technologies for municipal solid waste management. United Nations Environment Programme – Intl. Environmental Technology Center, Osaka, Japan.
[19] Wei, Y-S, Fan, Y-B, Wang, M-J, Wang, J-S (2000) Composting and compost application in China. Resources, Conservation and Recycling. Vol. 30 (4), p277-300.
[20] Rashid, S.Z. (2011) Composting and Use of Compost for Organic Agriculture in Bangladesh. Proceedings of the 4th Intl. Conference for the Development of Integrated Pest Management in Asia and Africa (20-22 January 2011).
[21] ECN (2014) Country Profile – the Netherlands. European Compost Network portal. On: www.ecn.net; 12-02-2014.
[22] Amlinger, F., Peyr, S., Hildebrand, U., Müsken, J., Cuhls, C., Clemens, J. (2009) The State of the Art of Composting. Austrian Ministry for Agriculture and Forestry, Environment and Water Management.
[23] Zurbrugg, C. (2003). Markets for Compost – a Key Factor for Success of Urban Composting Schemes in Developing Countries. City Matters Magazine, online Journal of Urbanicity (http://www.eawag.ch/)

TE-O_02 Farmers attitudes and potential barriers to the use of new organic fertilizers

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1. Objectives

Many regions of Europe have a surplus of nutrients from animal manures, among them Denmark, which has led to nitrogen overload of soils and subsequent pollution of the aquatic environment [1]. To reduce nutrient input to agricultural soils, more processing of organic wastes (e.g. manure, urban organic waste) is needed, which can then be exported to regions without a surplus. This will result in novel organic fertiliser products that farmers may or may not be willing to apply to their fields. To increase farmers' acceptance of processing organic wastes, it is important to understand farmers' attitudes towards their use. The objective of this survey-based study was to identify farmers' perceptions of advantages and potential barriers for their use.

2. Methodology

The survey consisted of 24 questions and took approximately 10 minutes to complete. Farmers were asked about their attitudes towards a number of organic fertilisers that fell into one of three categories – unprocessed manures, processed manures, and urban organic waste fertilisers (see Table 1 for types). They were asked about their current use of these fertilisers, their interest in their use three years from now, the barriers to using organic fertilisers, and also the most important advantages or reasons for using organic fertilisers.

Table 1: The types of organic fertilisers considered in this survey. AD = Anaerobically-digested, TD = thermally-dried.

Unprocessed manure	Processed manure	Urban organic waste
Raw slurry	Mechanically separated slurry – Solid or liquid fraction	Biocompost from municipal solid waste
Animal urine	AD slurry	Raw/dewatered sewage sludge
Raw solid manure	Composted or TD manure/slurry	Processed sewage sludge (composted, AD or TD)
	Acidified slurry	Mineral concentrate e.g. struvite

Farmers to participate in the survey were selected by the following process:

- Farmer identity and address was obtained from the public 2011 Danish fertiliser accounts registry, covering all farms in Denmark using N fertilisers (around 43 000 commercial operations).
- Farmers with more than 10 ha of agriculturally farmed land were selected, in order to avoid non-commercial operations.
- 1 800 farmers were randomly selected from this list, and their records checked against the 2014 records to ensure that their business number (CVR) was still active.
- 1 585 farmers remained after this selection process, and this population was checked against original database to see if it was representative of the entire population, which it was.

Farmers selected for the survey were sent a cover letter, survey form and a pre-paid return envelope inside. Farmers were also given the option to fill out the survey online instead of returning the paper survey. Respondents within a certain deadline had the opportunity to enter in a prize draw.

3. Results and discussion

Out of 1 585 letters sent, 448 farmers responded (425 by letter, 23 online), a response rate of 28%. The respondents were found to be generally representative of Danish agriculture; the primary farming activity of respondents was close to those from the Danish farming statistics [2], with 62% primarily farming field crops, 12 and 9% dairy or cattle farmers respectively, 6% pig farmers, with other land uses being less important.

3.1 Current use of organic fertilisers and interest in organic fertiliser use three years from now

72% of respondents presently used at least one type of organic fertiliser. Farmers could specify more than one type of organic fertiliser: 67% used an unprocessed manure compared with 19% who used a processed manure and 8% who used an urban organic waste fertiliser. Organic fertiliser use increased with farm size; 60% of farms sized 10-20 ha used organic fertiliser compared to 85% of farms with more than 100 ha.

Farmers were asked about the organic fertilisers they were interested in using three years from now. Two-thirds of respondents were interested in using an organic fertiliser that was currently available to them, whereas 47% were interested in using an organic fertiliser not currently available to them. Underlying this, 27% of respondents were interested in unavailable unprocessed organic fertilisers, 42% were interested in unavailable processed organic fertilisers, and 23% were interested in unavailable urban organic waste fertilisers. These results indicated a significant unmet interest in organic fertilisers.

Despite a significant interest in using different organic fertilisers, farmers generally expected to use the same amount of organic fertiliser (in terms of total N applied to their field) in three years as they do today. Only 14% of respondents expected to use more organic fertiliser in three years than today, with 79% expecting it to remain the same and 7% expecting a decrease.

3.2 Barriers to organic fertiliser use and most important advantages or reasons to use organic fertiliser

Farmers were asked to rank the top three most important barriers to their use of organic fertilisers. Difficulty in planning for organic fertiliser use compared to mineral fertiliser ranked first, followed by uncertainty in NPK content, odour issues for neighbours, machinery to handle organic fertilisers being too expensive, and the difficulty in handling or spreading the product.

Farmers were also asked to rank the three most important advantages or reasons to use organic fertiliser on their farm. The advantage with the highest overall ranking was the improvement of soil structure by organic fertiliser. After this, advantages related to low cost, certainty in NPK content, little pollution and ease of application were perceived to be important. The reliability of NPK content was chosen as both a barrier and an advantage; farmers with experience of organic fertiliser use ranked certainty in organic fertiliser NPK content as a less important advantage, and ranked uncertainty in NPK content as a more important barrier. This indicated that farmers who had experience with organic fertilisers were more familiar with their limitations.

4. Conclusion and outlook

The survey showed that a majority of Danish farmers use at least one type of organic fertiliser. Although farmers generally did not intend on using more organic fertilisers in three years than they do today, there was a significant interest in organic fertilisers that are currently not available to these farmers, particularly processed manures. This indicates that in Denmark there is an unmet need for access to organic fertiliser processing technology and the resulting fertiliser products. Farmers considered the difficulty of use, uncertainty in NPK content, and cost of use to be main barriers to using organic fertilisers. Improvement in soil structure was considered the biggest advantage to using organic fertilisers; an advantage that mineral fertilisers cannot provide.

The insights obtained into the attitudes of farmers towards organic fertilisers and their use will help identify processing technologies and resulting organic fertiliser products acceptable to farmers.

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References

[1] Sutton, M.A.; Billen, G., (2011): Technical summary, in: The European Nitrogen Assessment: Sources, Effects and Policy Perspectives. Cambridge University Press, Cambridge, UK.

[2] Statistics Denmark (2014): Agriculture, horticulture and forestry, website accessed 15/05/2015, URL: http://www.dst.dk/en/Statistik/emner/landbrug-gartneri-og-skovbrug

TE-O_03 Management of wetland areas – Tradition & innovation for sustainable land use and network between rural and urban areas

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1. Objectives

Wetland areas are important for grassland management, soil protection, carbon storage, water regulation and biodiversity. With ongoing structural changes in agriculture, a great number are abandoned because their conservation requires lots of work. Management with conventional high tech equipment is often hardly possible due to highly water-saturated soils. Today's challenge is the search for sustainable management practices that are not time-consuming and have the same ecological impact as manual work. Use of work-horses can be an innovative approach for preservation of natural and cultural landscape.

2. Methodology

In 2013 and 2014 the project "Sustainable Wetland Management with Workhorses" was implemented in the regions Ausseerland and Enns Valley (Styria/Austria), where different Natura 2000 sites and other protected areas with a wide range of wet lands can be found (Hochegger, Mayer, 2014). The project was realized by the Agricultural Research and Education Centre Raumberg-Gumpenstein in cooperation with land owners, farmers, Natura 2000 site managers, the Styrian League of Nature Protection and the Work Horse Association.

Activities of the pilot project were:

- Pilot tests of wetland management with work horses through combining tradition with innovation by using modern, adapted equipment as well as the comparison of conventional agricultural machines and manual work with hand mower and scythe.
- Monitoring of potential impacts on biodiversity in relation to the method
- Analyses of demand & supply for additional services, second income, local recreation, tourism
- Elaboration of benefits for regional added value and socio-ecological approach
- Awareness creation for (traditional) ecological sustainable landscape management practices to keep landscape open with extensive landscape management
- Public relations

Different wetland areas were selected as trial plots. Some of them have not been managed for the last 30-50 years, some were only managed by manual work (motor mower, hand work). For the mowing trials with workhorses an adapted mower with double knives was used (figure 1). The trials were implemented with the horse breed Noriker, a medium-weight, sure-footed, strong and persistent mountain-carthorse with high potential for working and free time riding.

Figure 1: Mower with double knives: advanced technology, cutter bar is 2,2 m long, a three v-belts actuate the chain, very robust, does not plug up, best adapted for mowing meadows with horses, costs: about 6,800 € (Hochegger, Mayer, 2014).

3. Results

Horse power has many regional benefits. The strengths and weaknesses of wetland management with work horses are shown in table 1. The multiple capabilities of work horses are a chance for sustainable ecological agriculture in rural areas and other services and a connection between rural and urban areas for recreation and sports as well as practical experiences. New income possibilities for horse owners and breeders and therefore can be seen as a valuable contribution to strengthen the rural economy.

The main results of the implemented pilot project can be summarised as following:

- Mowing of wetland areas is important to keep landscape open and to protect biodiversity e.g. in wetland areas and to reserve green infrastructure
- Big and heavy machines cannot fulfil the management demands and requirements for protected areas and small structured areas
- The use of horse power is possible under difficult site conditions and has low impact on wet soils. It maintains the biodiversity of wet meadows and keeps cultural landscape open, through the prevention of forest and scrub encroachment. A pasture management with slopes up to 30% is possible.
- The use of workhorses supports small-scale grassland farming to protect cultural landscape. Furthermore it has many ecological benefits: minimal noise, no emissions from engines, no fossil fuels are necessary, animals like birds, deer, insects can easily escape
- The financial effort for technical equipment is low with big potential for technical evaluation and optimization, e.g. adaptation of the mowing equipment to sites with other conditions, different application possibilities.
- The hay can be used for innovative products or as litter for stables.
- Apart from mowing there are multifunctional possibilities for use of horses and can generate second income for horse owners and breeders (thinning of forests, clearing of bushes or dwarf shrubs, cultivation and harvesting of potatoes, structural improvement of meadows, support of touristic services like riding, coach tours, horse sleigh rides and environmentally-friendly transport activities (e.g. transport to cabins, removal of e.g. plastic litter in touristic village Bad Mitterndorf or waste from other municipalities).
- Working with horses has positive effects on children, teenagers, people with mental problems.
- The Austrian Environmental Program (ÖPUL 2015) offers compensations for horse owners in less-favoured areas, rare horse breeds like Noriker, use of horses in agriculture (e.g. pasture management and management of protected areas)

Work horses for mowing of wet meadows is a good alternative instead of hand mowing, no mowing and for small areas which are hard to reach but it needs subsidies (table 2). Also the removal of hay with a tarpaulin that is drawn by horses is more efficient as manual work (figure 2).

Table 1: Advantages and disadvantages of using work-horses for the management of wetland areas.

Strengths	**Weaknesses**
Horse power has a multidisciplinary effect	Working horses need breaks
Working horses do not consume renewable energies	Time consuming management to keep landscape open
Under difficult conditions (wet meadows) they work more soil-conserving than agricultural machines	Working with horses needs special know-how which needs to be acquired
The slower working-process enables other animals to escape in time	In areas with more than 30% inclination the use of this mowing technic is not possible
Protection of small structured cultural landscape, biodiversity and rare breeds (e.g. “Noriker”), multifunctional effort	More funds and ambitious persons are needed (different initiatives are rising in Europe/ e.g. FECTU)
Attractiveness for urban population looking for rest and relaxation in nature and open rural landscape	
In comparison to machines, the noise emissions of working horses are very low	A distance of more than four kilometres to the next working-location is not economically
The mowing equipment for working horses is cheaper than a high tech tractor or machine	Working with horses needs a lot of personal initiative and a direct contact to the animals, nature and environment
Horses are no direct fodder-competitors against cattle	

Table 2: The following aspects compare sustainable management of wetland areas with work horses and other mowing methods from the economic side.

Machine/Method	Ground coverage per hectare [h]	Diesel consumption [L]
Tractor	1	5-8,3
Motor mower	6	4,1-8
Scythe	12	to get to the site by car
Horse-drawn mower	5	0
Horse-drawn tarpaulin	4	0
Removing by hand	12	0

h-hours; L-liter

Figure 2: The removal of hay with a horse-drawn tarpaulin that is more efficient than manual work. (Fuchs, E. 2014).

Another important reason for the management of protected wetland areas (Mayer, et al. 2013) with horse power is the protection of soil and the reduction of soil compaction compared with machines which can cause huge damages. Studies showed that horses can use the same spot at least eight times before an extensive hoof print produces a compression of soil (Fleischer, M.; Süß, D. 2002; figure 3). The compression from a tractor or other big machines spread on the whole surface. The wheel ruts affect the soil-water-balance in a negative way.

The benefits of the use of horse power comprise also the protection of species and the better chance for animals to escape in time during the mowing process.

Figure 3: The hooves of the work horses cause less damage to the soil than agricultural machines (Fuchs, E., 2014).

4. Conclusion and outlook

The local history of the Enns Valley and the Ausseerland is characterized by traditional grassland management to protect and preserve small-structured cultural landscape and wetland areas. The preservation of cultural landscape is essentially to stop forestation and the loss of biodiversity and at the same time to reinforce added value through sustainable tourism. The manifold landscape was and is an essential basis for the exchange between rural and urban areas and its population. The Enns Valley and the Ausseerland have a long tradition in recreation for people coming from cities like Vienna or Graz. Many secondary residences are still situated in these touristic areas. The sustainable tourism plays a very important role. The manifold landscape (mountains, pastures, river valleys with its high potentials of habitats and species) as well as typical plants as *Iris sibirica* and *Narcissus radiiflorus* are trademarks for the regions. The high air quality, culture and tradition and typical regional products are important for economic benefit. All age groups from urban areas benefit by using these services for recreation, sport and research of nature (nature science and sport camps). The Leader Region Enns Valley and Ausseerland represents an important impact for sustainable land management.

During the last decades a renaissance of the use of horse power is in progress. Horse management for agriculture, riding, coach tours and therapy on horse (social farms) played and still plays an important role in winter and summer time. Old sumpter paths crossed the Enns valley. The old knowledge is used for new innovative tourism attractions and new economic initiatives.

Today’s management is mainly based on agricultural subsidies. With the ongoing structural change in agriculture new strategies and innovative solutions are needed for future sustainable management. The realized project is a best-practice example for sustainable, ecological management of wetland areas to keep cultural landscape open and to develop a green puffer zone which can be used beside agricultural management also for recreation and awareness rising for protection of habitats and species. Local population and regional famers who still remember the use of work horses took great interest in the method which is at the same time an old and new approach. They were surprised about the modern machinery partly imported from Germany and USA. Also media and tourism showed great interest in the project.
The sustainable management of wet land areas e.g. Natura 2000 sites and other relevant protected areas with workhorses was implemented in the new Austrian ÖPUL Programme (Agri-Environmental Program 2015) to enhance environmentally-friendly management of agricultural areas.

References

[1] Hochegger, K., Mayer, R. (2014): Mowing wet meadows with horses, conference proceedings of the international expert conference Biodiversity and Leader organized by the Umweltdachverband in Vienna, 27.

[2] Hochegger, K., Mayer, R., Plank, C., Bohner, A., Schaumberger, J. (2013): Utilization History of Alkaline Fens in the Natura 2000 Area Ödensee Salzkammergut – New Strategies for Future Management. 5th Symposium for Research in Protected Areas, 10-12 June 2013, 299-306.

[3] Mayer, R., Plank, C., Plank, B., Bohner, A., et al (2013): BE-NATUR: Transnational Management of Natura 2000 sites, Protected area management, Intech, Rijeka, ISBN 978-953-51-0697-5, 149-182.

[4] Fleischer, M., Süß, D. (2002): Die Beanspruchung des Bodens beim Pferderücken. Starke Pferde, Ausgabe Nr. 24, 11-13.

TE-O_04 Practical experiences in Flanders towards a rural-urban sustainable biomass policy

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1. Background

Flanders has a long tradition of using biomass for food, feed and wood industry, as well as 25 years of separate collection and recycling of bio-waste in rural and urban areas. Stakeholders of the different governments and industries asked for a clear sustainable policy for the destination of biomass (residues) in Flanders because Flanders is depending on the import of biomass for material and energy production. The Flemish government ratified on the 10th of July 2015 the first action plan for the sustainable management of the most biomass and biomass residues for the next five years. The challenge is to find a balance between the more and more important place of biomass in the waste to material transition and the transition from a fossil to a non-fossil society.

2. Legislative demand

In order to prepare the action plan we made an analysis of the supply and destination of biomass in Flanders in 2013 and a forecast for 2020. We evaluated the local and some surrounding and EU-strategies and political regulatory measures which encourage the bio-economy by linking rural and urban biomass (residues) and which enable compromise solutions between stakeholders with conflicting interests of the food, wood and energy industry. We give an overview of the most essential shortcomings which ask legislative actions.

2.1 Necessity of priorities and adapted policy instruments

New ministers in 2014, new priorities. Different ministers for environment and energy do not make policy choices easier in Flanders. At the one hand there are the steps to be taken in the development towards a bio-based economy. At the other hand not all the targets for renewable energy can be realized at the same time and cannot be mainly based on biomass. Policy choices have to be made between creating/holding jobs, the position of competition of the industry, safe-guarden the (Flemish) soil quality etc. Even the choice between energy and materials is not an exclusive choice for one above the other. A dialogue with all Flemish stakeholders is necessary to come to a priority list for Flanders, to match the long term objectives and to develop the appropriate policy instruments to send in the desirable direction.

2.2 Need for more supply to respond to higher demand

Growth of the food industry means not only higher production per hectare of vegetables and feedstock, but also more import of wheat, sugar beets, soya, etc. 98% of the residues from the food industry find their way to food, feed, cosmetics, compost etc.

The demand for waste wood in the particleboard industry and for energetic valorisation will be 1.3 million tonnes higher in 2020 in comparison to 2014 (figure 2). The demand for the import of industrial wood pellets for energetic valorisation is about 2.5 million tonnes a year in 2020.

Because of the high demand, the focus lies on higher use of the biomass residues in own country. The total amount that is separately collected is about 6.95 million tonnes, more than 36% in the food industry,19% is woody fraction, as shown in figure 1. This figure doesn't include 23 million tonnes of manure.

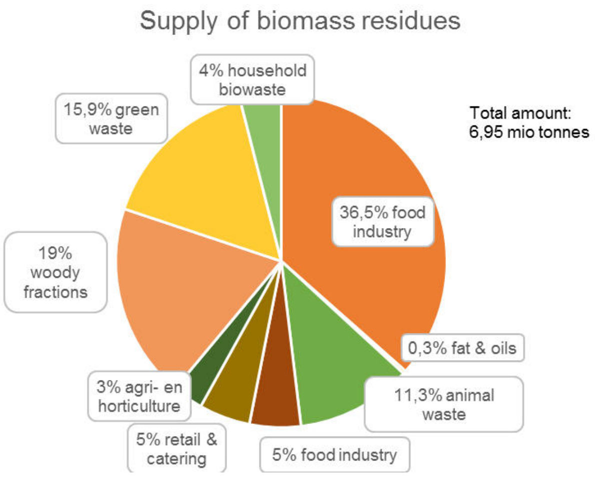

Figure 1: Locally collected biomass residues in Flanders in 2013 (excl. manure).

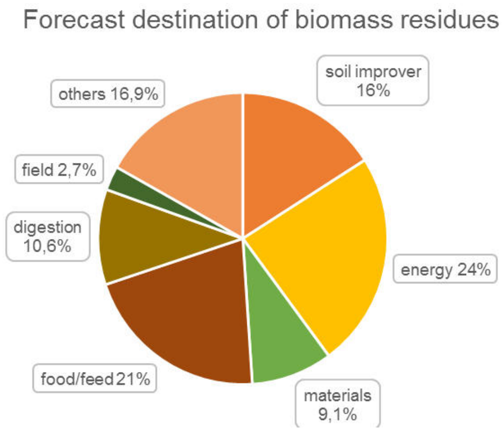

Figure 2: Local need for separate collected biomass residues in 2020 (excl. manure).

2.3 Financial barriers

A lot of challenges are waiting for the anaerobic digestion plants in Flanders because of the high standards for discharging liquid phases to surface water and the phosphate restrictions for using digestate on land. Different examples show the local potential of specific use of treated digestate. Depending on the current price of artificial fertilizer, new opportunities for struvite and other recycled nutrients are expected for the future. The composting plants for bio waste are looking for financing predigestion of the bio waste in order to combine the production of energy and good compost quality. The paper industry has converted to self-supply of energy by producing green power and heat by incinerating own biomass waste, together with wood waste from e.g. rural areas or RDF. Next to the need of investing in pilot and full scale plants, a certain budget for applied research in universities, research institutions and private companies is needed.

3. Future : New Biomass Management Plan 2015-2020

One year of intersectoral debate has led to an action plan towards a rural-urban sustainable biomass policy that is accepted by the Flemish government on the 10th of July 2015. It contains action programs with respect to 3 material chains:

- The chain of organic-biological biomass (residues) of the chain agriculture, food to consumer;
- The chain of biomass (residues) of parks, landscapes, forest etc.;
- The chain of woody biomass (residues) of industry and households.

The expected corrections and new headlines related to the rural-urban related streams (chain 1) are the following.

3.1 Policy made for the organic-biological biomass of the chain agriculture, food to consumer

In Flanders we already had a legislative framework for bio- and green waste from households. We will continue and intensify this policy of bio-cycling at home and composting of green waste in so called green regions, and bio-waste digestion/composting in the bio-waste regions in the next five years.

Figure 3 gives an overview of the future options for treating biomass and bio-waste in the whole chain. It's clear that quality control, green subsidies for treatment and research etc. will encourage in all phases of production, consumption and treatment of biomass and biomass residues. Some actions are highlighted in the next points.

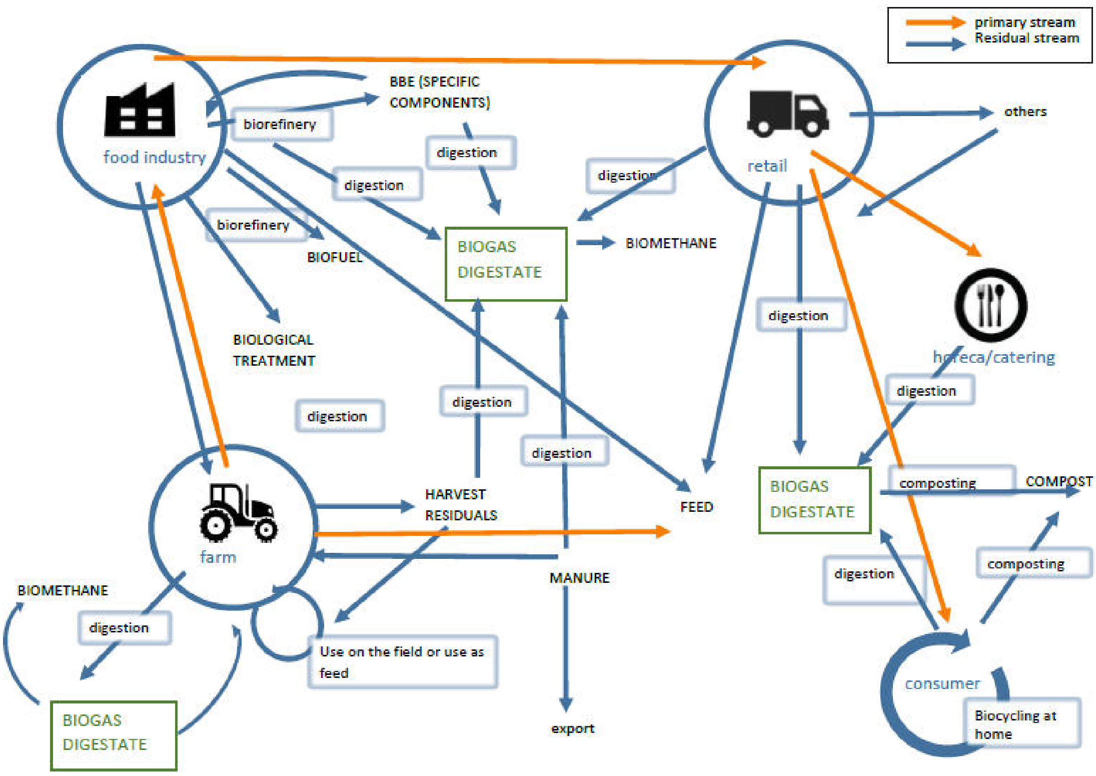

Figure 3: Policy management of biomass and residues in the chain of agriculture, food to consumer.

3.2. Prevention of food losses saves money

Process residues in the food industry will mostly be reused in the own production because of the growing resource costs (e.g. vegetables of the retail becomes a retail made sauce) or by industrial symbiosis with the production chain of another company. This gives a financial and an environmental win-win. The industry is stimulated by the condition of best available technique in the license policy and by the introduction of energy and material efficient processes by the innovation subsidy. The purpose is to minimize the production of waste streams. On the side of the consumer reduction of the consumption, Vlaco and other organizations try to increase awareness of the production of waste. Saving waste is saving money. The European ambitions of the reduction of food losses by min. 30% between 2017 and 2025 are translated in a Flemish roadmap for prevention of food losses and reuse of food waste as food.

3.3. Separate collection is the key for optimizing valorization

We focus on valorizing in the own region by intensifying separate collection for households (> 70%) and industry (>90%). Good quality of the bio-waste (less than 2% impurities) is essential.

Farmers are advised to leave less waste on the field where feasible in order to reduce N-losses and to create a higher value. Those catering and retail companies that are not collecting separately should look for a win-win. The further introduction of the PAYT-principle (pay-as-you-throw) will reduce the quantities in the collection. This is leading to a paradox. The availability of a sufficiently large volume of bio-waste is a requirement for the economic profitability of new sustainable treatment techniques.

3.4. Optimization of the integrated chain approach of the treatment of biological side-streams

In the future the integral use of biomass will be more and more important to close the chain on a sustainable way. Therefore the Flemish ambition is a recycling rate of 95% of biowaste collected from households and 90% of the industry. This ambition is possible by one or more of the following treatments:

- Use of the waste streams "as is", based on a good conservation technique, e.g. residues from the potato industry for feed, B-pears for juice production etc.
- Winning specific components by extraction, e.g. pectine out of fruit that is no longer convenient for food, the residues can be used for feed or go to digestion;
- Conversion of parts to biochemical based components e.g. aromates from lignocellulose streams, fatty acids from animal waste;
- The production of soil conditioners due to the Vlaco-quality control on the whole chain (Vlaco is the Flemish member organization for home and professional composting and digestion) is ECN-QAS-certified for compost and digestate. Vlaco coordinates the research project for using green compost in potting soil for hobby use and professional use. Compost from bio-waste from households is more used in agriculture because of the positive effect on the organic matter and universal soil use. The DIMA-project of Vlaco will focus on the local potential of specific use of treated digestate for organic fertilizers e.g. in public/hobby gardening etc. 2016 is the year of the Floraliën, an international well-known flower and garden event in Ghent. Vlaco is cooperating for showing e.g. the positive effects of home gardening and compost use in a demonstrating sustainable garden. In order to reduce the income dependancy of the seasonal variety in quantity of waste streams, some bio-waste composting plants plan to build a digestor (around 1 MW each) before the composting process in order to combine the production of energy (green power/gas) and good compost quality. To do so, subsidizing the investigation and the green power production is a conditio sine qua non.

4. Conclusion and outlook

The agreement of all stakeholders and the Flemish government on the biomass management plan and its priorities for 2015-2020 shows that compromises are possible between stakeholders with conflicting interests. It will be a big challenge for the rural and urban biomass and bio-waste sector to contribute to a more sustainable bio-economy on a cost effective way. This means that maximally creating economic added value and foreseen in the demand of bio-based energy will probably don't match together. The success is depending on the implementation of the new biomass action plan supported by a mix of different instruments e.g. realistic goals for consumers and industry, support for a quality based chain management, bio-refinery facilities, green support for producing green power of biogas and using biogas for transport from digestion.

This action plan provides a stable legal framework for investors in new and existing treatment plants and investments in the infrastructure for separate collection etc. Private and public stakeholders see it as an opportunity because of the increasing costs of waste treatment, and probably the potential gains. We are aware of enough flexibility of the policy instruments to respond to changes in priorities.

References

[1] Braekevelt, A., Vanaken, N., Bouteligier, S. (2014): Background document for the sustainable management of biomass (residues) in 2015-2020, 168 p., Public Waste Agency of Flanders, www.ovam.be

[2] Braekevelt, A., Vanaken, N., Bouteligier, S. (2014): Action Plan Sustainable management of biomass (residues) in 2015-2020, 97 p.,Public Waste Agency of Flanders, www.ovam.be

[3] Interdepartmental working group bio-economy (2013): Bio-economy in Flanders, vision, strategy and actions, 22 p., Flemish government, www.lne.be.

TE-O_05 Integration of sanitation infrastructure with agriculture in peri-urban communities of South Africa

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1. Objectives

To meet the sanitation demands for the residents staying in informal settlement far away from centralised wastewater systems, where Ventilated Improved Latrines (VIP) cannot be erected, Community Ablution Blocks are considered as a potential solution (SERI, 2011). The Decentralized Wastewater Treatment Systems (DEWATS) combined with the CABs can be used for effluent (grey and blackwater) treatment. Community agriculture areas are planned to make use of the high plant nutrient content in the treated wastewater from DEWATS plant. This would allow the communities to have a higher quality of sanitation as provided by the CABs, reduced municipal maintenance costs and the potential to contribute to improved livelihoods, employment creation and food security for the community through urban agriculture. This concept is being investigated at the Newlands Mashu experimental site in Durban where a DEWATS plant has been installed. The studies are focusing on processes that would ensure the safe use of effluent as well as understanding effluent movement through different soil types, root uptake mechanisms and the effect on crop growth and yield, while meeting environmental limits for N and P disposal. The specific objectives were to assess the DEWATS effluent chemical properties. Swiss chard (*Beta vulgaris* L.) yield, nutrient uptake and land area required for irrigation with a specific volume of DEWATS effluent in different soils was determined.

2. Methodology

Method 1: Characterization of the DEWATS effluent

The DEWATS effluent was analysed for Chemical Oxygen Demand (COD), mineral N, orthophosphate, K, Ca and Na. The COD test was done according to the standard methods for wastewater (Foxon et al. 2004). The mineral N and orthophosphate were determined using the Merck spectroquant ® according to standard methods for wastewater test. Inorganic elements (P, K, Ca and Na) were analysed using the Inductively Coupled Plasma Atomic Emission Spectroscopy (AES-ICP) according to the standard methods for wastewater (APHA, 1999). Data analysis were done using the statistical software GenStat® 14th Edition (VSN International, Hemel Hempstead, UK). Means were compared using Standard Error of Deviation (SED`s) at the 5% level significance.

Method 2: Tunnel crop growth experiments

The experiment was designed as 3 x 2 x 3 factorial treatment structure using a complete randomised design (CRD) with the following factors: Soil types (Sandy loam, Clayey loam and Acidic soil- 3 levels); Irrigation water sources (DEWATS effluent and tap water, (TW) – 2 levels); Fertiliser rate (no fertiliser, half optimum and optimum rate – 3 levels) replicated four times. The three soils; sandy loam (Cartref), clayey loam (Sepane) and acidic (Inanda) soils were collected from KwaDinabakubo (Hillcrest, Durban; 29°44.046'S; 30° 51.488'E), Ukulinga agricultural research farm (Pietermaritzburg; 30°24'S; 29°24'E) and World's View (Pietermaritzburg; 29°35.003'S; 30°19.7404'E) respectively. The soils were collected within the 30 cm depth and a composite sample for each soil type sent to CEDARA, Soil Fertility and Plant Analytical Laboratory, Pietermaritzburg for the analysis of chemical properties. Fertilizer treatments were based on the optimum recommended rates: Cartref (N 100 kg ha^{-1}, P 270 kg ha^{-1} and K 435 kg ha^{-1}), Sepane (N 100 kg ha^{-1}, P 195 kg ha^{-1} and K 10 kg ha^{-1}) and Inanda (N 100 kg ha^{-1}, P 195 kg ha^{-1} and K 445 kg ha^{-1}). Swiss chard (*Beta vulgaris* L.) was planted in pots filled with 10 kg of soil. Three seeds were planted per pot and thinned to one plant. The pots were irrigated to maintain 70 % field capacity on a five day irrigation cycle. Effluent and water quantities applied were recorded. Irrigation was applied using two litre polypropylene soft drink bottles (modified drip) to prevent crop contamination of leaves according to World Health Organisation (WHO), guidelines. After 11 weeks plant growth parameters (number of leaves, leaf area, fresh and dry yield) were measured. Above ground fresh mass was determined immediately after harvesting while dry mass was measured after drying at 70°C for 72 hours. The dried above ground samples were ground and sieved past a 1 mm sieve and, together taken to CEDARA for plant tissue analysis. Soil samples were also taken after harvesting, mixed and a sample was sent to CEDARA for chemical properties as previously described.

Method 3: Field experiments on nutrient leaching

The experiment was designed as single factor analysis comparing DEWATS effluent (DEWATS), tap water with fertiliser (TW) and rain fed conditions with fertiliser in randomised block design (RCBD) replicated three times. Wetting Front Detectors (WFDs) were installed 10 cm from the Swiss chard (*Beta vulgaris* L.) plant at 30 and 50 cm depth to collect leachates. Irrigation was scheduled at five day intervals using the Soil Water Balance (SWB) model (Water Group, University of Pretoria). The SWB was used to estimate the water requirements during irrigation. These values were used to estimate the land required. Soil moisture was monitored using a neutron probe (CPN 503DR Hydroprobe, Campbell Pacific Nuclear, CA, USA). Leachates were collected after each heavy rainfall event and analysed for nitrate-N and phosphate-P using the Merck® Reflectoquant according to standard methods. Changes in the concentrations of nitrates and phosphates at different soil depths; 30 cm (rooting zone) and 50 cm (below rooting zone) were determined.

3. Results and discussion

3.1 Characterisation of the DEWATS effluent

Table 1 shows the chemical properties of the DEWATS effluent used for irrigating Swiss chard during the experiment. Large amounts of ammonium – N was compared to total – N was observed, the same with respect to orthophosphate – P which was higher than total – P. This observation was attributed to the operational problems on the DEWATS since these chemical analyses were done on different samples collected at different times. The DEWATS effluent contains significant amount of nitrogen, predominantly in the form of ammonium. The analyses showed comparably higher quantities of ammonium compared to nitrates. Crops mainly take up nitrogen in the form of nitrates, hence nitrification is essential. However, the process is dependent on soil properties such as pH, texture as well as irrigation management practices. The effluent contains considerable concentrations of Na which can have an effect on soil aggregate properties and salinity.

Table 1: Chemical properties of the DEWATS effluent used for irrigating Swiss chard during the experiments.

Characteristic	Quantity (mg/L) (n=4)
Ammonium-N	94 ± 9.62
Nitrate-N	0.625 ± 0.625
Phosphate-P	19 ± 2.38
Total nitrogen	61 ± 6.33
COD	192 ± 21.1
K	14.2 ± 2.3
Total P	5.1 ± 1.45
Ca	28.9 ± 7.3
Mg	14.7 ± 4.73
Na	35.5 ± 0.76

3.2 Tunnel crop growth experiments

Figure 2 shows a significant decrease in exchangeable acidity in plants grown on acidic soils, being high in tap water irrigation (1.53 $Cmol_c\ kg^{-1}$) compared to ABR effluent irrigation (1.35 $Cmol_c\ kg^{-1}$) as expected due to the liming effect of DEWATS effluent (Bame et al. 2013).

Figure 3 shows the interaction between soil type and irrigation water on Swiss chard (*Beta vulgaris* L.) growth. A significant response to irrigation with DEWATS effluent was observed in the acidic soil ($P<0.05$), whereby all the growth parameters were higher compared to municipal water irrigation due to the liming effect of DEWATS effluent (Figure 3).

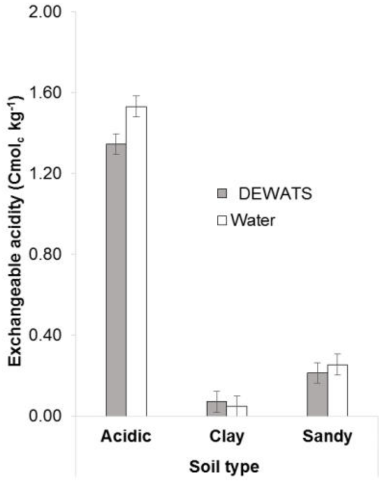

Figure 2: The effects of irrigation water sources on soil exchangeable acidity (Cmol$_c$ kg^{-1}) in different soil types .

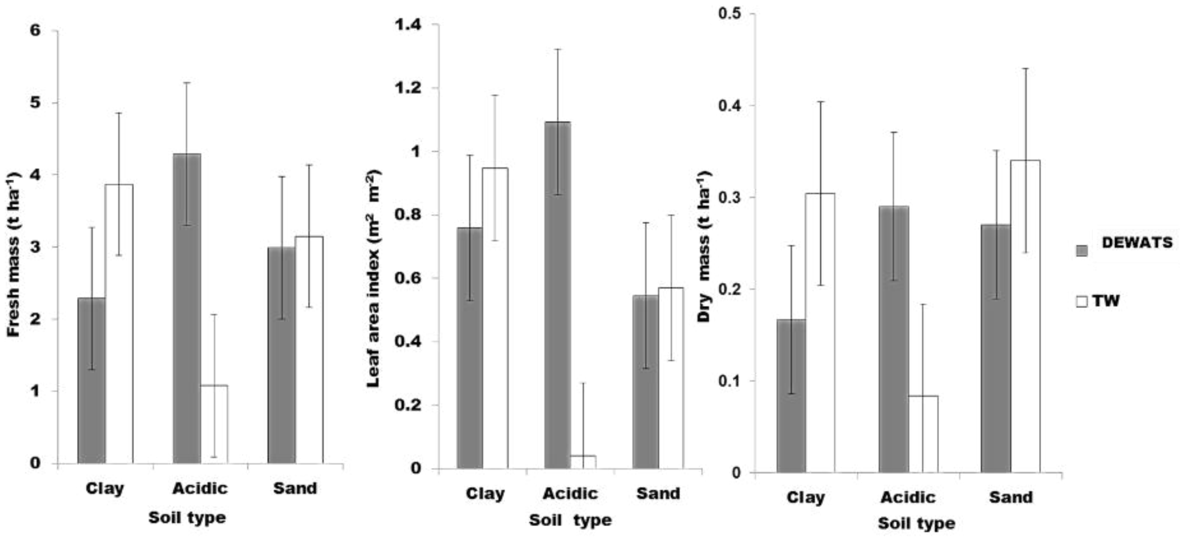

Figure 3: The interaction between soil type and irrigation water source on Swiss chard growth (Leaf area index, fresh and dry mass).

3.3 Nutrient leaching

Figure 4 shows the concentrations of nitrates and phosphates at two different depths (30 cm and 50 cm) between the two treatments (DEWATS effluent and TW). Nitrates and phosphate concentration differed significantly between the two depths (P<0.05) with more concentrated within the 30 cm than 50 cm for all treatments. This suggests that N and P leaching was not significantly different between the DEWATS effluent and tap water irrigation treatments and nutrient leaching could not be attributed to irrigation with DEWATS effluent.

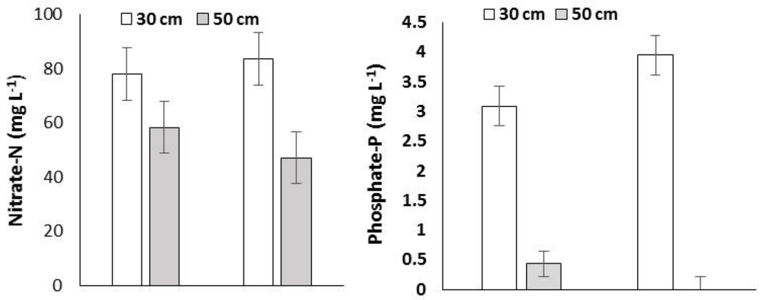

Figure 4: The concentrations of nitrates and phosphates in leachates from the field trials at two different depths (30 cm and 50 cm) after irrigating with DEWATS effluent.

3.4 Estimation of Land area

From table 2 more land area is required when irrigating in sandy loam soil especially during the summer season (1 120 m^2 m^{-3} of DEWATS effluent). Smaller amount of land is required on clay loam soil irrigated in winter season, (960 m^2 m^{-3} of DEWATS effluent) than in sandy loam soil. With this irrigation scheduling sufficient amounts of N can be applied, which are within the range for optimum Swiss chard production. However, there would be need for supplementing the P since it is far below the optimum required for Swiss chard production (56.7 kg ha^{-1}).

Table 2: Determination of the amount Chemical properties of the DEWATS effluent used for irrigating Swiss chard during the experiments.

	Sandy loam (Cartref soil)		**Clay loam soil (Sepane)**	
	Winter season (2012)	**Summer season (2012)**	**Winter season (2012)**	**Summer season (2012)**
Root zone deficit (m)	0.107	0.062	0.119	0.071
Root zone irrigation simulated requirements (m)	0.256	0.256	0.256	0.256
Total irrigation required (l ha^{-1})	3 630 000	3 180 000	3 750 000	3 270 000
DEWATS effluent-N (mg l^{-1})	61.1	61.1	61.1	61.1
DEWATS effluent-P (mg l^{-1})	5.1	5.1	5.1	5.1
Simulated total-N (kg ha^{-1})	221.8	194.3	229.1	199.8
Simulated total P (kg ha^{-1})	18.5	16.2	19.1	16.7
DEWATS capacity m^3 (90 days)	900	900	900	900
Land served by each m^{-3} of DEWATS effluent yr^{-1}	0.1	0.112	0.096	0.112

4. Conclusion and outlook

The DEWATS effluent contains mineral nutrients important for crop growth, such as nitrogen, phosphorus and potassium. Nitrogen levels in the effluent are relatively high but there will be need to supplement phosphorus and potassium from inorganic fertiliser. The ability of DEWATS effluent to increase nutrient uptake is influenced by soil physical and chemical properties. The liming effect of DEWATS has also been reported on acidic soils and was evident from the reduction in exchangeable acidity and increased crop growth in the acidic soil. This suggests that the effluent can be used as a liming agent in acidic soils. Irrigation with DEWATS effluent did not have a significant effect on the leaching of N and P because N and P movement in the soil could not be attributed to the effluent. The results suggest that a DEWATS plant can be able to irrigate about 2 400 m^2 to 2 800 m^2 of agricultural land under a Swiss chard crop, depending on the soil type.

There is potential for DEWATS effluent use in agriculture for crop production as an irrigation water and nutrient source. The study shows that soils and plants could act as a final wastewater treatment phase, in which soils retain mineral nutrients thus making them available for uptake by crops for biomass production. Integration of urban agriculture with urban sanitation though community ablution blocks, DEWATS and agricultural land has a potential for sustainable agriculture, while ensuring the energy-food-sanitation nexus which will benefit poor urban farmers.

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References

[1] American Public Health Association (APHA) (2012): Standard Methods For the examination of Water and Wastewater. 22nd Edition. United States of America.
[2] Bame, I.; Hughes, J.; Titshall, L.; Buckley, C. (2013): Leachate characteristics as influenced by application of Anaerobic Baffled Reactor effluent to three soils: a soil column study, Chemosphere 93, 2171–2179.
[3] SERI (Socio Economic Rights Institute of South Africa) (2011): Basic Sanitation in South Africa: A Guide to Legislation, Policy and Practice. URL: http://www.nwu.ac.za/sites/www.nwu.ac.za/files/files/p-fl/documents/SERI_Guide_to_Basic_Sanitation_Jul11.pdf
[4] www.borda-net.org

TE-O_06 Planning the sustainable use of agricultural bioresources as fuel for city busses – Example case Turku, Finland

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1. Objectives

Transport produces about 20% of Finland's greenhouse gas emissions. Transport is the second largest CO_2 emitter following the electricity and heat production emissions. Although Finland has invested heavily in biofuels in heat and power production and the share of renewable in energy production is 36.8% (in 2013), the share of petroleum products in transport is still about 90%. Many studies have concluded that biomethane is one of the most sustainable biofuels available today [1]. Biomethane is a gaseous fuel, which is produced from biogas and as well as natural gas, it can be used as vehicle fuel in different types of vehicles. The sustainability of biomethane is partly because biogas can employ various types of waste materials as feedstock and when biomethane is produced from energy crops, the residual from the process can be used as fertilizers, thus minimizing the use of inorganic fertilizers. The air emissions from the use of gaseous fuels in vehicles are also lower than from liquid fuels, because of the higher ignition temperature and a higher lower flammability limit than liquid fuels [2].Feedstock availability, the most critical requirement for biomethane promotion, is linked to systematic analysis and planning the sustainable use of regional biomass resources. In many cases the theoretical biomasses potential for bioenergy production may be large, nevertheless a detailed analysis of land availability and the infrastructure may limit the actual amount of potential biomass to half of the theoretical amount [3].

The role of financial supports, both national and EU funded, that control the use of fertilizers and agricultural land is essential when planning the biogas production from field biomasses. Well managed biogas production from field biomasses can sustain fertility of the fields by closing the nutrient cycles. However, the harvested phosphorus yield may exceed the allowed phosphorus fertilization when field phosphorus level is good or high under the terms of Environmental support of Agriculture in Finland [4]. Under previous years (2007-2013) 95% of the field area [5] used in the Finnish Agricultural production was run according those terms, implying importance of those restrictions. Thus the area needed for spreading digestate may exceed the area needed for biomass production additionally the digestate may be needed to be processed to some extend to separate the main nutrients (P and N) from each other. Further some additional nitrogen fertilization may be needed for good biomass yield [6].

Logistics and storage play important role in economics of biogas production and their role is pronounced in Finland where spreading of digestate is not allowed in wet soil or during winter (from November to March) [7] and wet soil is restricting the spreading typically some periods each year. Transportation costs of digestate may easily exceed the respective value of digestate nutrient content [6]. Further, the effects of logistics on the local traffic and dwellings should be taken into account.

The objective of the research was to plan sustainable biomethane production from energy crops for city busses in Turku. The aim was to assess how much fields are needed and how digestate can be used as fertilizer in sustainable way in surrounding areas. The objective was that the biogas plant and fields in surrounding area form a closed cycle of nutrients and that this example case brings tools for future biogas plant planning.

2. Case study description

In Turku, about 3 Mm^3 of biomethane is produced per year from waste materials. Needed amount of methane for city busses in the area is about 5.6 Mm^3. Basis for the study was that extra 2.6 Mm^3 of methane could be produced from energy crops. The potential location of new biogas plant (Topinoja area) was selected as a request of Turku city, as there is an existing biogas plant processing biowaste and sludge in the same area and the area is logistically potential area for biomethane distribution. The area (Fig 1) is between rural and urban area. Grass was selected as raw material as methane potential of ensiled grass is good, years of crop failure are rare and production costs compared to energy content of product are reasonable. Also, cultivation, harvest and preservation of grass are known and all technical solutions are available (compared to e.g. maize production in Nordic countries) [8]. The needed amount of field for grass cultivation and nutrient recycling was based on two alternative scenarios. In base case only grass species were cultivated requiring strong nitrogen fertilization. In alternative case, cultivation was partly based on a mixture of grass and clover that is able to fix nitrogen from atmosphere. Mixture was cultivated on the fields where spreading digestate is not allowed due to the high phosphorus status of the field, where cultivation of clover would enable harvesting relatively good yields without fertilization. It was assumed that cultivation will follow EU nitrate regulation [9] and terms of Environmental Support of Agriculture 2014 [4] (e.g. taking phosphorus status of the field in Turku area into account). It was also assumed that nutrients from biogas digestate will be recycled back to fields and the inevitable nitrogen loss will be replaced with chemical fertilizers used on grass fields. Soluble nitrogen and water soluble phosphorus of digestate were taken into account when calculating the amount of nutrients that should be returned on to the fields. Solubility of nitrogen and phosphorus in digestate were assumed to be 65% and 85%, respectively. The field phosphorus level was calculated from statistics of soil samples [10].

For field location information and distance calculations, geographic information system (GIS) was used. Fields over 1.5 hectares were taken into account. Transportation distances were derived using the OD Cost Matrix application in ArcGIS Network Analyst which uses the Dijkstra's algorithm in order to determine the shortest path from a starting point to a destination location. The criteria for storage locations were good roads between Topinoja and storages, large field density and low population density between storage areas.

3. Observations

In was assumed that the harvested 7500 t dry matter (DM) grass/ ha would remove 22 kg/ha phosphorus and the yield (5625 t DM /ha) of clover –grass mixture would remove 13 kg/ha phosphorus. The phosphorus amounts are above the maximum allowed yearly phosphorus fertilization of the grass fields according to the terms of Environmental support of Agriculture in Finland 2012, assumed that the field has phosphorus status at least satisfactory/tolerable. Phosphorus status of the fields of the study area was satisfactory/good, meaning that part of the digestate should be used for fertilizing something else than grass for biogas production. Further, it was taken into account that the pure grass species require nitrogen fertilization despite the phosphorus status of the fields. When calculating the field area needed for sustainable raw material production for biogas plant, the factors taken into account included the phosphorus status of the fields, crop yield, amount of harvested nutrients and allowed fertilization of grass and nurse crop of establishment year.

The needed grass area for producing 2.6 Mm^3 methane is from 1100 to 1300 ha depending on the proportion of clover in the grass mixture (Table 1). High grass yield is achieved by the pure grass species if additional source of nitrogen fertilization is used (in addition to the digestate 87 kg nitrogen/ha grass). Area needed for grass re-establishment is from 280 to 350 ha yearly, barley would be cultivated as nurse crop. Additionally, some phosphorus surplus (2 kg/ha) will be establish in the case of pure grass species, meaning that additionally over 200 ha grain area will be fertilized with the digestate. Further approximately 10 % safety margin was suggested to cover the yearly variation in the yields. Thus regardless of the composition of grass sward 1800 hectares would be needed to operate nutrient cycle and biomass production of the biogas plant.

The needed 1800 ha area can be obtained within 9 km radius from Topinoja biogas plants if presumed that all field plots above 1.5 ha in size are available for biomass production (Fig 1). With increasing distance, the field area increases rapidly enabling energy crop cultivation along with food and feed cultivation. If 20 % of the farmers of surrounding area are producing grass for biomethane production and/or using digestate to fertilize their fields, the needed 1800 hectare field would be located less than 15.4 km radius from biogas plant.

Quite often silage and sludge transportation is done by tractors when the distance is less than 10 km. However, Topinoja area is located near big roads so tractor transportation would have a negative effect on local traffic. Using temporary storages would enable truck transportation and possibility to choose the time for transportation, e.g. avoiding rush hours.

Table 1: Calculated field area needed for running sustainable biogas production to produce 2.6. Mm^3 biomethane. Two alternative scenarios compared.

	Red clover included*	**Without clover****
Methane production m^3/kg DM	0.30	0.31
Harvest kg DM/ha	6 769	7 500
Nitrogen yield kg/ha	184	180
Phosphorus yield kg/ha	18	22
Methane production m^3/kg ha	2 068	2 355
Needed grass field area, ha	1 257	1 104
Area for grass sward establishment, ha	348	276
Additional area needed for digestate nutrients recycling	0	213
Total amount of field, ha	1 605	1 593

* Mixture of red clover and grass species (typically timothy and meadow fescue) cultivated on the fields having good or high phosphorus level (39 % of the field area, phosphorus class above satisfactory).

** Only grass species (timothy, meadow fescue etc.) cultivated and additional nitrogen fertilization included

DM = dry matter

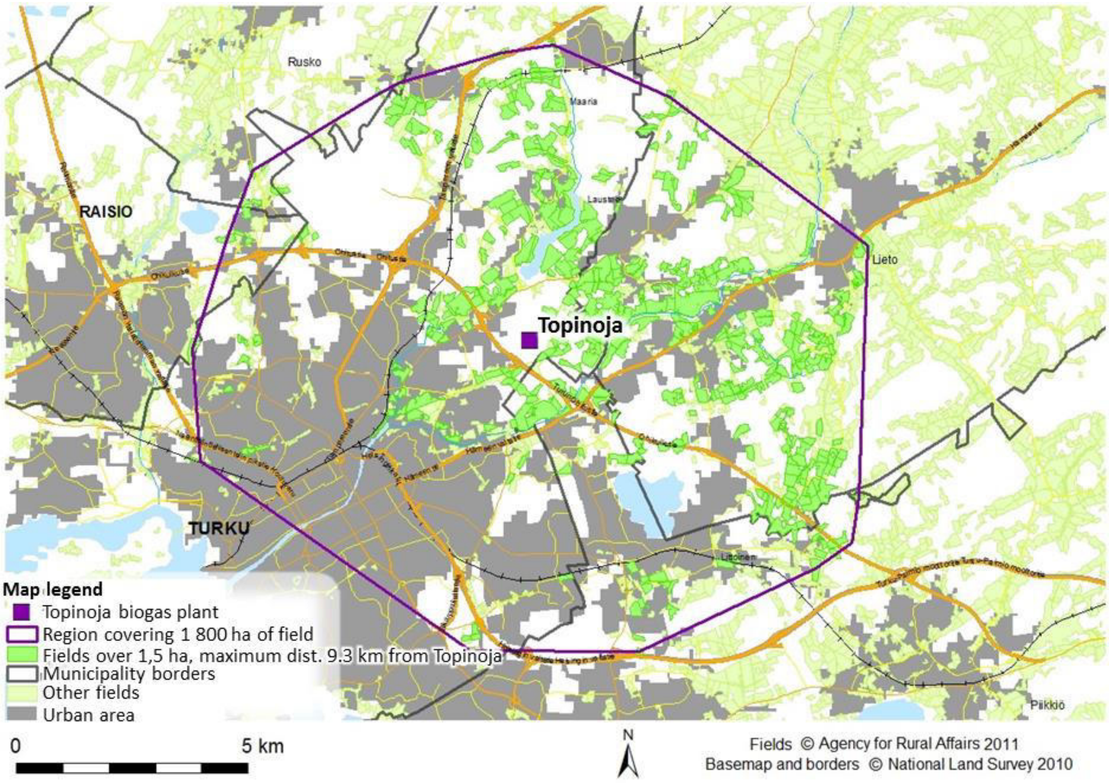
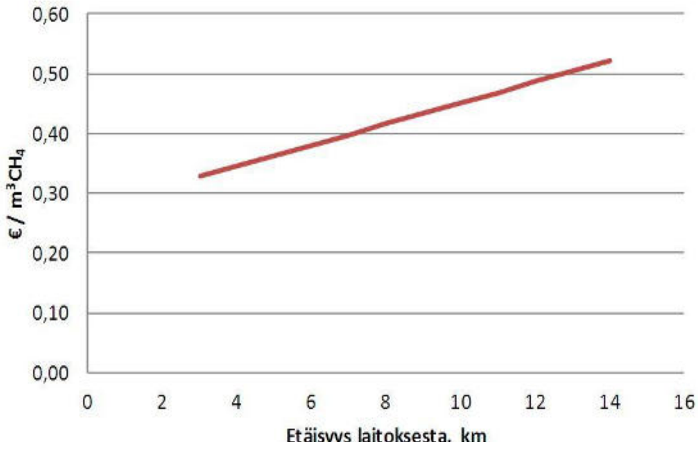


Figure 1: The needed 1800 ha fields close to Topinoja area (distance max 9.3 km along roads).

Due to traffic reasons, the temporary storages were suggested during this project, both to grass silage and to digestate. Amount of fields near selected storage areas is about 9 000 ha. This area is big enough to produce the needed amount of energy crops if at least 20% of fields in the area are producing energy crops. If all the field close to storage area are used for energy crop cultivation, the needed transportation distance from field to storage is on average 2.5 km. If about 33% of fields in the area are used for energy crop cultivation, the transportation distance will increase to about 4.6 km. With 20% filed, the transportation distance is in average 6 km.

The economics of the biomethane production is depended on the distance from fields to the biogas plant but also the price of oil and the alternative crop, e.g. barley. The price of biomass will increase if the distance to biogas plant is increasing (Fig 2). The price includes compensation to farmers of the field use, the price of harvest and transportation and the price of nutrients (including spreading of fertilizers and digestate, digestate transportation and profits of selling nutrients). The high cereal price increases also the prize of energy crops as alternative use of fields is more tempting for the farmers.



The distance from biogas plant, km

Figure 2: The price of biomass is depended on the distance from the biogas plant.

In previous studies, it was calculated that the biomethane production is more profitable when all the produced gas can be sold for transportation use as gaining profit from electricity or heat is more difficult in Finland [11]. In this case, the amount of produced biomethane is based on the need of local bus transportation if they were using biomethane as a fuel. However, at the moment, city of Turku has not made the decision to change all local bus transportation running on biomethane.

4. Conclusion

When planning a biogas plant which utilizes field biomasses, the amount of nutrients in digestate and the allowed/needed fertilization of the crops should be taken into account in order to obtain closed nutrient cycle. In Turku case, the needed field area for digestate distribution is larger than the area needed for biomass production. In conclusion, 2.6 Mm^3 of biomethane can be produced on the target area based on closed nutrient cycle and grass silage as raw material and the bioenergy production can be done along with food and feed production. In generally, crop and fertilization planning is in crucial role when planning a biogas production from field biomasses and should be done prior to the investment decision. The total economics of the biogas plant could also increase, if the digestate was processed to more valuable fertilizer product, reminding more chemical fertiliser and thus being more familiar for farmers. Although, for that, more advanced digestate treatments technologies are needed.

References

[1] Adelt M, Wolf D, Vogel A. LCA of biomethane. J Nat. Gas Sci Eng 2011;3:646-50

[2] Pierce JL. Landfill gas for vehicle fuel: Assessment of its technical and economic feasibility. SWANA's annual Landfill Gas Symposium (2007). Accessed 10/2008. Available: http://www.scsengineers.com/Papers/Pierce_LFG_to_Vehicle_Fuel_SWANA2007.pdf/

[3] Zubaryeva A, Zaccarelli N, Del Giudice C, Zurlini G. Spatially explicit assessment of local biomass availability for distributed biogas production via anaerobic co-digestion – Mediterranean case study. Renewable Energy 2012;39:261-270.

[4] MAVI 2012. The terms of Environmental support of Agriculture in Finland (in Finnish). http://www.mavi.fi/attachments/mavi/ymparistotuki/66eIWeNri/Ymparistotuen_sitoumusehdot_2012.pdf

[5] The Ministry of Agriculture and Forestry 2011. Environmental support of Agriculture (in Finnish) http://www.mmm.fi/fi/index/etusivu/maatalous/tuet/merkitys/ymparistotuki_luonnonhaittakorvaus.html.

[6] Seppälä A, Kässi P, Lehtonen H, Aro-Heinilä E, Niemeläinen O, Lehtonen E, Höhn J, Salo T, Keskitalo M, Nysand M, Winquist E, Luostarinen S, Paavola T. 2014. From grass to vehicle fuel (in Finnish). MTT Report 151.

[7] Finnish Government degree 2014. Degree of agricultural emissions restrictions 1250/2014 (in Finnish) https://www.finlex.fi/fi/laki/alkup/2014/20141250

[8] Seppälä, M. (2013): Biogas production from high-yielding energy crops in boreal conditions. Jyväskylä Studies in Biological and Environmental Science 266.Seppälä, M. 2013. Biogas production from high-yielding energy crops in boreal conditions. Jyväskylä Studies in Bio-logical and Environmental Science 266.

[9] Finnish Government degree 2000. Government Decree on the restriction of discharge of nitrates from agriculture into waters. 931/2000 http://www.finlex.fi/en/laki/kaannokset/2000/en20000931.pdf

[10] Hortilab Ltd http://www.hortilab.fi/fi/about-us

[11] Rasi, S., Lehtonen, E., Aro-Heinilä, E., Höhn, J., Ojanen, H., Havukainen, J., Uusitalo, V., Manninen, K., Heino, E., Teerioja, N., Andersson, R., Pyykkönen, V., Marttinen, S., Pitkänen, S., Hellstedt, M. & Rintala, J. 2012. From Waste to Traffic Fuel Final Report – Finland case areas. MTT Raportti 50.

TE-O_07 SMART-3S and CST system for zero waste low-rise and high-rise urban settlement – A concept

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1. Objectives

More than 50% of the world's population lives in urban area [1]. Since then, one of the world's main problems is the increasing amount of urban solid waste along with rural to urban area conversion. The SMART-3S (Short Simple and Save) system is prepared to solve the urban solid waste management problems within densely populated urban area in Indonesia, mainly through waste reduction at source. Currently Indonesian municipal solid waste management is moving from "collect-transfer-dispose" to "collect-separate-treat-utilize and dispose" paradigm.

BEST-NGO experienced for more than 10 years on the community-based waste management through Material Recovery Facility (MRF). MRF is a facility for community-based waste reduction at source. By end of 2014 there are more than 500 MRF exists in Indonesia and this is one of the most successful indicator for the improvement of municipal solid waste management in the country. Unfortunately, each MRF needs a minimum of 200m^2 land. As the land availability is becoming an issue in the cities, there is a necessity for new concept of urban solid waste management that still leads to waste reduction at source but occurs at a smaller land. In Indonesia the SMART-3S is considered as a new technology that is introduced to the community level. The SMART-3S concept is addressed for the regions that are excluded from MRF's coverage.

This paper describes the pilot implementation of SMART-3R technology at the household level, taken from the example of the implementation in a mid-to-high settlement and in a low-to-mid income settlement, both located in Tangerang Selatan City, Indonesia. The paper also describes the suitable characteristic location and budget for the technology including the main actors who play role during the operations and maintenances. In addition the paper also describes the analysis of the liquid compost quality as an end product of SMART 3S technology and the comparison of wastewater nutrition and hazardous contents. Currently the pilot observations are still ongoing. Therefore the data and information described in this paper are not in the complete forms and the analysis is not thorough.

2. Methodology

This paper introduces The SMART-3S as an in-house technology. The SMART-3S, shown in Figure 1 is proposed to serve high-rise and low-rise settlement. As shown in Figure 1-ii, this technology uses a small shredder machine to blend the organic waste, after the waste is excluded from hard material such as bones, thick fruit skins and hard seed to cover the knife of the shredder. Thus the process creates slurry. From these sinks all slurry are collected in a private or communal-storage, shown in Figure 1-iii. The liquid compost, which is produced from the slurry, will be emptied regularly from time to time and can be used as a fertilizer [2].

- Method 1: learning-from-experiences
  The first method was learning-by-doing; by this method the liquid compost product from the SMART-3S was tested in the organic farmland in the front yard. The selected plants are tomatoes, chili and eggplants. A 3-months observation was conducted to get the result of direct implementation. This method also plays roles as a "showcase" or "to be seen" method that is proposed as live entertainment and educational campaign to grab people's interest in the surrounding area. It is expected many people will replicate the system whenever they see many types of the fruits are ready to be cultivated in front of the house.

- Method 2: quantitative analysis
  The second method is a laboratory test of the qualitative analysis method for the household waste. In order to convince people about the nutrition and hazardous ingredients of the prod-

ucts, preliminary test for BOD, TSS and fat was measured. Also the PH, C-Organic, N-Total, P_2O_5, K_2O and some metal contain like Fe, Pb, Cd, and Hg were measured. In addition the E-coli and Salmonella Sp were also measured. As a comparison, the liquid compost from the SMART-3S was compared to the wastewater analysis from the black water storage at the same source using decentralized wastewater treatment system at the same household. The analysis results will lead to further treatments and improvements of the SMART-3S. Also the observation for the plants using the liquid compost from the SMART-3S is still on progress.

Figure 1:. The SMART- Short Simple and Save (3S) System.

3. Results

Challenged by the lack of green/open space and also different human daily life behaviors, the idea of implementing this technology is adjusted from the favorable things in our life: low cost and low maintenance technology. The main proposed goal from this technology is that human should change their views to waste as resources and they should not threat their household waste far away from the waste source. Thus it will reduce the amount of fuel consumption from waste collection and transfer activities and promote zero waste urban settlement. In this regard, the necessity of environmental action based on waste-as-resources principle, such as urban farming, can be more recognized.

Urban area faces many environmental problems including un-covered waste generation at the landfill that is lack of green/open space. Urban people spent many working hours out side the house. In this way, as The SMART-3S waste in-place, they are proved to make life a way better and simpler, especially for the households with assistance which is common in the cities. This system emphasizes less contact with waste but bring direct benefit from the products. Since it can be applied for private and communal waste treatment that happens at home, it is assumed that the product has less contaminant compare to the products from the industries or other farming. The treatment happened before the organic waste is mixed with any other non-organic material, moreover hazardous waste/material. The investment cost for the SMART-3S is considered low for a long-term period. Also it is considered as easy maintenance as the organic waste still can be stored until the storage is full. It triggers people to do urban farming that address water scarcity and food sovereignty.

Table 1 shows the result of analysis of the liquid compost from the SMART-3S in December 2014.This preliminary test showed that only 2 from 5 parameters were allowed; those are the pH and the Total Suspended Solid (TSS). Meanwhile, the Chemical Oxygen Demand (COD), Biological Oxygen Demand (BOD) and oil and flat exceeded the maximum level.

Table 1. Analytical Result of the liquid compost from the SMART-3S, status per December 2014.

No.	Parameters	Unit	Method of Analysis	Result of Analysis	Maximum Level
1	pH	-	SNI 06-6989.11-2014	6.1	6-9
2	Total Suspended Solid, TSS	mg/L	SNI 6989.3-2004	96	100
3	Chemical Oxygen Demand, COD	mg/L	SNI 6989.2-2009	619	200
4	Biological Oxygen Demand, BOD	mg\L	JIS K 0102-21-2008	327	100
5	Oil and Fat	mg/L	SNI 6869.10-2011	26	10

Source: Center of Environmental Impact Laboratory, Environmental Agency, 2014

Further test was conducted after the liquid compost has been processed for the next 4 months. This time the test included some major nutrients and metal contents. In general the major nutrients in the liquid compost are lower than the standard requirements and therefore is considered poor. Many of them, such as the C-Organic, N, P_2O_5, K_2O, could not meet the requirement. The pH and some metal contents like As,Pb, Hg, and Cd are suitable with the standard requirement. But many of other metal contents like Fe, Cu, Mn, Zn, B and Co are far bellow the requirement. Besides, the E.coli and the salmonella Sp in the liquid compost exceeded the limit.

In order to get clearer composition of nutrients, the analyses result was compared with the wastewater from the same household. In general, the allowed parameters in liquid compost are also allowed for the wastewater material and so does the exceeded parameters. The main difference is, that the E.coli and the salmonella Sp did not exceed the maximum requirement that is 36 from the limit of 100. Please refer to table 2 for more detail result.

Table 2. Analytical Result of the SMART 3S liquid compost and Wastewater status per April 2015.

No.	Parameters	Unit	SMART 3S	Waste water	Method of Analysis	Standard Requirement
1	C-Organic	%	0.1	0.04	Walkley and Black/Spectronometri	Min. 6
2	As	ppm	0.4	n.a.	HNO_3/F-AAS	Maks 2.5
3	Hg	ppm	0.01	0.02	HNO_3/F-AAS	Maks 0.25
4	Pb	ppm	6.4	0	HNO_3/F-AAS	Maks 12.5
5	Cd	ppm	0.1	n.a.	HNO_3/F-AAS	Maks 0.5
6	pH	-	5.8	7.2	Potensiometri/pH meter	4-9
7	N	%	0.07	0.07	Kjeldah/destilasi	3-6
8	P_2O_5	%	2.0	10.2	HNO_3/Spectronometri	3-6
9	K_2O	%	21	23.3	HNO_3/F-AAS	3.6
10	E.Coli	MPN /ml	270	36	Permentan no.70/Permentan/SR.140/10/2011	Maks 100
11	Salmonella sp.	MPN /ml	210	36	Permentan no.70/Permentan/SR.140/10/2011	Maks 100
12	Fe	ppm	0.2	1.3	HNO_3/F-AAS	90-900
13	Mn	ppm	0.2	0.4	HNO_3/F-AAS	250-5,000
14	Cu	ppm	0.2	0.1	HNO3/F-AAS	250-5,000
15	Zn	ppm	4.5	5.6	HNO_3/F-AAS	250-5,000
16	B	ppm	2.0	2.7	HNO_3/Spectronometri	125-2,500
17	Co	ppm	0.1	0.05	HNO_3/F-AAS	5-20
18	Mo	ppm	n.a.	n.a.	HNO_3/F-AAS	2-10

Source: Center of Soil Research Laboratory, the Ministry of Agriculture, 2015

Although the first investment is quite high for buying the shredder, the SMART-3S is considered as a low cost and low maintenance technology for a long term. However, since the level of difficulty of the pretreatment and post treatment of the compost products are different, it is better to characterize the specific locations to implement this technology. As the SMART-3S needs more tools and it is located to the sink in each house, it is suggested to be allocated for mid-to-high income housing settlement and to have a communal wastewater treatment from this facility only and other houses with sink in the kitchen. It is also assumed that not every house in the level of mid-to-high income has own sink in their house.

Table 3. The characteristic of The SMART-3S System.

No.	Parameter	The SMART 3S
1	Technology	Not new
2	Tools requirement*	Shredder, storage, sink
3	Land requirement	200 cm x 100 cm
4	Investment Cost*	Around 380 US$
5	Operational	Private and communal
6	Housing characteristic	mid to high income high-rise, settlement/apartment, Low to mid income, single landed house with sink and centralized wastewater treatment plant
7	Product	Liquid fertilizer
8	Cultivation (1 household)	Up to 1 year (depend on the size)

*excluding the land ownership and working tools

4. Conclusion and Recommendation

Urban people need solutions to manage their waste from home, under a simple technology, less-space and direct benefit to their life [3]. Waste treatment at source guaranty the products of these systems contain less contaminants. This technology cut-off some additional budget and costs. The SMART-3S adjusts urban people's life, while promoting urban waste separation at source and waste-as-resources principles so the result is highly efficient. These technologies are suitable for the housing settlement in densely populated area in Indonesia where the MRF is absence. It can be implemented in private house and/or decentralized level. They promote waste-as-resources principle, by applying urban farming such as verticulture, and permaculture at the front/back yard for organic products, thus creating zero waste from organic waste.

As the technology applied at the community's house, the main responsible person to take in-charge is also the community and the owner of the house. Thus the local authority can manage the non-organic waste and the residual of the organic waste. However, table 2 shows the poor nutrients of the liquid compost from the SMART 3S. As it promotes the quality food production through organic waste reduction, for further treatment it is recommended to explore some nutrition and specific treatments in order to improve the quality of the liquid compost as soil improver. Moreover, to convince the safety food for consumption it is recommended to do further testing to the food/harvested products.

References

[1] PRB. (2014): The Population reference Bureau: 2014 Population Datasheet. Link: http://www.prb.org/pdf14/2014-world-population-data-sheet_eng.pdf (retrieved May 13th 2015)

[2] Krogmann, U.; Koerner, I. (2000): Technology and Strategies in Composting. In H.J. Rehm, G. Reed, P. Puehler & P. Stadler (Eds.), Biotechnology, Vo. 11-C, Second, Completely Revised. Weinheim, Germany: Willer-VCH.

[3] Candra-Dewi, O. (2013): A Decision Support Model for Waste Management in Support of Developing Low Carbon, Eco-Region, Ph.D Thesis, Hamburg University of Technology.

TE-O_08 GHG emissions from manure management – Emissions, mitigation and interactions

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1. Objectives

This study focuses on manure management as a source of greenhouse gas (GHG) emissions to

- Understand emission sources and quantify GHG emissions
- Identify effective GHG mitigation options and estimate their reduction potential to reduce GHG emissions
- Determine trends and estimates of pollutant interactions from implementing proposed mitigation options
- Develop an integrated systems approach to estimate emission reductions from mitigation options, accounting for interactions
- Improve GHG emissions, mitigation options and pollutant interactions in existing parameterization of the integrated assessment model – Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) developed by the International Institute for Applied Systems Analysis (IIASA)

2. Methodology

In this study we investigate emission sources, abatement methods and pollutant interactions, in order to develop an integrated approach to mitigate GHG emissions from manure management for the European Union (EU-28). Mitigation of GHGs, nitrous oxide (N_2O) and methane (CH_4) are the primary focus, although interactions with ammonia (NH_3) emissions are also analyzed. The methodology consisting of five steps is described below and also illustrated in Figure 1.

- Step 1: Identify sources of emissions within the manure management chain
- Step 2: Construct a reference scenario, along with their emission estimates
- Step 3: Identify GHG mitigation options and quantify their reduction potential
- Step 4: Estimate emission reduction w.r.t reference scenario, accounting for interactions
- Step 5: Develop an integrated manure management model incorporating mitigation options and pollutant interactions

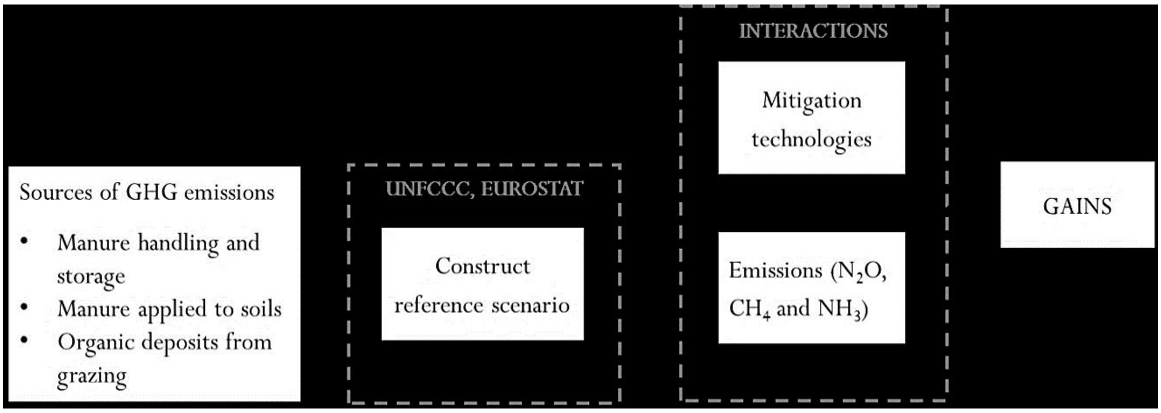

Figure 12: Modelling framework.

3. Results and discussion

3.1 Emission sources

Manure management is a major source of GHG emissions in the agricultural sector. It contributes to around 37% of GHG emissions associated with total agricultural emissions. The sources of emissions from manure management are emissions from animal housing, manure storage and handling, application of manure to soils and organic deposition by grazing animals. In addition there are also indirect emissions from leaching and volatilization of organic matter. Table 1 gives a breakdown of the GHG emissions associated with various sources within manure management. Indirect emissions are calculated separately and assigned to the respective emission source in the table. Nitrous oxide contributes to major share of the emissions (70%) as compared to methane (30%). Emissions from housing and storage represents the largest source. Emissions within each source is further sub-divided and separated based on different animal and manure types (solid, liquid and slurry). Country specific emissions data for EU-28 is obtained from FAOSTAT (2010), EUROSTAT (2009) and UNFCCC (2010) databases.

Table 11: Breakdown of emissions from manure management for EU-28 (2012).

Emission Sources	CH_4 (Gg CO_2 eq.)	N_2O (Gg CO_2 eq.)	Total (Gg CO_2 eq.)	% contribution to total emissions
Housing and Storage	51,585	30,487	82,072	47%
Organic deposition from grazing	NA	43,278	43,278	25%
Animal manure applied to soils	NA	49,311	49,311	28%
Total	51,585	123,077	174,662	

3.2 Mitigation options

The mitigation options considered in this study refers to technologies that are currently available in the market and in use. Hence, more speculative mitigation options are not considered. Mitigation options are identified based on different stages within the manure management flow. This is illustrated in Figure 2. It is important to note that implementation of mitigation measures for a particular emission source may affect emissions from another emission source. This interaction is described in detail for N_2O, CH_4 and NH_3 and subsequent changes in emissions are quantified. Country specific emission reduction potential for the described mitigation options is derived from published literature and reports. A sample table outlining the emission reduction potential and interaction effects of selected mitigation options are presented in Table 2. An integrated approach will be developed using the emissions reductions and associated interactions to estimate the overall effectiveness of a single and combination of mitigation options.

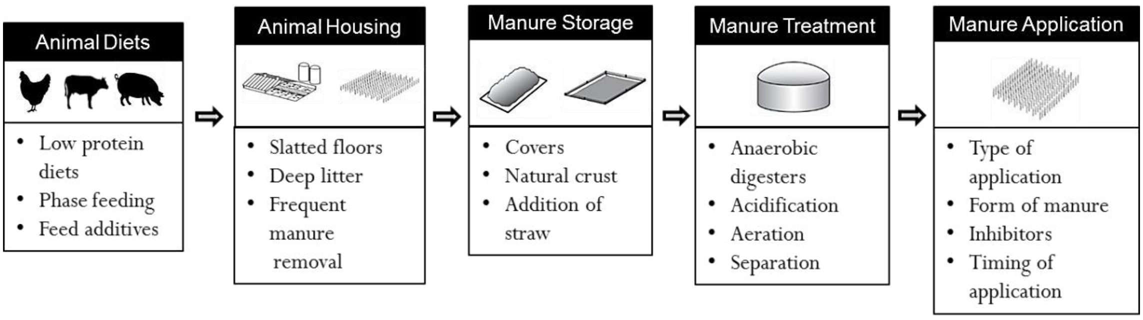

Figure 2: Manure management chain and proposed mitigation options.

Table 12: Emission changes (relative to Reference). Compilation from a literature survey comprising peer reviewed studies (preliminary results)

Mitigation Stage	Reference Scenario	Mitigation Options	Emission Changes		
			N_2O	CH_4	NH_3
Housing	Deep Litter	Slatted Floors	-55%	+54%	-9%
		Frequent Removal	-41%	+55%	-9%
Storage	No Cover	Artificial Film	-66%	-33%	-89%
		Chopped Straw	+100%	-3%	-71%
		Granules	+100%	-5%	-75%
		Wooden Lid	-17%	-16%	-32%
		Granules + Acids	NA	-26%	NA
		Surface Crust	0	-11%	-57%
Application Material	Raw Slurry	Digestate	+2%	NA	-11%
		Liquid Fraction	0%	NA	-24%
		Solid Fraction	-16%	NA	+132%
Application Technique	Surface Spreading	Injection	+119%	NA	-71%
		Incorporation	+119%	NA	-58%
		Band Spreading	+51%	NA	-44%
		Inhibitors	-35%	NA	NA
Feeding	Conventional Feed	Reduced CP (1 to 3%)	NA	NA	-29%
		Reduced CP (3 to 5%)	NA	NA	-54%
		Reduced CP (>5%)	NA	NA	-81%
		Addition of Tannins	NA	NA	-29%
Treatment	No Treatment	Anaerobic Digesters	-61%	-52%	NA
		Slurry Separation	+200%	-34%	NA
		Acidification	NA	-26%	-23%
		Composting	NA	NA	NA
		Aeration	+82%	-57%	NA

3.3 Implementation in GAINS

GAINS uses exogenously supplied current and projected activity data, emission factors, resulting GHG emissions and the technical potential for emission controls along with the costs of these measures. The model also has the capability to account for the interactions between emission controls of different pollutants (Amann 2011, Isaksson 2012). Total emissions of each pollutant are calculated according to Eq. 1. The equation accounts for cases where emission controls are applied and where it is not (*eff* = 0).

$$E_{i,p} = \sum_{j,a,t} \{A_{i,j,a}\} \text{ x } \{ef_{i,j,a,p}\} \text{ x } \{1 - eff_{t,p}\} \text{ x } \{X_{i,j,a,t}\} \qquad \text{Eq. 1}$$

Where

i, j, a, t country, sector, activity, abatement technology

$E_{i,p}$ emissions of the specific pollutant p in country i,

A activity in a given sector,
ef "uncontrolled" emission factor,
eff removal efficiency, and
X actual implementation rate of the considered abatement.

The cost estimates are a combination of investment, operating and maintenance costs along with the cost-savings from mitigating a unit of the pollutant. This is indicated in Eq. 2. Abatement technologies considered here are assumed to be accessible to all countries at the same conditions in GAINS. However, country specific costs are differentiated and calculated based on possible capacity of operation, applicability of specific abatement measures, wage rates and resource availability.

$$C_{ton} = \frac{I^{an} + OM - CS}{E_{ton}} \qquad \text{Eq. 2}$$

Where

C_{ton} costs per ton of pollutant removed
I^{an} investments
OM operating and maintenance cost
CS cost savings
E_{ton} emissions mitigated

Different environmental targets ranging from maximum technically feasible reductions to lower mitigation targets generates different abatement strategies for GHG emissions from management of manure. Country specific emissions reductions and abatement costs depend on environmental target levels. Cost functions will help compare and contrast between countries, emission sources and abatement options.

4. Conclusion and outlook

The agricultural sector is a source of food and is indispensable to society. However, it is a major source of GHG emissions (~10–12 $GtCO_2$ eq./yr). International agreements to reduce agricultural emissions have not been effective, in part due to missing concepts of realistic "low- carbon" situations. GHG emissions from management of manure is an important emission source within the agricultural sector. There is also significant interactions between GHGs and NH_3. This study quantifies and characterizes GHGs from management of manure and identifies low-carbon mitigation strategies accounting for interactions between GHGs and NH_3. This holistic approach and results would be helpful for farmers, crop advisors and policy makers struggling in a sector that itself is highly vulnerable to climate impacts.

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References

[1] UNFCCC: Common Reporting Format (CRF) tables and National Inventory Reports (NIRs) (downloaded on May 2015), United Nations Framework Convention on Climate Change.
[2] IPCC: 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Intergovernmental Panel on Climate 10 Change, Japan, 2006.
[3] Amann, M., Bertok, I., Borken-Kleefeld, J., Cofala, J., Heyes, C., Höglund-Isaksson, L., Klimont, Z., Nguyen, B., Posch, M., Rafaj, P., Sandler, R., Schöpp, W., Wagner, F., Winiwarter, W. Cost-effective control of air quality and greenhouse gases in Europe: modeling and policy applications. Environmental Modelling and Software 26, 1489-1501 (2011). http://dx.doi.org/10.1016/j.envsoft.2011.07.012
[4] Lena Höglund-Isaksson, Wilfried Winiwarter, Pallav Purohit, Peter Rafaj, Wolfgang Schöpp, Zbigniew Klimont. EU Low Carbon Roadmap 2050: Potentials and costs for mitigation of non-CO2 greenhouse gases. Energy Strategy Reviews 1, 97-108 (2012). http://dx.doi.org/10.1016/j.esr.2012.05.004

TE-O_09 Urban farming to grow a greener future

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1. Objectives

In 2050 the world population is estimated to grow up to 9.1 billions demanding for a 70% raise in food production [1]. In a world increasingly urbanized and characterized by a scarceness of green spaces and natural resources, cities have been playing, along the recent years, an important role in the conservation and increment of productive areas by implementing crop production within urban limits, and using organic materials as fertilizers. Although in many European cities urban gardening has a long tradition, currently in expansion and fulfilling other roles than food production, in other continents urban farming represents a major source for population subsistence, namely in southern Asia, in Latin America and in Africa, where it stands with economical importance since the early 90's [2]. This concept has been evolving to Northern America, to Europe and to Australia with contrasting objectives of pleasure and recreation, socialization and social inclusion, education, aesthetics, environment and therapy [3]. However, in Portugal the concepts of urban and peri-urban agriculture (UPA) are still a quite recent novelty in academic and political agendas and regulated urban gardens start emerging from the 2008s onward, largely due to the deep economic crisis that has been hitting the country since then. The major goal of this paper was to understand the main gardeners' motivations to cultivate urban plots located in the first three regulated Urban Parks implemented by the Lisbon City Hall.

2. Methodology

Three pilot municipal Urban Parks located in three different Lisbon neighborhoods implemented by the City Hall were studied (*Quinta da Granja, Campolide* and *Telheiras*). These Parks were selected because their gardeners were the unique with some experience about gardening on a Municipal Horticultural Park (HP) when the fieldwork started. To evaluate urban farming potential to contribute to a greener growth, the link between townsmen residents and farming motivations was evaluated by enquiring 49 from a total of 80 gardeners. Questionnaires were conducted face-to-face, from June to September 2013, and included several topics in order to: (i) identify the demographic and socioeconomic profile of the respondents, (ii) former links with agriculture and the rural, (iii) gardening frequency, daily time spent on the garden and agronomic practices, (iv) characterization of the farm, and (v) gardener's motivations and their perceptions about the contributions of urban agriculture to the city and city dwellers. Gardeners were approached in the garden and the surveys were realized early in the morning or at the end of the day and during the weekend when they were gardening.

3. Results and discussion

3.1 Urban gardening in context

Most of the respondents in the three case-study gardens are men (61%), middle aged (69% are over 45 and less than 64 years old), with high levels of education (61% have 12 years of formal school or higher education), and are actively employed (57%). They live in small households, e.g. with two or three people (55%), and 16% of the total gardeners households have someone unemployed (Table 1). This last percentage reflects the unemployment rate in the country that has been increasing from the 2008s onwards, reaching 16.2% in 2013 [4], in consequence of the economic crisis. It is interesting to note the close link of the most part of the respondents with the rural areas (82% and 57% frequently visit and/or spent their childhood in rural territories, respectively), and the agricultural activity: 31% had already exercised the agricultural activity before migrating to the city, 37% are sons/daughters of farmers and 31% still have relatives and/or friends who farm a small plot of land (Table 1).

In general, gardeners live very close to the cultivated allotments, which is a prerequisite to apply for a plot in a Municipal Horticultural Park. Gardening is a routine task for most gardeners who spend less than 3 hours (61%) cultivating a considerable diversity of vegetables every day. This was observed in the HPs Campolide and Telheiras. Gardeners from Quinta da Granja HP are those who spend more hours in their garden (between 3 and 7 hours per day: 39%) and also perform their activity more frequently: every day including Saturday. For most gardeners, farming is not an activity that they perform individually since 67% of the gardeners are helped or at least have the participation of their wife/husband or, less frequently, of a son/daughter.

Table 1: Survey results on demographic and socioeconomic characteristics of inquiree.

Data type	Overall	Quinta da Granja	Telheiras	Jardins de Campolide
Sample (n)	49	22	16	11
Gender:				
women (%)	39	32	44	46
men (%)	61	68	56	54
Age group				
30-34	1	1	–	–
40-49	9	3	4	2
50-59	16	4	7	5
60-64	11	4	4	3
65-69	6	4	1	1
≥ 70	6	6	–	–
Education				
illiterate	2	1	–	1
6 years of formal school	15	11	1	3
9 years of formal school	2	1	–	1
12 years of formal school	10	4	2	4
higher education	20	5	13	2
Occupation				
actively employed	28	10	10	8
retired	19	12	6	1
unemployed	2	–	–	2
Household				
one person	4	3	–	1
two people	23	11	5	7
three people	10	4	4	2
four people	12	4	7	1
with someone unemployed	8	3	1	4
Former links with agriculture and the rural				
Son/daughter of farmers	18	11	2	5
Former farming experience	15	8	2	5
Relatives/friends still farming	15	5	9	1
Time spent in the rural during the child hood	28	10	13	5
visiting regularly the rural	40	17	15	8

When urban farming is supervised by municipalities in developed countries, it usually follows some rules of organic farming for economic reasons which reduce or prevent the use of commercial fertilizers, or to avoid risks of environmental contamination and to promote public health. Agriculture is known as an activity with a potential for environmental contamination from fertilizers inefficiency and from the use of pesticides. The opposite is expected from urban farming.

Agricultural practices used by the gardeners from the three HPs do not differ significantly among each other. Tillage is always done manually, using a hoe or a garden rake, to prepare seed bedding. The soil is always kept free of weeds which are removed by hand and used after for the production of a homemade sort of compost. Gardeners claim that keeping their garden free of weeds is both a necessity to eliminate competition with vegetables, and a question of aesthetics. The majority (92%) of the inquiries use the compost they prepare out of the material collected at weeding and food waste, to fertilize the soil, representing a spare of mineral fertilizers and resources conservation. Manure collected from local animal breeding facilities is also used for fertilization. Horse, goat, chicken or rabbit manure are the most frequently used and generally free, from military institutions (National Guard) breeding horses located in the vicinity, or from relatives or friends raising poultry. Only four respondents said they have used mineral fertilizers for soil amendment, a practice which is permitted by the garden code.

No gardener claims to have used commercial pesticides, but a few elderly use copper sulphate. Due to the low precipitation level in Lisbon, water stress may occur in Spring and Summer, when the gardens need to be irrigated. Water is applied using a can or a hose connected to the public supply network. Gardeners do not capture rainwater for future watering.

Because gardeners do not use chemical fertilizers nor pesticides they label their gardening practices as natural or organic and identify themselves as organic producers. However, and although the city Council provided them some technical sessions about organic farming when allotments were distributed, gardeners ignored or misunderstood what organic practices really are. In other words, many gardeners classify their practices as organic because they have in mind two models of agriculture: the “bad agriculture” (large-scale, intensive, polluting and producing non-healthy food) and the one they practice and clearly identify [3]. In contrast with gardener’s self-evaluation about their “good and sustainable agricultural practices” a recent study reveals that urban garden production systems, namely in Quinta da Granja HP, used high application rates of nitrogen and water. Both the use of high doses of nitrogen from organic amendments, which surpass crop requirements, and the excess of irrigation has important negative impacts on environment and human health [5].

3.1 Benefits from gardens and the gardeners motivations

In this experimental analysis, urban gardening has proven to be an activity with multiple social, health and economic benefits, which serves to mitigate problems related to food insecurity, unemployment and social disintegration. Most gardeners see socialization with other gardeners as a factor which has contributed to improve the quality of their lives. Gardening is not only an individual activity but also a moment to socialize in gardening context, to share technical advices, to exchange seeds, to disseminate less frequent vegetable species which are not usually included in the traditional Portuguese diet, for example zucchini, eggplant or sweet potato, among others. Due to this diversification of species grown in the HPs, 43% of the respondents claim to have cultivated or eaten one or more vegetables for the first time. Apart from this innovation and diversification in the diet, gardening has allowed the gardeners and their families to have a more healthy diet: 67% of the respondents eat more quantity of vegetables and with more frequency.

All the vegetables grown in the HPs serve for domestic consumption (c. 100%). Whenever there is surplus and the vegetables production is not totally consumed by the gardener in his(her) household, it is offered to other members of the family, mainly sons/daughters (53%) or friends (14%). Just one gardener donated the vegetables produced to a food bank and one other sold the surplus. Although saving money is far from being a driving motivation for gardening, the vegetables grown for domestic consumption have a positive impact in reducing household vegetables purchase: 45% of the respondents estimate savings between 50% and 75% of the total amount of money spent to buy vegetables, and 41% value these savings between 25% and 50%.

Finally, gardening has other kind of benefits. As gardeners, in particular the elderly, retired and those who fall into the most modest social groups enjoy to emphasize “when I am here [in the garden] I am doing something”, “I garden because I like and I do something useful” or “gardening means to spend my free time in a productive way”. In short, urban agriculture promotes social inclusion within society of the elderly and poorest, in particular, who by gardening can openly show not only that they carry with them specific skills and knowledge, but also that the latter are useful both for themselves and society. The usefulness or the positive contributions of gardening to society and the city are issues quite well perceived by gardeners: 47% assume that urban agriculture contribute to the beautification of the city; 41% are proud for providing and disseminating agricultural activity among children and city dwellers, and ¼ nurture the reconnections between the rural and the urban, or bring city dwellers close to nature.

When asked how they used to spend the time currently occupied with gardening, 78% said "at home/alone/doing nothing" [3]. Gardening is also a way socialize in a friendly atmosphere and pleasant context, "where one can listen to the birds", to foster interaction between social groups and, in some situations, to promote a mix of generation. At the same time, gardeners recall good memories when physically reconnect with the land. The socialization role of the gardens goes beyond contacts between gardeners: almost 2/3 of the respondents use gardens to make outdoor meals with friends and/or relatives, during the spring and or summer [3].
The gardeners motivations for cultivating indicated in the questionnaire is in the following order: pleasure or passion for agriculture and working with the land (71%), to grown their own food/to be sure about the good quality of the consumed vegetables (55%), to practice organic farming, as they perceive it, (33%), to occupy free time (31%), to remind/recall the rural life-style (25%), to have physical contact with nature (25%), to socialize/make friends or to make physical activity (23%), to save money (20%). Only very few identity gardening with a way to develop creativity in decorating the plot (12%) and fewer mentioned motivations related with promoting biodiversity (three respondents from Telheiras HP) [3].

4. Conclusion and outlook

Gardeners from the recent regulated Municipal Urban Parks are predominantly men over 45 years old, with a great percentage of retired individuals. Urban farming is not related to unemployment, but with pleasure or passion for agriculture and working with the land, along with the need to alter consumption habits towards the use of organic products and zero use of fertilizers. The majority of urban farmers have or had a previous link to rural areas or/and farmer's history in the family.
Although in Lisbon the still tiny available area occupied by regulated urban gardens, compared to other European cities, has resulted less as a strategy of public policy for the alleviation of poverty, but as a way to diversify the Municipal Ecological Infrastructure and recover part of the city landscape occupied with non-regulated urban agriculture, we consider that the Lisbon city Council approach to urban gardening is a missing opportunity to recognize and value the benefits of urban gardening to the city, as evidences from many European cities have been proving. Resource recycling and conservation, therapy and recreation, education and safe food provision, green architecture, open space management, besides food security in times of economic crisis are some illustrative examples of those benefits. In addition, urban gardening should also be used by the city Council as a tool to promote community development, urban governance and civic participation, as well as the environmental benefits which include the revitalization of brownfield sites in the cities, improved air quality and reduced energy consumption for food transportation, improved urban biodiversity and a greener economy. These are crucial assets still underdeveloped in the Portuguese society, when compared to Central and Northern European countries.
Finally, the city council also could make use of Urban Parks to disseminate among urban dwellers the several advantages to grow one's own food, and to promote biodiversity and better agronomic practices among urban gardeners.

References

[1] FAO 2009. How to Feed the World in 2050?. http://www.fao.org/fileadmin/templates/wsfs/docs/expert_paper/How_to_Feed_the_World_in_2050.pdf;
[2] FAO 2012. Programa de Horticultura Urbana e Peri-Urbana da FAO. Available at: http://www.fao.org/docrep/015/i1610p/i1610p00.pdf;
[3] Gonçalves RGG 2014. Hortas urbanas. Estudo do Caso de Lisboa. MSc thesis Instituto Superior de Agronomia, Portugal. 130 p.
[4] INE 2011. National Statistics Institute. Census from 2011.
[5] Cameira MR, Tedesco S, Leitão TE 2014. Water and nitrogen budgets under different production systems in Lisbon urban farming. *Biosystems Engineering*, 125: 65-79.

TE-O_10 Waste recycling through integrated farming system – An Assam agriculture experience

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1. Objectives

Waste minimization, more precisely waste recycling has become a world-wide concern now-a-days. This is mainly because of the serious problem of waste accumulation creating pollution, health hazard and natural havoc particularly in developing countries like India. In such a system wastes of one enterprise are treated as resources or raw materials for another. In the present study an attempt has been made with the following objectives to explore the possibility of waste management through recycling in an Integrated Farming System (IFS) model in rainfed areas with judicious use of wastes and farm resources.

- To analyse the possibilities of recycling the agricultural waste in the Integrated Farming System model to sustain agricultural productivity.
- Improving economic viability through zero waste strategies or optimization of waste minimization.
- To present a zero waste model focussing essentially on using available scarce resources and waste recycling more efficiently, effectively and sustainably for natural resources.

2. Case study description

Assuming a family having 5 members with 10,000m^2 cultivated area, an Integrated Farming System (IFS) model with Crop + Cattle + Fishery + Apiary components under rainfed situation was developed to meet their annual food requirement as far as possible and to present a zero waste model after conducting field experiment under All India Coordinated Research Project on Integrated Farming System at Assam Agricultural University, Jorhat, Assam, India. The data presented here relates to the year 2013-14.

In this system, animals reared for milk were fed on crop waste along with fodder crops and the voids in turn were used as manure for crops grown in the system, in fishery unit, for vermicomposting and for liquid manure production. De-siltation of Fish ponds were carried out after every two years and the fertile silts were used for the cultivation of horticultural crops.

- Components: Field crops (Cereals, pulses, oilseeds, fodder crops), horticultural crops (fruits and vegetables), fodder crops, livestock (2 milch cows and 1 Heifer), Fishery (two ponds with 500 m^2 and 420 m^2 area), Apiary (5 beehives), Vermicompost unit (8.41 m^3), Liquid manure production unit (1.79 m^3), Biogas unit (2 m^3).
- Vermicompost unit: The vermicompost were produced four times in a year and cow dung along with the crop residues/ bio-wastes were used in the ratio of 2:3. Crop residue/ bio-waste include dried water hyacinth, vegetable wastes, straw and wastes obtained at the time of winnowing of rice.
- Liquid manure unit: Every day output of cow dung and urine mixture was channelized to a four-chambered tank and liquid manure was produced through the process of sedimentation.
- Bio-gas unit: Initially 2000 Kg cow dung along with required amount of water were poured in the bio-gas plant and kept for fermentation for a period of 45 days. When release of gasses were observed, then 55 Kg cow dung per day was used to continue the release of gasses which were utilized as fuel.

3. Observation

3.1 Recycling of agricultural waste in the Integrated Farming System model Bio-waste production in the IFS

In the IFS model designed, total quantity of bio-waste obtained from different components were quantified and presented in table1.

Table 1: Bio-waste production in IFS model.

Items	Quantity produced
Paddy straw (Kg)	2920
Crop residue (Kg)	1820
Litter fall (Kg)	2050
Cowdung (Kg)	23119
Cow urine (lit)	14235
Silt (tons)	146.40
Rice bran (Kg)	802.56
Mustard oil cake (Kg)	240.60

Bio-wastes obtained from different components of the system are very rich in essential plant nutrients which are presented in Table 2.

Table2: Nutrient content of different bio-waste.

Bio- waste	Nutrient content		
	Nitrogen (%)	Phosphorous (%)	Potassium (%)
Paddy straw	0.28	0.17	0.27
Crop residue	0.30	0.19	0.32
Litter fall	1.29	0.09	1.00
Cowdung	0.4	0.2	0.1
Cow urine	0.5	0.003	1.35
Silt	0.0293	0.025	0.0175
Mustard oil cake	4.50	1.50	1.00
Vermi-compost	2.04	1.22	2.58
Liquid manure	0.66	0.32	0.50

Bio-waste utilization in the IFS

Bio-wastes obtained from the system were utilized in different components within the system itself and are presented in Table 3.

Table 3: Bio-waste recycling in IFS model.

Bio-waste	Quantity produced (Kg/lit/ton)	Recycled within the system in different component (Kg/lit/ton)						
		Crop unit	Hort. unit	Dairy unit	Vermi-compost unit	Fishery unit	Liquid-manure unit	Bio-gas unit
Un-processed waste								
Paddy straw	2920	-	-	2800	120	-	-	-
Crop residue	1820	-	-	1220	600			
Litter fall	2050	Used for organic amendment of the soil						
Cowdung	23119	2000	-	-	480	204	835	19600
Cow urine	14235(l)	-	-	-	-	-	14235	-
Silt	146.40 (t)	-	146.40	-	-	-	-	-
Rice bran	802.56			252.56		550		
Mustard oil cake	240.60					240.60		
Processed waste								
Vermi-compost	544	544		-	-	-	-	-
Liquid manure	4866.67 (l)	-	4866.67	-	-	-	-	-
Bio-gas	511 m^3	Used to meet up fuel requirement of the household						

Processing and value addition of bio-wastes

Some of the bio-wastes generated within the system were processed as vermi-compost, liquid manure and bio-gas to increase their nutrient content, reduce bulkiness for easy handling, reduce nutrient loss and increase market value.

Vermi-composting
Vermi-compost was prepared in 8.41 m^3 vermi-compost unit from cow dung, paddy straw and different vegetables and fruit wastes like discarded cabbage leaf, waste obtained at the time of winnowing of rice, dried water hyacinth, semi dry banana leaves *etc.* From 120 Kg cowdung and 180 Kg crop residue including paddy straw (bio-waste) of the IFS models itself 136 Kg vermi-compost was produced in one cycle. The process was carried out for 4 cycles thereby generating 544 kg vermi-compost annually in the system.

Liquid manure production
Every day output of cow dung and urine mixture from the cowshed were channelized to a four chambered tank and through the process of sedimentation in the three chamber, complete liquid manure was collected in the 4th chamber. The collection of liquid manure was quantified as 40 litres/three day thereby in a year approximately 4866.67 litres of liquid manure was obtained.

Bio-gas plant
Cow dung obtained from the cattle component was also utilized to produce bio-gas in a 2m^3 size bio-gas plant. Bio-gas production was quantified approximately as 1.4 m^3 per day and thereby in a year 511 m^3 bio-gas was produced which is equivalent to 219.73 Kg of Liquid Petroleum Gas.

3.2 Improving economic viability through zero waste strategies

Besides improving environmental quality, waste recycling reduces the requirement of market purchased input for farmers thereby decreasing cost of production. This in turn increases the net return for farmers and secures economic viability by sustaining farm income. Table 4 shows the market value of bio-waste produced in the IFS model.

Table 4: Market value of bio-waste produced in the IFS model.

Bio-waste	**Quantity produced (Kg/lit/ton)**	**Price/unit (Rs.)**	**Market value (Rs.)**
	Un-processed waste		
Paddy straw	2920 (Kg)	1.00	2920.00
Crop residue	1820 (Kg)	0.30	546.00
Litter fall	2050 (Kg)	0.20	410.00
Cowdung	23119 (Kg)	0.50	11560.00
Cow urine	14235 (Kg)	0.50	7118.00
Silt	146.40 (ton)		
Rice bran	802.56 (Kg)	10	8026.00
Mustard oil cake	240.60 (Kg)	12	2887.00
Total			33467.00
	Processed waste		
Vermi-compost	544 (Kg)	10	5440.00
Liquid manure	4866.67 (liter)	0.562	2735.00
Bio-gas	217.5 Kg LPG (511 m^3)	63.72	13860.00
Total			22035.00

Market value of different bio-wastes prevailing at the locality were considered.

Total requirement of paddy straw and crop residues required for the cattle and vermi-compost unit was obtained from the bio-wastes itself and thereby saving an amount of Rs.3466.00 approximately. Similarly rice bran and mustard oil cake produced as a bi-product of paddy and toria (rapeseed) processing which were used as fish and cattle feed saved about Rs.10913.00. The processed wastes, vermi-compost, liquid manure and bio-gas saved a total of Rs.22035.00 for the household whereas un-processed waste amounted to Rs.33,467.00.

Bio-wastes produced in the system (both processed and unprocessed), also substituted a major amount of chemical fertilizer required for the crop components in the system. Table 5 shows substitution of chemical fertilizers by the bio-wastes both in terms of quantity and money value in the system.

Table 5: Substitution of chemical fertilizers by the bio-wastes (in terms of quantity and money value) in the system.

Bio- waste	Quantity produced (Kg/ton/liter)	Amount of chemical fertilizer substituted (Kg)			Market value (Rs.)
		Urea	SSP	MOP	
Paddy straw	2920	17.74	31.03	13.17	724.66
Crop residue	1820	11.85	21.61	9.73	509.68
Litter fall	2050	57.39	11.53	34.24	1305.40
Cowdung	23119	200.67	288.99	38.61	5591.56
Cow urine	14235	154.45	2.67	320.93	7347.89
Silt (ton)	146.40	93.08	228.75	42.79	3988.46
Vermi-compost	544	24.08	41.48	23.44	1077.52
Liquid manure (litre)	4866.67	69.70	97.33	40.64	2401.80
					Total: 22946.97

SSP: Single Super Phosphate; MOP: Muriate of Potash Market rate of Urea, SSP and MOP prevailing in the locality was considered

Thus, the zero waste model designed as IFS model serves as an economically viable model with the system profitability (net return) of Rs.1,26,828.00 with a B:C ratio of 1.68. Such a high B:C ratio could be possible due to recycling of wastes generated within the system.

3.3 IFS model as a zero waste model

Integrated Farming System (IFS) model designed can be considered as a zero waste model on account of utilization of all crop and animal wastes as raw materials and resources for each other through efficient recycling (Figure 1). Jayanthi (2006) also opined that by virtue of adoption of one of the modern agricultural technology *viz.,* Integrated Farming System (IFS) in the farm activity, there is a possibility of improving untapped potential of each and every produce by recycling with dual benefits. As per report of FAO (1985), total production of crop residues and animal voids in India by 2020 AD would be to the tune of 447 million tonnes and 1130 million tonnes respectively, which would cause havoc to the environment if not recycled within the agricultural system.

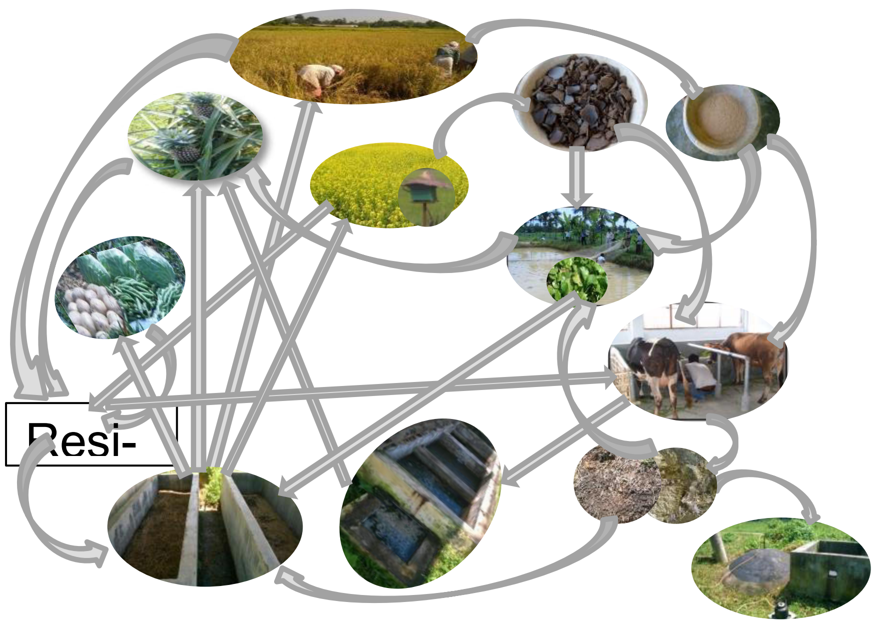

Figure 1: Bio-waste recycling in IFS model

4. Conclusion

Thus, it can be concluded that Integrated Farming System (IFS), besides playing a significant role in securing sustainable production of quality food and other products, also helps in sustaining farm income by reducing the cost of production in one hand and eliminating environmental degradation in other. Due to efficient recycling of agricultural waste as manure, animal feed *etc*., cost of production decreases and soil fertility also improves to a greater extent. Such a system fulfils basic food requirements of the household as well as act as an important approach or strategy for waste management. Therefore, an inferences may be drawn that the IFS model developed can provide an excellent opportunity for organic waste recycling besides reducing farmer's dependency on external market purchased inputs leading to improved farm income in one way and also improving environmental quality in other by reducing the pollution from the bio-waste accumulation.

Acknowledgements

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References

[1] FAO (1985): Production year Book Vol.34. Food and Agricultural Organisation, Rome, Italy.
[2] Jayanthi,C. (2006): Crop and animal waste utilization. Integrated Farming System A Path to Sustainable Agriculture. Second edition. Directorate of Publication, Tamil Nadu Agricultural University Offset Printing Press, Coimbatore-641003, P:31

TE-O_11 Analyzing consumer-related nitrogen flows – A case study on food and material use in Austria

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1. Objectives

National nitrogen budgets trace and quantify flows of nitrogen (N) in and out of various compartments or pools on a national level. Commonly used pools include energy and fuels; materials and products in industry; agriculture; forest, semi-natural vegetation and soils; waste; humans and settlements; atmosphere; and hydrosphere [1]. While the aspects of agricultural food production and fuel combustion dominate in many existing studies due to their large magnitude and environmental relevance, other flows of N related to consumers (i.e., humans and settlements) are frequently neglected. However, it has been estimated that non-food N might account for more than 50% of the (reactive) N that is available for consumers [2].

The aim of this study was thus to extend the boundaries of conventional nitrogen budgets towards the consumer sphere by considering the N contained in human-made goods and compounds. Using Austria in 2010 as an example, following flows have been quantified: N in food and agricultural products, N in non-food industrial products (synthetic & natural polymers, wood & paper products, waste), N related to pets, gardens and green spaces. Full details of the study have recently been reported by Pierer et al. [3].

2. Methodology

The study is based on the principles of material flow analysis and covers flows within the national borders of Austria as well as imports of food or material products. All results refer to annual flows, based on the year 2010, for which data were most consistently available.

In an iterative process, first, potentially relevant flows of N related to products and processes were qualitatively identified. Next, respective mass flows and nitrogen concentrations were estimated and combined to quantify N flows. The analysis covered only the nitrogen contained directly in the respective products, and did not account for N emissions that occurred along the entire product or process chain as in lifecycle assessment or footprint analysis. Nitrogen flows that accounted for more than 100g N per inhabitant and year (i.e., about 850 t N for Austria in total) were considered as relevant. Data sources used include appropriate statistics by Statistics Austria (food supply balance sheets, foreign trade statistics) and FAO (food supply data, production data), as well as national reports such as the Federal Waste Management Plan published by the Federal Ministry of Agriculture and Forestry, Environment and Water Management, scientific literature and industry reports.

To account for uncertainties caused by imprecise or lacking data and necessary assumptions, “uncertainty factors” were assigned to all quantified flows based on the type and quality of the respective data sources used. Uncertainty factors ranged from 1.1 (low uncertainty, e.g. current official statistics) to 4.0 (high uncertainty, e.g. calculation derived from assumptions only). The uncertainty range of the results was determined by dividing (for the minimum value) and multiplying (for the maximum value) the best estimate result by the respective uncertainty factor [4].

3. Results and discussion

As shown in Table 1, the most relevant N inflows to the consumer sphere stemmed from food supply (52% of total inflows of 127 000 tonnes N), but material products were also relevant (28%). Compost and fertilizer application in gardens and public green spaces accounted for 12%, and pet food for 7%. N outflows (92 800 tonnes N in total) could largely be attributed to human excretion and food waste (54% and 13%, respectively), followed by garden waste (16%), waste from pets (10%) and material waste (7%).

Separate analysis of inflows and outflows allowed to identify a gap, a surplus of roughly 34 000 tonnes N. This discrepancy indicated that either flows had been overlooked in this analysis, and/or N accumulated in the form of durable consumer goods. Focusing on household consumption rather than (industrial) production further limited the availability of appropriate data and statistical information. In addition to the main results, Table 1 gives uncertainty factors and the derived minimum and maximum values for all quantified flows.

Table 1: Consumer-related N flows in Austria 2010. Differences in totals due to rounding.

Flow name	N flow (t N a^{-1})	UF	N min (t N a^{-1})	N max (t N a^{-1})
		Inflows		
Food supply	66 200		60 100	72 800
(animal food)	*36 600*	*1.1*	*33 200*	*40 200*
(plant food)	*29 600*	*1.1*	*26 900*	*32 500*
Material products	35 500		23 500	55 800
(synthetic polymers)	*11 900*	*2*	*5 900*	*23 800*
(textiles)	*9 400*	*1.33*	*7 100*	*12 600*
(wood & paper products)	*13 500*	*1.33*	*10 100*	*17 900*
Others (detergents, tobacco)	*800*	*(various)*	*400*	*1 600*
Pet food	9 400	1.33	7 000	12 400
Input to private gardens & public green spaces	15 700		8 400	29 900
(mineral fertilizer)	*2 200*	*1.33*	*1 600*	*2 900*
(compost)	*13 500*	*2*	*6 800*	*27 100*
		Outflows		
Food waste	11 900	1.33	9 000	15 900
Human excretion	50 400		41 600	74 300
(to sewage system)	*47 200*	*1.33*	*38 500*	*68 200*
(to hydrosphere)	*3 100*	*1.33*	*3 000*	*5 300*
(to atmosphere)	*200*	*4.0*	*50*	*800*
Material waste	6 000	2	3 000	12 100
Waste & excretion from pets [a]	9 400	1.33	7 000	12 400
Green waste & garden waste	15 000	2	7 500	30 100
		N balance		
Total inflows	126 700		99 100	171 000
Total outflows	92 800		64 400	138 100
N balance (inflows – outflows)	33 900		34 700	32 900

UF- uncertainty factor;

[a] due to lack of data outflow set equal to inflow;

Food: With 66 200 tonnes N, 60% of which comes from domestic sources, food supply accounted for the largest share of N inflows. However, a remarkable part of the food supplied is wasted (18%). This indicates a large potential for reducing N losses that might be achieved given the growing concern and public awareness related to food waste. It was assumed that the entire N consumed as food was excreted by humans as well. Thus, population dynamics (i.e. N accumulation in the bodies of children, and the net change in population related to births, deaths and migration) were not considered in the study. Depending on sewage connection and treatment, the excreted N either passes through the sewage system or goes directly to the hydrosphere.

In 2010, 55% of the total food nitrogen supplied came from animal-based food, which is a noticeable increase from 47% in 1961 (data not shown). Comparing food supply per capita over time shows that total food supply has increased (from 6.8 kg N/cap/year in 1961 to 7.9 kg N/cap/year in 2010), with a clear rise in animal N supply (+39%) offsetting a slight decrease in vegetable N supply (-2%). In interpreting these figures, it has to be considered that food supply is not equal to food consumption. To derive food consumption, food waste and other losses (such as human-edible food that is fed to pets instead) would have to be subtracted from total food supply. However, since no historical data on food waste are available, this could only be done for 2010 as the principal year of analysis. It seems reasonable to assume, however, that the share of food waste in Austria increased over time, as Europe moved from food scarcity during and after the world wars towards the current situation of massive food surpluses in just a few decades [5].

Pets: Pet food supply accounted for 7% of inflows and was quantified based on recommended protein intake for the most common non-agricultural animals, including dogs, cats, ornamental birds, aquarium fish, small mammals, and pleasure riding horses. Due to lack of data, pet food supply was set equal to waste and excretion from pets, accounting for 10% of all flows on the output side. The uncertainty factor associated with pet food accounts for the fact that although pet food was assumed as a separate inflow, a certain share of it might actually come from human food supply.

Material Products: Accounting for 28% of inflows but only 7% of outflows, material products were the main factor explaining the balance surplus of 34 000 tonnes N. Accumulation of durable consumer goods alone cannot explain this gap. While Gu et al. [6] assume that 25% of material inflows accumulate in the consumer sphere, it would have to be 83% in our case. Two other aspects may contribute to this discrepancy: First, with regard to the outflows, there are residual waste streams that are not included in the waste statistics and/or not directly assigned to households (for instance end-of-life vehicles). Second, with regard to incoming material products, some industrial materials might have been incorrectly assigned to households, overestimating the inflows.
While material products are not commonly quantified in nitrogen budgets, our results suggested that due to their magnitude they are worth considering. As the N is bound in the materials, it can be argued that its environmental effects may be rather small – but that preposition is yet to be confirmed.

Gardens & green spaces: Similar to material products, gardens and green spaces do commonly not get much attention within national nitrogen budgets, as they are assumed to be of minor importance. In addition, data are scarce in particular for private gardens due to their heterogeneous, fragmented, small-structured and private character [7]. However, considering that N flows related to gardens and green spaces accounted for 12% of inflows and 16% of quantified outflows in our study, these aspects deserve closer observation. Although not covered comprehensively in statistics, there are indications that urban gardening systems cause N accumulation due to high N application rates [8]. While generally viewed very positively, the environmental impacts of gardens may need consideration, too. In contrast to agriculture, there are no regulations in place with regard to nutrient use for private garden owners [7]. Further studies in this area could help raising awareness about the environmental effects of different garden management practices (concerning the use of mineral fertilizer and compost, but also pesticides). This might become even more important given the current trends towards urban (and rural) gardening in different forms.

4. Conclusion and outlook

Focusing on the apparent knowledge gap of flows involving non-food products and processes, this study indicates ways to contribute to more complete consumer related nitrogen budgets in the future. Although concentrating on Austria, the study is exemplary in identifying previously neglected relevant flows of N. A better understanding of these flows is needed to shape future policies with regard to nitrogen-efficient consumption. In particular, nitrogen flows related to private gardens, urban gardening and public green spaces and their environmental effects as well as interconnections with conventional agriculture are worth a more in-depth analysis. This applies even more as the results are strongly dependent on national characteristics and the availability of appropriate data sources, with potentially large differences between countries within the EU.

Acknowledgements

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References

[1] UN ECE (2013): Guidance document on national nitrogen budgets. ECE/EB.AIR/119. Executive Body for the Convention on Long-range Transboundary Air Pollution.

[2] Leip, A.; Achermann, B.; Billen, G.; Bleeker, A.; Bouwman, A.F.; Vries, W. De; Dragosits, U.; Döring, U. et al. (2011): Integrating nitrogen fluxes at the European scale. In: Sutton et al. (eds): The European Nitrogen Assessment. Sources, Effects and Policy Perspectives. Cambridge University Press, Cambridge/UK, 345-376.

[3] Pierer, M., Schröck, A., Winiwarter, W. (2015): Analyzing consumer-related nitrogen flows: a case study on food and material use in Austria. Resources, Conservation and Recycling 101, 203-211.

[4] Egle, L.; Zoboli, O.; Thaler, S.; Rechberger, H. & Zessner, M. 2014. The Austrian P budget as a basis for resource optimization. Resources, Conservation and Recycling 83, 152–162.

[5] Evans, D.; Campbell, H.; Murcott, A. (2012). A brief pre-history of food waste and the social sciences. The Sociological Review 60:S2, 5-26.

[6] Gu, B.; Chang, J.; Min, Y.; Ge, Y.; Zhu, Q.; Galloway, J.N. & Peng, C. (2013): The role of industrial nitrogen in the global nitrogen biogeochemical cycle. Scientific Reports 3, 2579.

[7] Dewaelheyns, V.; Elsen, A.; Vandendriessche, H. & Gulinck, H. (2013): Garden management and soil fertility in Flemish domestic gardens. Landscape and Urban Planning 116, 25-35.

[8] Cameira, M.R.; Tedesco, S. & Leitão, T.E. (2014): Water and nitrogen budgets under different production systems in Lisbon urban farming. Biosystems Engineering 125, 65-79.

TE-O_12 Food and phosphorus security – Bridging the global-local and the rural-urban gaps

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1. Objectives

To assess to what extent households, industry and local authorities could contribute to reduce, reuse and recycle P-rich waste products in agriculture. The purpose is to show, conceptually, how to improve global food security through sustainable management of nutrient resources in waste products, and thus prevent prematurely crossing planetary resources boundaries.

2. Methodology

Phosphorus (P) is an essential nutrient for plant growth, and around 80% of mined P is lost in the process of bringing food to the plate. The remaining 20% is usually wasted through sanitation systems [1]. A systems approach to turn nutrient-rich waste into a resource for agriculture presupposes both physical infrastructural adjustments and minor behavioral changes.

A life-cycle thinking is applied here, using an extended-waste-hierarchy comprising both solid and liquid wastes. Step 1 is to reduce (a) waste generation, and to (b) minimise harmful contents in flows and products in order to recover P in wastes. P-rich solid (e.g. biowaste) and liquid (e.g. urine) waste can be reused (Step 2) more or less as they are. Step 3 concerns P-rich liquid and solid waste e.g. black water and sludge that can be recycled into new products (also after extracting biogas). The conventional solid waste hierarchy comprises two more steps: incineration and landfilling. These are not included here, since the EU parliament has already decided that only waste that cannot be reused or recycled should be incinerated, and landfilling is only for residues after exhausting the previous steps.

Two examples of managing P from households (Chapter 3.1) and P on a global scale are presented using data from various studies (Chapter 3.2).

3. Results and discussion

The OECD [2] estimates that at the end of this century 1.5 billion people will reside in rural areas, providing food for the 8.5 billion living in cities. At the same time, cities and infrastructure encroach on prime farmland, acreage for fuel production increases, the nutrient-rich organic wastes from cities is mostly wasted and not recovered and recycled as fertilizer. The ongoing global shift towards more meat in diets puts further pressure on food production, because more land, nutrients and water are required per kilogram of meat protein than vegetable protein [3]. A business as usual scenario shows that current trajectories of food production and consumption are not sustainable, and the following outlines ways to enhance nutrient management for food security [1].

3.1 Bending linear nutrient flows from households – systems perspective

Cities are fast becoming “nutrient hotspots” in two senses – first as centres of demand for Nitrogen (N), Phosphorous and Potassium (K) in the form of food to be consumed, and second as location of large amounts of these nutrients in excreta and food waste. For example, urine is the largest single source of N, P and K emerging from cities [4]. However, most nutrients in urban waste and wastewater are not recovered and reused [5]. For the most part, solid organic waste is collected and transported to landfill sites, where the nutrients will often remain for years, unless they leach into groundwater or are emitted into the atmosphere (N). Toilet waste (excreta) from urban households, in the best of cases, ends up as sludge in a wastewater treatment plant. However, sludge is also often sent to landfills and to some extent incinerated due to its perceived or real toxicity. Such data are used in Figure 1 to depict common conditions in today´s urban areas.

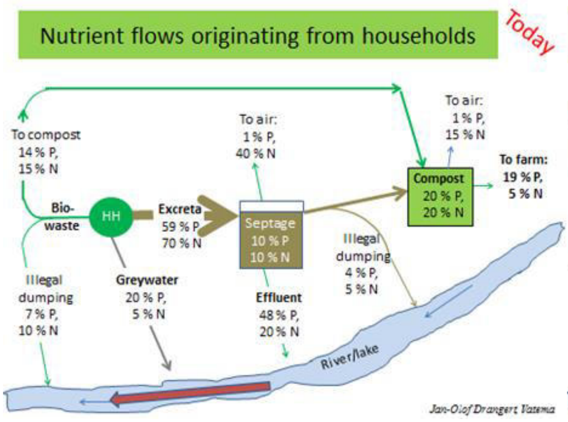


Figure 1: Nutrient flows from households (HH) today.

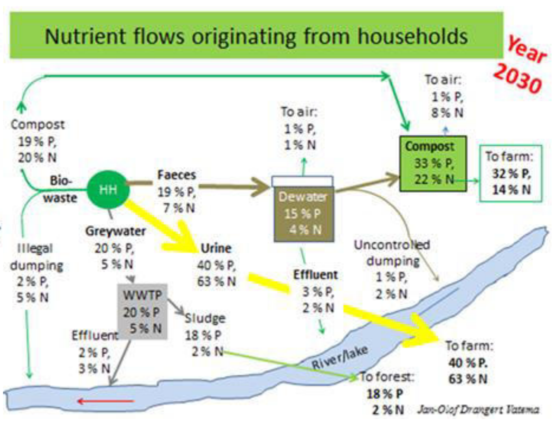


Figure 2: Scenario for nutrient household flows 2030.

The term “bio-waste” refers to such items as food waste, and paper and garden waste. It is usually easier to manage solid organic waste than liquid organic waste, which gets caught in sludge that remains after wastewater treatment. For example, food remains, fat and grease from plates, pans and cutlery are easily swept into the organic waste bin, which makes it possible to use it for biogas production and/or to use the compost as soil conditioner in agriculture. Besides, the alternative to wash away fat, oil and grease with water, clogs sewer pipes, and costly repair is often necessary.

Worldwide, the nutrient-rich excreta are commonly flushed to a septic tank for partial treatment. Part of the nutrients remains in the effluent, while part of the sludge is collected and ideally brought to a compost facility. However, illegal dumping is commonplace in developing cities. The theoretical flows in Figure 1 indicate a modest one-fifth of the P that households discharge is gainfully used, and only 5 % of the N.

Nutrients in liquid waste are largely discharged to water bodies (red arrow in Fig. 1). Organic matter in sewers causes eutrophication and algal blooms in receiving water bodies. This may result in less aquatic flora and even dead zones on lake floors and reduced living space for fish.

Considerable improvements of the capacity to reuse and recycle can be achieved by modifying sanitation systems (see Figure 2). Here, a hypothetical scenario for a typical city in the developing world has taken some steps to make its sanitation system more sustainable. Residents separate household solid organic waste, and the waste company composts it, and this added value reduces illegal dumping. Also, urine-diverting toilets have been installed, which collect urine separately, while dewatered faecal matter is stored in line with World Health Organisation recommendations before being applied to soil [6]. The wastewater treatment plant has been improved to remove 90% of the P, but, the same effect could have been achieved by prohibiting phosphate-based detergents, as the European Union did in 2013.

The nitrogen-deficient greywater sludge contains polluting substances that may accumulate in soil. Therefore this sludge is only applied at tree plantations, after a treatment that removes the available organics to avoid clogging of soil pores. The urine can safely be applied on agricultural soil [6], and it represents the least polluted fertiliser available on the market, and has a well-balanced nutrient composition [4]. The nutrient loss from well-managed urine is insignificant [7]. Likewise, the organic compost is likely to be of good quality and possible to apply on soil for food production. The short nutrient loop when using urine and composted organic matter in the garden is sustainable, whereas sludge from treated mixed wastewater is more risky and difficult to monitor.

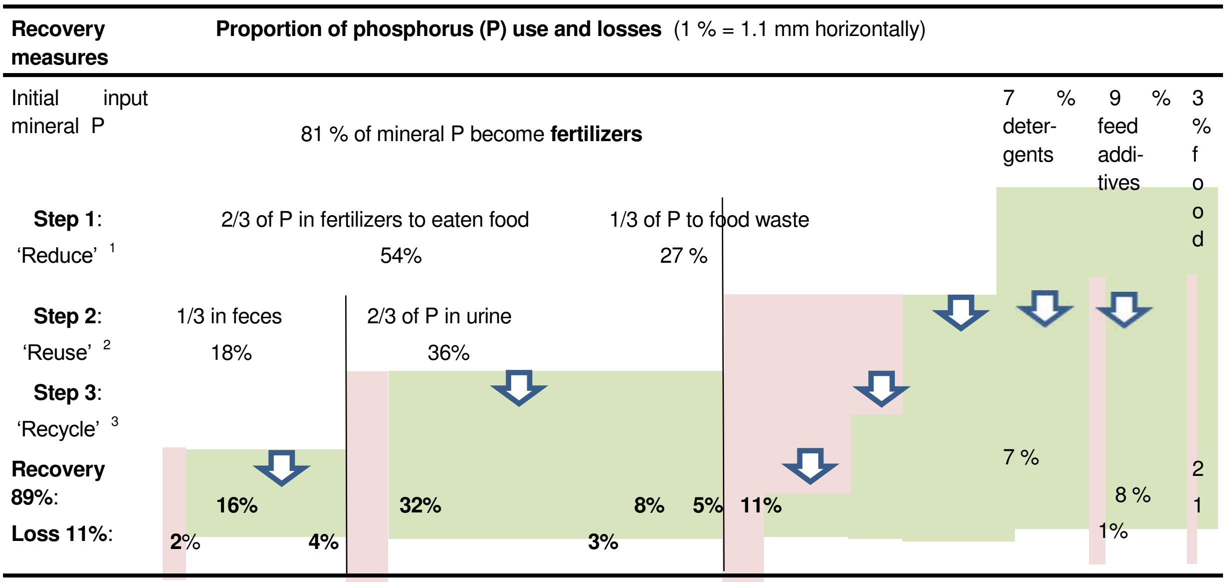
With these few changes of the system, productive use of the P originating from households increases from 19 % to 90 %, while N increases from 6 % to 79 %. This drastic reduction in wastage also means that water bodies are saved from nutrient pollution and eutrophication. It is equally important to note that these recycled nutrients can replace a significant part of the purchased mined chemical fertilizers, as indicated below.

3.2 Ways to recover nutrients and reduce mining of P

The global potential to utilize recovered P from waste streams is considerable. Figure 3 shows the fate of P through the three first steps of the 'extended waste hierarchy'. The mined phosphate rock is mainly used to manufacture fertilizers (81%), feed additives (9%), detergents (7%) and food additives (3%) [1], [8]. Reducing use of P (Step 1a) is possible for additives (10%) and detergents (7%). However, no shift back to more vegetarian diets is suggested, although it could reduce P demand considerably. The status attached to meat diets is strong and likely to be difficult to change.

A reduction of mineral P fertilizer is a matter for the agricultural sector and not part of efforts to recover urban nutrients. We assume initially that there are no losses of P in agriculture.

By not mixing various flows one may create a few flows that are not contaminated by our chemical society (Step 1b). Just like polluting industries today have to collect and contain their sewage and treat it separately, households could also dispose of its polluted greywater (often containing more varied chemical composition than industrial wastewater) in a separate sewer and recover and recycle nutrients from a fairly clean blackwater or urine and faecal matter.



[1] Food waste reduced to 20%, detergents down to nil, use of additives reduced to 2%

[2] Reuse 90% of all urine, reuse 30% of bio-waste.

[3] Recycle 90% of fecal matter and 70% of bio-waste (compost/biogas)

Figure 3: Potential to recover phosphorus (P) in the food chain, human excreta, and detergents with the help of the extended waste hierarchy. Green areas = recovered P and red areas = losses of P.

P in fertilizers ends up in food one-third of which (27% of all mineral P) is estimated to be wasted before eaten [9]. This waste is the only suggested item to be reused in Step 1. A realistic, say, 40% reduction (11% of all mineral P) is possible to achieve by simple measures in households, shops and storage facilities.

The P in eaten food is excreted as urine (2/3) and faeces (1/3), and urine can be captured and reused in Step 2 (32% of all mineral P) with only small losses of nutrients. Also, some (5% of all mineral P) of the remaining wasted food can be reused as animal feed and compost material.

With a well-designed city infrastructure it is realistic to recover 90% of P in urine and faecal matter (or from blackwater). Thus, 16% of all mineral P in faecal matter can be recycled in Step 3. Some 70% of the remaining food waste can be recycled (8% of all mineral P). Anaerobic digestion and composting allow the nutrient-rich digestate and compost to be used as a fertilizer in agriculture.

The above measures increase P recovery from a few percentage points to 89%, and losses are limited to 11%. The recovered P is enough to replace all mined P used for fertilizer production. Recovered P together with soil P and manure is enough to secure food supply.

However, actual P losses in agriculture would lead to a deficit of P. If the loss from mine to harvest is 50%, the recovery will go down to half i.e. 36% plus 17% from reduced detergents and food additives, in total 53%. The gap of 28% (81% minus 53%) has to be compensated by mined P. Other values of losses in agriculture leads to a proportional gap.

4. Conclusion and outlook

The two presented methods linking urban and rural areas by returning used P back to agriculture are complementary and indicate a great potential to save on mined P. Thus, extraction of nutrient resources is deferred by hundreds of years, and so is a transgression of planetary resources boundaries.

The three hierarchical steps applied to non-agricultural waste are likely to increase the recovery of P from a few percentage points to some 90 %, all of which are plant-available. This, in turn, would save all mined P had there been no losses in agricultural sector. Even if these losses are 50%, the mining of P can be reduced by two-thirds. Food security is ensured for the predicted 50% population increase during the present century, and in addition sustainable levels of harmful emissions from food production, consumption and nutrient waste management are achieved.

Now is a window of unprecedented opportunity to design urban infrastructure, since houses and infrastructure for an additional 5 billion new urban residents in the 21st century have not yet been planned [2]. Therefore, necessary changes in infrastructure to achieve the above will require no extra resources, just early planning and gradual changes of old houses when these are renovated anyway. A win-win situation is imminent, providing both food security and reduced harmful emissions to air, water and soil.

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References

[1] Cordell, D., Drangert, J-O. and White, S. (2009): The Story of Phosphorus: Food security and food for thought. Global Environmental Change 2009a; 19: 292-305.

[2] OECD (2013): Green Growth in Cities. OECD Green Growth Studies, OECD Publishing. http://dx.doi.org/10.1787/9789264195325-en.

[3] Prud'homme, M. (2011): The Global Supply of Nutrients. European Commission Workshop: NPK -will there be enough plant nutrients to feed a world of 9 billion in the year 2050? Brussels 5-6 December 2011. Available from: http://eusoils.jrc.ec.europa.eu/projects/NPK/.

[4] Spångberg, J., Tidåker, P., and Jönsson, H. (2014): Environmental impact of recycling nutrients in human excreta to agriculture compared with enhanced wastewater treatment. Science of the Total Environment. 493 (2014) 209-219.

[5] EC 2008: Waste Framework Directive 2008/98/EC;. Available from: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:32008L0098:EN:NOT

[6] WHO (2006): WHO Guidelines for the Safe Use of Wastewater, Excreta and Greywater. Volume 4: Excreta and Greywater Use in Agriculture. World Health Organisation, Geneva.

[7] Vinnerås, B., Palmquist, H., Balmer, P. and Jönsson, H. (2006): The composition of household wastewater and biodegradable solid waste– proposal for new norms for the flow of nutrients and heavy metals. Urban Water 3:1, 3-11.

[8] Winger, R. J., Uribarrid, J., Lloyd, L. 2012. Phosphorus containing food additives: An insidious danger for people with chronic kidney disease. Trends in Food Science & Technology 2012; 24: 92-102.

[9] Gustavsson, J., Cederberg, C., and U. Sonesson (2011): Global food losses and food waste - Extent, causes and prevention. A study commissioned by FAO for the International Congress SAVE FOOD at Interpack 2011. Dusseldorf. Germany; 2011.

TE-O_13 Encouraging local organic cycles in urban Europe with a collaborative tool

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1. Objectives

Based on the observation that around 70% of the EU's citizens have their home in urban areas [1] and that their associated organic material flows lead to resource depletion and environmental pollution, a concept is provided that shall support the implementation of sustainable urban cycles based on more experimentation and better communication between various stakeholders.

This work is based on the assumption that a lot of technological and organisational solutions for organic cycles within cities and between cities and the land are just waiting to be implemented. For example, topics like urban farming and green infrastructure were already discussed by Leberecht Migge in the 1920s and before [2]. However, a lot of these solutions require the active participation of the citizens and some flexibility in legislation. In contrast, today's recycling efforts for organic waste in cities focus mainly on central solutions and unified legislation.

The idea is to bring pioneers of local urban cycles and interested citizens together with urban administrators and planers, as well as with scientists and regulators in order to start fruitful discussions or to enhance them. To support this, the concept of a transdisciplinary toolbox was developed and partly implemented. The novelty of this toolbox lies neither in its content nor in the applied methods, but in their holistic integration.

2. Methodology

The toolbox is based on Mediawiki, the software that runs Wikipedia, as well as on Semantic Mediawiki (SMW) [3] and some other extensions, which provide databank capabilities. These allow the user-friendly input and analysis of structured data. In addition, data can easily be extracted for the use with other software. The strength of the toolbox lies in the linkage between the tools. Existing data from EU and national statistic offices can be combined with single research results and citizen observations in a meaningful way. While the current framework was created with an engineering viewpoint, the further development of the adaptive toolbox is based on the needs of its future users.

While the flexible management of the content includes several more aspects, the following three elements, which are also visualised in Figure 1, provide the basic functionality for the toolbox.

- Pages contain wiki-text only, although formats and attributes are described with a simple markup language. This translates the created content, including linked graphics into the output page seen by any visitor.
- Templates are pages which can be linked from any other page, including from other templates. They allow the multiple re-use of specific information, formats or structures in pages, without re-writing them again and again. A simple example is the structure of an info-box. The toolbox relies heavily on templates to structure and combine data.
- Forms allow structured data entries and updates. Each field of a form combines the entered data with a specific property. A form also links the page or parts of it to a specific template which turns the entered data into processable information.

In contrast to traditional databases SMW offers a high transparency because, like in Wikipedia, each edit is easy to trace and if necessary to undo. Therefore, a rigorous user management is not necessary. The toolbox is based on a very flat hierarchy with only three types of users:

- Unregistered visitors have access to all information and can potentially export data from the wiki, yet unlike in Wikipedia they cannot add or change any data, nor can they register automatically to become a full user.
- Registered (full) users can add and change data and information on pages which are editable by forms. Exceptions are some restricted fields in certain forms, which can only be edited by

users with administrator rights. The same applies to non-form pages which are protected. While users can move (rename) pages, they cannot delete them. Therefore any page move leaves a re-direct from the old page behind, which also makes it easier to undo bad actions.

- Administrators can edit and delete all pages, including templates and forms. They are also responsible to register and eventually to block users if they do not adhere to the toolbox rules of conduct. Administrators can be experts in certain fields who evaluate user generated solutions or scenarios, but they can also be wiki developers who implement new features or improve existing ones.

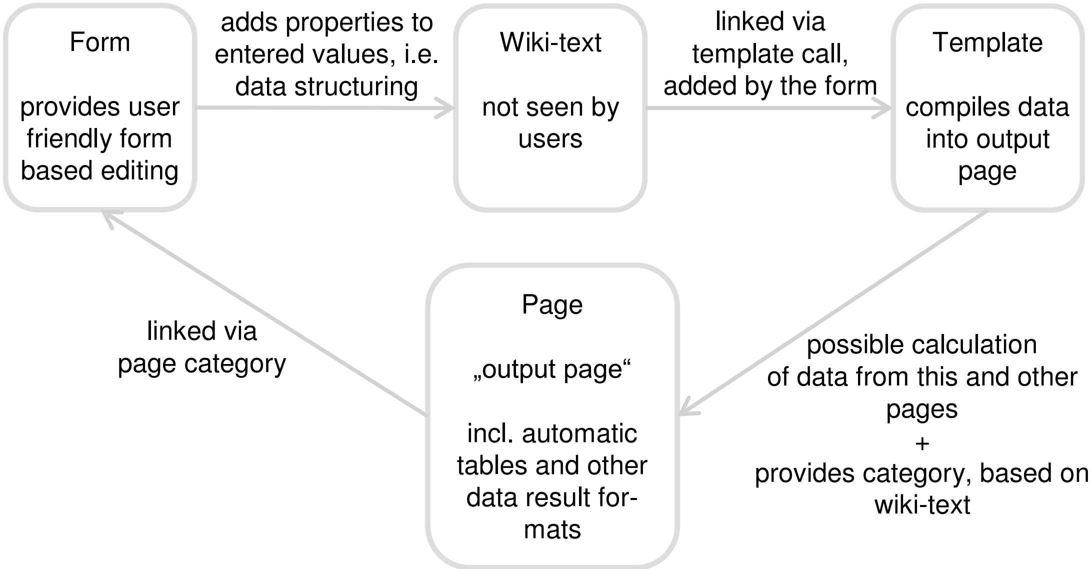

Figure 1: Basic operating principles of Semantic Mediawiki.

3. Results and discussion

The partly implemented toolbox is a wiki with databank capabilities based on SMW, which provides registered users with forms for structured data input and update. This allows the supplement of data not available from statistics offices and other central data providers. The geographical structure of the tool is based on standards used by Eurostat (NUTS). Basic geo-information capabilities allow quick visual estimates of biomass potentials and their use for specific urban areas or whole regions. Local specifics, like city ordinances or implemented solutions can be entered and retrieved by local stakeholders or used for the networking between regions. Beside the information use at the local and regional level, policy advisors and scientific research could profit from the collated data as well.

The toolbox contains currently six tools in different expansion stages. The tools are based on existing methods to structure and analyse data. Unlike in Wikipedia, any visitor, human or machine, can view and extract the collected data. Data is mostly entered via forms, which provide a user-friendly interface and avoid unstructured data collections. Relevant statistical data is usually imported by administrators, which can use the built-in import functionality for mass data.

Analyses of data are provided via pre-defined queries, which can output the results in various formats, like tables, diagrams, or even maps. In addition, also specific export formats are available, like csv-tables or xml-files. For specific interests not yet serviced by an existing query, any visitor can use the semantic drill-down or the search function, which covers the whole toolbox.

The main relations between the single tools are illustrated in Figure 2. The following subsections provide more insight into their details. As the development of the toolbox progresses, certain functionalities will change. Some ideas will become obsolete because they were not relevant for any target group. Other ideas may lead to new implementations. This applies not only for specifics within a tool, but also for the tools themselves.

Organics: This is mainly a database of organic materials which includes mirrored data from statistic offices and results from literature. Each material got its own page with links to related regulations and official classification systems. In addition, research results can be included to create averages and variations for specific material properties. Like in all other tools, provided data has to be referenced, so that other users can check and if necessary improve the data quality.

Solutions: This tool is based on the pattern language concept by architect Christopher Alexander [4]. It is a way to break up complex tasks into manageable smaller solutions, which are hierarchically connected with each other. The required data for solutions include results from material and energy flow analyses. This includes precise definitions of spatial and temporal system boundaries [5]. Monetary information will not be implemented; because the current monetary systems in Europe are inherently instable [6] and therefore not appropriate to describe the costs of material, energy and labour. To account the latter, qualified working hours can be added.

Especially for organisational solutions qualitative descriptions play a vital part, while they also add important information to technical solutions. For all entries certain quality standards apply. All data, may they be numerical or descriptive, have to be referenced with help of the References tool.

Hard data can be entered with a high flexibility. Depending on the availability of data, a constant value, a range of values, a median, or an average can be entered. A fitting unit can be chosen via drop-down menu. The conversion of units is done internally by SMW, which allows the further use of the values in comparisons of solutions and within scenarios.

Scenarios: With this tool several solutions can be combined to create different scenarios. The inputs and outputs of the chosen solutions are added up and are combined with given organic material flows based on the selected system (municipality or individual). This tool provides the most flexible choices and requires a skilled user to produce a meaningful scenario. While it is currently not possible to automatically generate graphical overviews from the scenarios data, users can add their own illustrations to make their scenarios more comprehensible.

Regulations: Beginning with international and EU-law, any regulation related to organic material is hierarchically structured down to municipal ordinances. Together with statistical data on material flows and geo-referenced solutions interesting regulations for potential organic material cycles could be identified, helping active citizens on site and legislators alike.

References: Unlike in Wikipedia, each reference such as a book, an article, or a presentation has its own entry. Such a reference can then be cited at multiple other locations within the toolbox. This shall increase the quality of citations and it allows seeing which reference is used where. The structure of the references is based on the time-proven BibTeX format [7].

Existing references can be retrieved with a dynamic table that lists the reference page and four of the roughly 24 BibTeX properties: type, title, keyword(s), and author(s). A real time search narrows the listed entries to the ones matching the entered string. In addition, there is also a built-in result format that can output all or specifically queried references in a single BibTeX file.

Each reference page offers four tabs. The first lists all reference properties and an abstract, if provided. The second contains quotations which can be cited on several pages. This functionality is similar to the reference itself. It avoids the repeated entry of the same quote and it shall increase the quality, i.e. the accuracy. The third tab provides the reference in different citation styles, including as BibTeX entry. The last tab lists the pages on which the reference is cited.

Geo-Portal: The geo-portal is based on two Eurostat-codes, NUTS for referencing countries and subdivisions for statistical purposes and LAU for smaller administrative units like districts and municipalities [8]. Each solution, scenario, organic material flow or regulation can be geo-referenced and would then show up on built-in map queries.

The pages for each administrative unit can be reached either via structured lists or in some cases via a map interface. The map interface is based on the Mediawiki Map Extension and Google Maps with page-embedded overlays, which are linked to the respective administrative units. The necessary administrative boundaries were retrieved from Eurostat and converted for the toolbox.

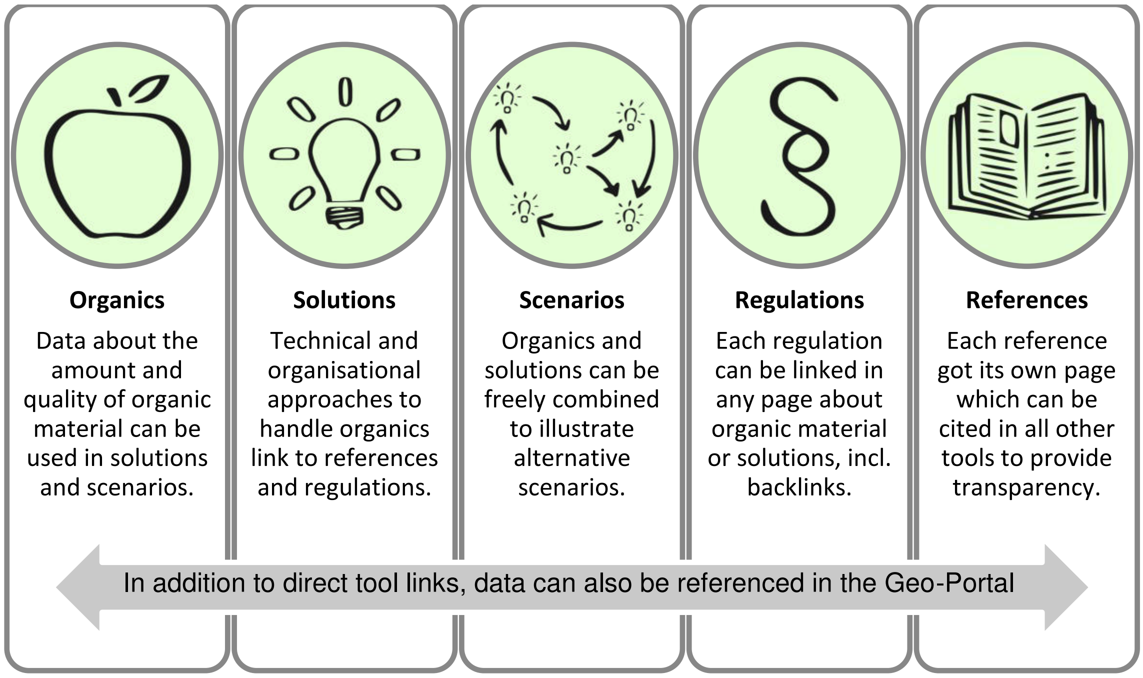

Figure 2: Interaction between the tools of the toolbox.

4. Conclusion and outlook

The developed concept of a transdisciplinary toolbox for urban organic cycles in Europe integrates several methods to cope with the complexity of the topic. The partial implementation of the concept, which can be found at www.localorganic.eu is highly flexible and can potentially be adapted to any needs of the intended user groups.

A collaborative toolbox with Semantic Mediawiki (SMW) is nothing new. A working example is the 2009 established Open Energy Information (http://en.openei.org) backed by the US Department of Energy. Yet, a toolbox with SMW can potentially much more than record, store, and retrieve non-linked data collections. It should be possible to create the outlined toolbox that represents the complex relations between a multitude of alternative solutions and scenarios for handling organic material in European cities. Hopefully, this could support a widespread bottom-up movement of experimentation and innovation to turn organic waste flows into local cycles.

Acknowledgements

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References

[1] EUROSTAT (2014): Regional Yearbook 2014. Statistical Office of the European Union, ISBN 978-9279389061. URL dx.doi.org/10.2785/54659
[2] Migge, L. (1926): Deutsche Binnen-Kolonisation. Dt. Kommunal-Verlag, Berlin-Friedenau, URL http://goobipr2.uni-weimar.de/viewer/resolver?urn=urn:nbn:de:gbv:wim2-g-2761890
[3] Koren, Y. (2012): Working with MediaWiki. WikiWorks Press, CreateSpace, ISBN 978-0615720302, URL http://workingwithmediawiki.com/
[4] Alexander, C.; Ishikawa, S.; Silverstein, M. (1978): A pattern language: towns, buildings, construction. Oxford University Press, New York, 1978. ISBN 978-0195019193.
[5] Brunner, P.H.; Rechberger, H. (2004): Practical Handbook of Material Flow Analysis. CRC Press LLC, ISBN 1-5667-0604-1
[6] Lietaer, B.; Arnsperger, C.; Goerner, S.; Brunnhuber, S. (2012): Money and Sustainability – The Missing Link, A report from The Club of Rome – EU Chapter. Triarchy Press, Axminster UK, ISBN 978-1-908009-7-77
[7] Fenn, J. (2006): Managing citations and your bibliography with bibtex. The PracTEX Journal,(4), URL http://tutex.tug.org/pracjourn/2006-4/fenn/fenn.pdf
[8] Schäfer, G. (2013): Europa vergleichbar machen− der Weg zu einer kleinräumigen harmonisierten Regionalstatistik. 289-295, In: Flächennutzungsmonitoring V, IÖR Schriften (61), Rhombos Verlag, ISBN 978-3-944101-18-7

TE-O_14 Software tool for a global evaluation of different manure management systems focused on specifics scenarios – MANEV TOOL

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1. Objectives

MANEV is a free software tool aimed at the supporting for the election of a manure management system that better fits into every agricultural scenario according to the main features involved: environment, economy, energy, agronomy, social impact, sanitary guarantees and legal framework. The environmental protection and the sustainability of the agricultural sector enhancing the use of treatment technologies in areas with high livestock concentration are the main targets of this work.

2. Background

There is a wide variety of manure treatments available in the market aimed at different handling features. Each management strategy has its advantages and disadvantages when considering environmental, agronomic, technical, energetic, cost and labour issues [1]. Consequently, before investing money, it is of paramount importance to get a support tool that could assist stakeholders and farmers on identification, evaluation, and selection of the more suitable option of the manure management for a specific area and aim [2].

Integrated assessment tools should provide awareness in the multiple risks of manure management and guidance to the farmers into cost-effective practices through the use of efficient technologies. Decision support systems are necessary to fill the gap between the scientific knowledge, technological market and practices of the livestock breeding sector. Economic and market policies and regulations of manure management present boundary conditions which determine whether a given technology is attractive to the farmer or not [3].

3. Methodology

The MANEV tool was developed within the framework of the LIFE+ MANEV project. The technical knowledge and experience of the project partners, coming from 8 different European regions with a strong livestock industry in 4 different countries, was crucial in the definition of the goal and scope of the tool and in the design of the simulation algorithms.

There are 25 different technologies included in the software based on separation techniques, energy production, valuable end-products, nutrient recovery and/or removal and tertiary treatments for pre-treated effluents. Additionally, more than 80 different predefined management systems have been designed combining these technologies and covering a wide range of management strategy approaches, technological complexity levels and investment and operating costs.

In order to draw the tool to the wide range of potential users in the agricultural sector, two different types of use were established: a help assistant and an advanced mode.

The programming language used was C# in a .NET environment. Comparable results are obtained in all Europe using for the calculations European databases and information from official organisms and public administrations classified by country, as well as scientific papers.

The MANEV tool was validated with the data obtained from the monitoring of 13 large scale treatment plants across Europe. A specific common protocol was designed to obtain comparable results in the evaluations using the same criteria, indicators and parameters included in the tool.

4. Results and discussion

4.1. General guidelines of use

The MANEV tool is available on the web page www.lifemanev.eu in English, Spanish, Italian, Polish, French and German (figure 1).

Figure 1: Home of the MANEV tool

Each user has a personal account with a database created with the information of his farms, plots and location. All these units can be saved and edited and are the starting point for the creation of projects in which different technological solutions are assessed in specific scenarios previously defined (figure 2).

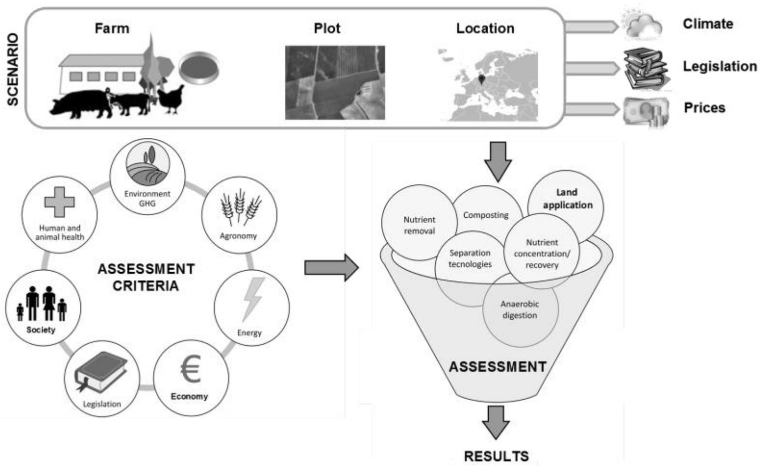

Figure 2: Evaluation scheme of a project.

The MANEV tool is targeted at the main stakeholders involved in the manure management: farmers, technicians, public administrations and private companies of the agricultural sector.
The software offers two working options:

Guided user: There is a **help assistant** aimed at stakeholders with less technical knowledge in treatment technologies. This functionality orientates the user towards the management systems that better fit into his scenario and requirements. To address this target, the user should answer a questionnaire in which he will choose the management strategy, the technological complexity and the economic burdens he can afford. Finally, bearing in mind these features, the tool will show a list of predefined and closed management systems that fulfil with the requirements. The land spreading will be the key strategy and the recommended practice in those cases where possible.

Advanced user: A user with a deeper technical knowledge in treatment technologies will be able to combine different technologies and build up a specific management system. Only the value of some parameters in the input composition, depending on the treatment, will restrict the combinations. These limits can be modified by the user. Additionally, the advanced user, based on his expertise, is able to change the value of some of the operational parameters involved in the calculations.
In this case no management strategy is prioritized and it can be evaluated a whole management system or just a treatment unit standing alone (table 1).

Table 1: Treatment units available in the software tool.

Treatment units				
Mixture	Separation Centrifugation	Separation Chemical settling	Composting Intensive windrow	Stripping
Storage	Separation Chemical centrifuga-tion	Separation Pressing	Composting Passive windrow	Filtration + Reverse Osmosis
Acidification	Separation Screening	Separation Chemical pressing	Composting Static pile	Land spreading
Transport	Separation Chemical screening	NDN	Evaporation	Exportation
Anaerobic digestion	Separation Settling	SBR	Thermal drying	Phytodepuration

In addition, more than one project can be compared bearing in mind: environment, economy and energy balances, agronomy and social concerns and sanitary and legal protection. The result of this comparative assessment will support the user deciding which option could better address the local necessities.

4.2. Challenges

The development of this tool has addressed the following challenges:

- Developing a decision supporting tool based on the use of treatment technologies which is an ever-changing environment. Therefore, the software has to be modular, flexible, easily updated and progress in parallel to the development and appearance of new technologies in the market.
  It is crucial to establish an on-going process of improvement and a continuous supervision of the tool use in order to identify its weaknesses and comply with the user demands.
- The obtaining of sound and efficient results locally within a European framework. The assessment requires trade-offs and the key is balancing accuracy and relevance of the results obtained. Therefore, the data gathered in the final report generated after the evaluation of every management system should be well-interpreted. The MANEV tool is meant for guidance and leads the user to the management option that better meets his necessities in comparison to other options, but absolute values should be cautiously understood. Some of the calculations, especially those related to economic data are strongly influenced by several external and local circumstances that cannot be gathered at European level. This constraint is due to his wide geographical approach.
- The on-line tool available through the project website makes easier the access for the users and speed up its dissemination but also makes crucial the agility of the calculations.
- The obtaining of external data directly from open databases on line allows the use of data updated instantly but may multiply exponentially the number of processing steps to carry out every calculation. The programming work must be carefully planned in order to save as many unnecessary steps as possible in the calculations delaying the processing time.

- The environment of the software should be friendly and easy to use. A huge load of data is required to the user, therefore the program must be clear and have an attractive and simple design in order to prevent the lost of interest of the users. Therefore, practical functionalities such as geo-location using maps and default values are provided to the user to speed up the processing.

5. Conclusion and outlook

MANEV is a sound and consistent web tool that unifies the knowledge and experience of the different treatment technologies and the criteria used for their evaluation focused on the environmental protection and the sustainability of the livestock sector in Europe.

The software fits into the user's expertise in the field of treatment technologies and management systems and responds to two basic needs in the choice of a management scheme: obtaining a general view of the suitable techniques under a specific scenario and the assessment from a global point of view of specific treatment process lines combining different technologies.

The MANEV showcases all the research work and experience acquired during last years in the use of different technological solution in all Europe and make easier the access to all this information to the final users. This work intends to provide a helping tool to all the stakeholders encompassing from policy makers to local farmers and lay the cornerstone for the foundation of a good practice learning showing the weaknesses and strengths of different management systems from a global point of view.

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References

[1] Fumagalli, M.; Acutis, M.; Mazzetto, F.; Vidotto,F.; Sali, G.; Bechini, L. (2012): A methodology for designing and evaluating alternative cropping systems: application on dairy and arable farms, Ecological Indicators 23, 189-201.

[2] Acutis, M.; Alfieri, L.; Giussani, A.; Provolo, G.; Di Guardo, A.; Colombini, S.; Bertoncini, G.; Castelnuovo, M.; Sali, G.; Moschini, M.; Sanna, M.; Perego, A.; Carozzi, M.; Ermido, M.; Fumagalli (2014): ValorE: An integrated and GIS-based decision support system for livestock manure management in the Lombardy region (northern Italy). Land Use Policy 41, 149-162.

[3] Petersen, S.O.; Sommer, S.G.; Bèline, F.; Burton, C.; Dach, J.; Dourmand, J.Y.; Leip, A.; Misselbrook, T.; Nicholson, F.; Poulsen, H.D.; Provolo, G.; Sorensen, P.; Vinneraas, B.; Weiske, A.; Bernal, M.P.; Böhm, R.; Juhász, C.; Mihelic, R. (2007): Recycling of livestock manure in a whole-farm perspective. Livestock Science 112, 180-191.

TE-O_15 Availability and use of urban areas green cuttings for methane production through anaerobic digestion

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1. Objectives

In recent years, as a consequence of the national financial crisis, several municipalities had to abate their management costs. Among these costs, significant budget reductions involves the conservation and maintaining of green spaces. According to Italian regulation green cuttings from urban areas is currently disposed of as a waste. Nevertheless, such biomass might be valorized as feedstock for anaerobic digestion plants. The objective of this study was to assess the yearly availability of green cuttings biomass in the Turin (Italy) Municipality and its suitability for biogas production as a way to increase Municipality's revenue.

2. Methodology

The green areas of Turin were quantified according to the data provided by the Municipality and classified into five categories: i) public gardens, ii) parks, iii) sport fields, iv) public buildings, and v) traffic islands. The amount of green biomass produced by each category was afterwards calculated according to the adopted management technique (e.g. fertilization, irrigation, cut frequency) and considering the information provided by the municipality. Samples of green cuts coming from a major urban park (surface area of about 80 ha, grass cutting 4-6 times per year between April and November according to climate conditions) were collected in September, just after 4th cutting, and analyzed for their biochemical methane potential (BMP). The BMP was measured at mesophilic (40°C) conditions according to VDI 4630 (2006) [1] methodology. In detail, 2 L batch reactors connected to tedlar gas bags (1, 3 L capacity) for biogas collection were used. Batch trials lasted 60 days. The biogas volume was collected daily into the gas bags and measured using a volume meter (Drum-type Gas Meter, Ritter). The methane concentration was measured by a Draeger XAM 7000 gas analyzer. Biogas and methane yields were normalized at 0°C and 1013 kPa. At the beginning of the test the biomass samples were analyzed for their dry matter (DM), volatile solids (VS), hemicellulose (H-CEL), cellulose (CEL), and lignin (ADL) content, according to standard procedures [2, 3]. Grass species present in the tested biomass samples were also identified and recorded.

3. Results and discussion

The City of Turin has approximately 2,100 ha of urban green areas (23 m^2 per habitant). About 80% of these surfaces are represented by public parks and gardens, whereas the remaining 20% by green areas belonging to public buildings, sport fields and traffic islands (Figure 1).

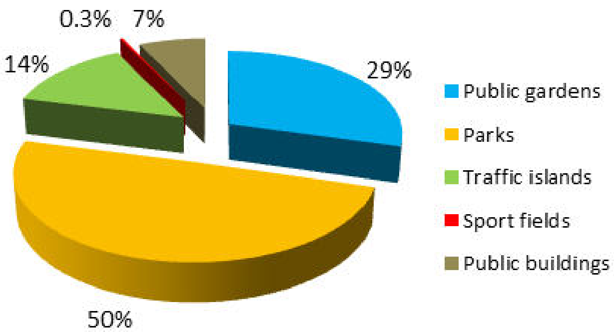

Figure 1: Average composition of Turin urban green areas.

Currently, about 1,600 ha of these surfaces (roughly equivalent to 80% of total urban green area) is being regularly cut. The number of grass cuttings per year are defined according to the use of urban green areas. In particular, parks and gardens grasses are cut on average 4-6 times per year (Figure 2). Naturalized areas (forests, river banks, etc.), however, are subjected to less maintenance. Only 5% (100 ha) of green spaces are cut 10-12 times a year or more. The latter are represented by public gardens, flower beds and portions of parks of particular interest. They are generally fertilized and irrigated. About 75% of green cuttings biomass is currently destined to landfill, whilst the remaining 25% is mixed with other municipal organic wastes (e.g., pruning residues) and used for compost production. According to the information provided by the municipality, the average disposal cost for such biomass is equal to approximately 30 €/t.

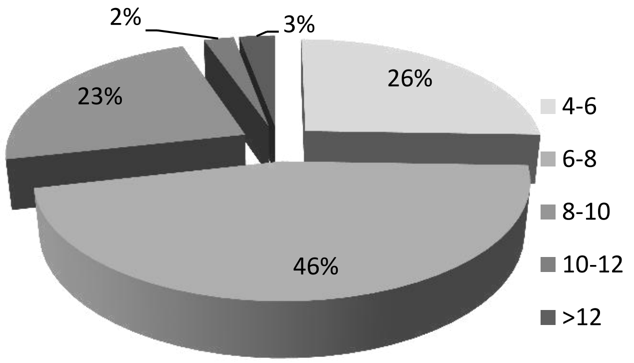

Figure 2. Frequency of grass cutting number per year in green urban areas in Turin

In areas where a lower number of cuts (about 4-6 per year) are practiced, biomass is commonly cut when it reaches the phenological stage of flowering or ripening, with vegetation height of about 20-30 cm. In the latter areas the calculated production of biomass was of 6 t DM per hectare and year (Table 1). In areas on which a greater number of cuttings occur (over 12 per year), grassland generally presents at the time of cutting a height of 3-5 cm. On these surfaces, the annual production of biomass resulted of about 8 t DM per hectare. Based on these data the potential biomass from the urban green areas of the city of Turin, that might be used for biogas production was estimated to be approximately 10,000 tDM per year (average 6.10t DM/ha) (Table 1).

Table 1: Evaluation of biomass production from grass cutting in city of Turin.

n° of cuttings/year	Surface (ha)	Biomass production (tDM/ha)	Total biomass production (tDM)
4-6	408	6	2450
6-8	735	6	4410
8-10	372	6	2232
10-12	39	8	312
>12	46	8	368
Total	1600		9770

In Table 2 are listed the characteristics of the biomass samples collected to estimates the potential biogas yields obtainable from grass cuttings of the urban green areas of the city of Turin.

Table 2: Species composition, average characteristics and specific methane yield (n=3) of the tested biomass samples

Species composition (%)	DM (%)	VS (%)	H-CEL (%DM)	CEL (%DM)	ADL (%DM)	Specific methane yield (m^3_N/tVS)
- Cynodon dactylon (35%) - Lolium perenne (25%) - Poa pratensis (15%) - Trifolium repens (10%) - Festuca rubra (10%) - Plantago major (5%)	22.0	20.2	22.5	29.0	3.5	240.1

The floristic composition of the samples mainly consisted of graminaceous species (Cynodon dactylon (L.) Pers., Lolium perenne L., Poa pratensis L., Festuca rubra L.), while only a small percentage (10%) were legumes (Trifolium repens L.).
According to the information provided by the municipality, the public gardens are generally seeded with graminaceous species because they are best suited to trampling and make the turf homogeneous. On average, the concentration of DM in the biomass samples resulted 22%, whereas the VS and DM ratio was higher than 0.90. Fibers (H-CEL, CEL and lignin) accounted for more than 50% of the DM. However lignin, which is not degradable under anaerobic conditions and may prevent microbial access to hemicelluloses and celluloses [4], resulted lower than 4% of the DM. Total methane yield recorded during the anaerobic digestion batch trial ranged from 221.1 and 266.8 m3N/tVS (average 240.1 m3N/tVS) (Table 2). Average methane concentration in biogas was 50%. These figures are comparable to those (155-298 m3N/tVS) reported by [5] for landscape management grass. A study [6] carried out in Denmark reported methane yields from roadside grass ranging from 220 and 390 m3N/tVS. However, biomass yield and energy potentials of grasslands is strongly affected by growing conditions, floristic composition, cutting height and frequency, level of irrigation and fertilization [7, 8]. In addition, the substrate-specific biogas production was found to be reduced with growing age of vegetation due to increasing contents of structural carbohydrates (such as lignin) that is not easily available for anaerobic degradation [5].

4. Conclusion and outlook

The study results suggests that grass clippings coming from the maintenance of urban parks could be collected and used to produce biogas instead of being disposed. In this way, the community reduces its landfill disposal costs (by approximately 1.6Mio € per year in the case of the city of Turin) and, at the same time, contributes to protect the environment by replacing fossil fuels with renewable energy. Specifically, according to the study results, by the anaerobic digestion of the whole biomass derived from the green urban areas of the city of Turin, it would be possible to produce approximately 2100x103 m3N of methane per year. This amount of biogas could be used to produce 9.0GWh of thermic energy and 8.5GWh of electric energy. Considering the current Italian incentives, by selling the whole amount of produced electric energy, the average revenue for the Municipality would be of approximately 1,6Mio € per year. This income would cover the 10% of the total annual cost for the municipal green areas management. Nevertheless, it must be pointed out that currently some operative limits to the use of grass clippings as feedstock have to be overcome. Their seasonal availability, concentrated in summer and spring, requires reliable storage techniques (e.g. silaging) to make grass clippings available also in autumn and winter conditions. Other biomasses from public green areas might be used as feedstock for A.D. plants: e.g. fallen leaves are known [9] to yield up to 240m3 CH4 t-1 VS and, thus, might contribute to widen the range of collectable biomasses from urban green areas. Nevertheless, these types of biomass can be contaminated by inorganic pollutants (e.g., heavy metals, plastics, cans, broken glass, metal fractions) that should be properly removed prior to anaerobic digestion. Further research is needed to evaluate i) the concentration of inorganic pollutants (with special regards to heavy metals) in the grass clippings and, ii) if a positive net energy production can be obtained when specific equipment are used to remove contaminants and/or undesirable coarse material.

References

[1] VDI 4630, (2006). Fermentation of organic materials, Characterisation of Substrate, Sampling, Collection of Material Data, Fermentation Tests, VDI Gesellschaft Energietechnik.
[2] AOAC, (2000). Official Methods of Analysis, 15th ed. Association of Official Analytical Chemists, Arlington, VA, USA.
[3] Van Soest P.J.; Robertson J.B.; Lewis B.A., (1991). Methods for dietary fiber, neutral-detergent fiber and non-starch polysaccharides in relation to animal nutrition. J. Dairy Sci. 74:3583–3597.
[4] Weiland, P., (2001). Grundlagen der Methangärung – Biologie und Substrate (Fundamentals of methane fermentation – biology and feedstocks). VDIBerichte Nr. 1620, S., pp. 19–32
[5] Prochnow A.; Heiermann M.; Drenckhan A.; Schelle H, (2005). Seasonal pattern of biomethanisation of grass from landscape management. Agricultural Engineering International: vol. VII.
[6] Meyer, A.K.P.; Ehimen E.A.; Holm-Nielsen J.B., (2014). Bioenergy production from roadside grass: A case study of the feasibility of using roadside grass for biogas production in Denmark. Resources, Conservation and Recycling Vol. 93, pp 124–133
[7] Ruzgas V.; Kadžiulis L., (1989). Terms and frequency of mowing cereal grasses in cases of different levels of fertilization with nitrogen. Žemdirbystė: mokslo darbai., vol. 37, p. 123– 134.
[8] Salette J.; Lemaire G., (1981). Sur la variation de la teneur en azote des graminées fourragères pendant leur croissance: Formulation d'une loi de diluition. Comptes Rendus de l'Académie des Sciences de Paris, série III 92: 875-878.
[9] Dubrovskis, V., Kotelenecs, V., Zabarovskis, E., Celms, A., Plume, I. (2012). Biogas production potential from agricultural biomass and organic residues in Latvia. Renewable Energy and Energy Efficiency. Biogas and production technologies.115-120.

TE-O_16 Addressing the nexus of sanitation and energy towards increased living conditions in rural areas in Kyrgyzstan

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1. Introduction

The population of the Kyrgyz Republic remains largely rural with 64 % of its population residing in rural areas. Rural inhabitants face many problems in daily life: lack of safe sanitation, WASH related diseases, energy poverty (lack of heating, light and fuel) and low nutritional status. The project Home Comfort, which gained support from the EU, has created local capacity for improved rural living standards through addressing the nexus of energy and sanitation with the innovative technologies: urine diverting dry toilets (UDDT), solar water heaters (SWH), energy efficient stoves (EES) in Issyk Kul and Naryn oblasts in Kyrgyzstan.
Alongside with capacity building and awareness raising, a number of objects demonstrating the innovative technologies have been implemented in demonstration centres, in resource centres, in vulnerable households (project funding) as well as replications in public places and households (co-funding or by own financial means), see figure 1.

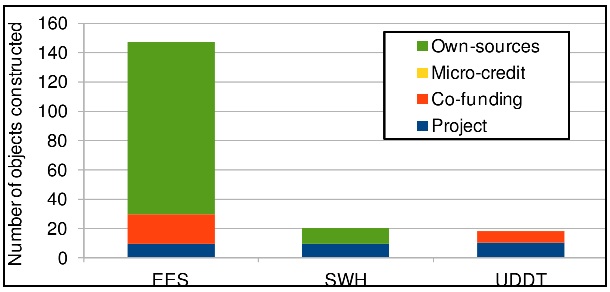

Figure 1: Number of technologies constructed during project duration of 2 years.

2. Methodology

The feasibility of the innovative “nexus” technologies was analysed based on a baseline and a final survey among 407 villagers (161 women and 246 men) of the target villages.
For this paper, additional sources of information were used, which included informal interviews, monitoring the results of objects constructed in the context of the project, and reviewing guest books at 8 demonstration centres in the project area. Data for the cost-benefit assessment were gathered from project managers, a couple of villagers and prices in local markets. Also results of previous projects in the area were used.

3. Description of the innovative “nexus” technologies

Energy efficient stoves (EES): burn fuel more efficiently (1/3 of fuel is needed than for a conventional stove) and produce less ash and less smoke than conventional ovens due to better combustion efficiency. The energy efficient stoves can heat a house with any kind of fuels, such as coal, dried dung and wood. The cost occurred to install a stove for cooking was in 2011 ca. €150, and €207 for a stove for cooking and heating.
Solar water heaters (SWH), also called solar collectors, use the energy from the sun for water heating and work without electricity supply. They provide hot water for: showers, kitchen use, laundry and home heating (depending on the size). Households using solar collectors have no extra fuel consumption for warm water heating and save money, which they would otherwise have spent on fuel. The price to install a standard SWH with a 200 l tank was €293 in 2011.
Ecosan or Urine diverting dry toilets (UDDT), also called ecosan toilet, can be implemented inside the house or attached to the house. The ecosan toilet does not need water for flushing, it does not smell, nor does it attract flies. Urine diverting toilets do not mix urine and faeces by using a separating toilet seat. Urine is collected and stored in a reservoir. Faeces, which are collected underneath the toilet, must be directly covered by dry materials such as sawdust, soil, ashes, or a mix-

ture of those. The toilet products, urine and faecal compost, can be used as organic fertilisers. The price for a standard UDDT is €428.

4. Results of the assessment of costs and benefits of the technologies

In this chapter, the financial costs and benefits of the technologies (UDDT, EES and SWH) at the household level are analysed and compared to the traditional technologies, which consist of simple pit latrines or ventilated improved pit latrines (VIP) for sanitation, conventional stoves for heating and cooking, public or private banya for warm water body washing.

Quantitative assessment

The quantitative assessment uses a time period of 10 years with an annual discount rate of 10%. Soft indicators like health, comfort and time saved are not included in the analysis because figures are very subjective, but they are nevertheless considered in section 5.2. The calculations are based on the assumption that the households have to replace their toilet, stove or bathing facility or that a new house is being constructed.
In the graph, it can be seen that a UDDT is cheaper than a VIP latrine after 4 years. Because the pit latrine has to be replaced, the cash saved on relocation by using UDDT accumulates over the years, while the fertilizer (urine and faecal compost) adds to its financial benefit.

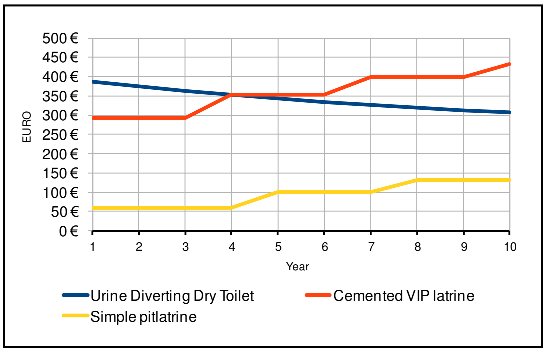

Figure 2: Cumulative and discounted costs of the sanitation technologies.

The EES has an investment costs of €207 in case of a standard stove for heating and cooking and €150 for a simple stove only for cooking. A conventional stove costs €256. The EES uses 40% less inputs in the form of coal, wood and dried cow dung (kyzyak) to operate. According to the survey, people spend an average of €300 on these combustibles using traditional stoves, resulting in an annual saving of €120. In our model, we assume that after 5 years, the EES needs to be repaired for €80.

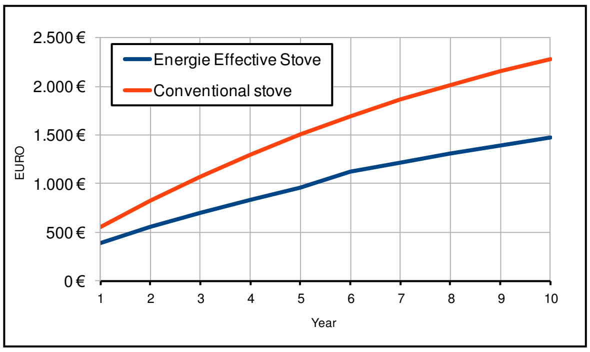

Figure 3: Cumulative and discounted costs of the stove technologies.

The SWH has an investment costs of €293. After 5 years, the SWH needs maintenance for an estimated cost of €77. Additionally, €177 is needed to build a shower. The SWH saves €36 in fuels and electricity costs from water heating (calculated according to the insulation in Kyrgyzstan, the efficiency of solar collector and local energy prices). The banya is an individual sauna which provides also hot water and is used for showering in the households in rural Kyrgyzstan. The construction of a new banya cost €565, and after 5 years, maintenance at the cost of €40 is

needed. The cost for running the banya is approximately €150 per year assuming that the householders use the banya every week. Some people choose to go to a public banya where they pay approximately €250 per year.
A banya is far more expensive than a shower. Over a period of 10 years using the latter instead of the former can allow households to save up to €1400. A solar water heater saves €273 over 10 years compared to an electrical boiler, which cuts the costs of personal hygiene by half.

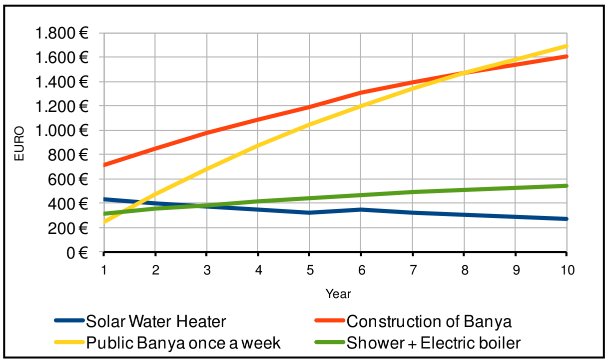

Figure 4: Cumulative and discounted costs of warm water solutions.

Qualitative assessment
The soft indicators play a crucial role for householders when it comes to deciding whether to invest or not. The technologies improve the living conditions in a way that rural families can enjoy a level of comfort similar to their urban counterparts. Especially women and girls benefit from the technologies in their daily life, see also table below.

Table 1: Qualitative Benefits and Disadvantages.

	Benefits	**Disadvantages**
EES	More comfort; time-saving; energy-saving; More rooms can be heated; Improved health, better air quality in the house; There are options to integrate house heating; It stays warm for a long time;	Must be built by a master to ensure the quality
UDDT	Increased comfort and hygiene: no bad smell clean, can be built inside; Improved health and nutrition; Relatively simple explanations; can be built and repaired with locally available materials; No water source required; UDDT does not require water; Multiple designs possible; can be used by children.	Adequate training is needed; Maintenance required (cleaning, emptying urine tank at least 4 times a year and the faecal chamber every second year).
SWH	Availability of hot water; Time-saving; Basic principles easy to understand; Flexible placement opportunities.	Must be built by a master to ensure the quality Lower efficiency in the winter; Problems with freezing prevention; Risk of construction mistakes.

5. Conclusions

The three technologies EES, SWH and UDDT are “nexus” technologies to improve the living conditions in villages of the oblasts: Issyk Kul and Naryn. They can contribute to improve health, create opportunities to save resource and increase the level of income in families. The technologies are much appreciated by the villagers, and more specifically by women, whose empowerment is a crucial component of development. The decision to buy an innovative technology for householders is perceived as a risk. For people living in poverty, the costs of the technologies are relatively high, and the investment often represents several months of income for the family. This study however proved that villagers who are challenged by current problems in their daily routine are ready to take risks to improve their comfort and security. This is confirmed by many self-financed replications of the technologies during the project.

Acknowledgement

The project was supported by the EU and implemented by Women of Europe for Common Future (WECF), Kyrgyz Alliance for Water and Sanitation (KAWS), CAMP Ala-Too (CAMP) and Rural women's association "Alga" during 2011-2013.

References

[1] Bakashova et al (2013) Gender-sensitive feasibility study for rural areas in Issyk Kul and Naryn http://www.wecf.eu/english/publications/2013/homecomforts-publication.php

[2] Jorritsma et al (2009) Introducing Sustainable Sanitation in Kyrgyzstan Case Study: An analysis of success factors and barriers http://www.wecf.eu/english/publications/2009/sanitation-kyrgyzstan.php

[3] UN (2014) Thematic Debate of the General Assembly "Water, Sanitation and Sustainable Energy In the Post-2015 Development Agenda" http://www.un.org/en/ga/president/68/settingthestage/1wsse.shtml

[4] Wendland et al (2012) Experiences with urine diverting dry toilets (UDDTs) for households, schools and kindergarten in Eastern Europe, the Caucasus and Central Asia (EECCA) http://www.wecf.eu/download/2011/February/SSP-06_Jan2011_16-221.pdf

TE-O_17 Resources recovery and economic aspects in the application of terra preta sanitation system in Arba Minch, Ethiopia

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1. Objectives

The overall objective of the study was to evaluate the application of terra preta sanitation [1, 2] system in a case study settlement in Arba Minch, Ethiopia. Terra preta sanitation is a recent development under resources-oriented sanitation approach and there was no fully developed community level system embracing the whole concept. The first specific objective of the study was to assess the potential of the sanitation system implementation on recovering nutrients and organic matter contained in human waste for reuse in agriculture. Secondly, the study aims at evaluating the potential of the sanitation system to provide low-cost sanitation alternative in the case study area compared to other conventional sanitation systems.

Terra preta sanitation (TPS) system was planned for the case study settlement based on results of experimental study [2], field data and information from literature. Based on the established system components and boundary conditions material flow analysis was conducted to evaluate the nutrient and organic matter flows through the different processes. Simplified economic evaluation was carried out based on lifecycle cost analysis of the sanitation system considering the estimation of capital investment, and operation and maintenance costs of the different sanitation system components as well as revenues from final treated sanitation products.

2. Case study description and methodology

The case study area is a settelement located in Arba Minch town, Ethiopia. 1250 households live in the settelement with average family size of five. The settelement is characterized by poor sanitation coverage with about 80% of the households using pit latrine as sanitation facility and 20% of the households without any kind of sanitation facility.
The planned TPS system consists of two pathways for treatment and processing human excreta collected in terra preta sanitation toilet. The first pathway applies vermicomposting [3] as a method for further treatment and stabilization of collected toilet waste in combination with other organic materials and biochar to produce nutrient and organic matter rich soil amending material which was considered to be utilized in local agriculture. This system is referred as terra preta sanitation (TPS) system. The second pathway applies anaerobic digestion with subsequent vermicomposting of digested slurry for further treatment and stabilization of collected toilet waste with other organic materials, and it is referred as terra preta sanitation with anaerobic digestion (TPSAD) system. The scheme for the application of TPS system is shown in the figure 1.

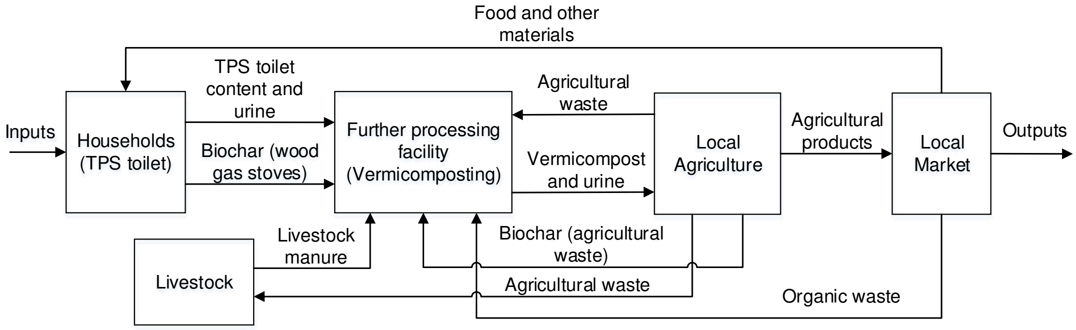

Figure 1: Scheme of TPS system application.

The components of the TPS system consist of:

- Facilities at the households which include TPS toilet which operates applying lactic acid fermentation process, urinal, urine collection container and wood gas cook stoves;
- Collection and transport system which include vacuum and dump trucks, mini-vacuum tankers, temporary storage tanks at transfer stations, and operating personnel;
- Further processing facility which include vermicomposting plant, lactic acid fermentation and urine storage tanks, temporary waste collection system, warehouse, site office and operating personnel; and
- Utilization of the final treated sanitation products in local agriculture.

TPSAD system is a modification of TPS system where the substrate collected in TPS toilets was considered to be digested in anaerobic digesters, combined with livestock manure and market organic waste, and the produced biogas energy is considered to be utilized by the local households. The digested slurry is further processed by applying vermicomposting, combined with biochar and other organic materials, to produce soil conditioner for agricultural application. The sanitation facility at the households and the scheme for collection and transport of filled toilet contents were considered similar to that of the TPS system.

The daily per capita production of faecal matter and urine at Arba Minch is 0.3 kg and 1.1 L, respectively. It was assumed that 80% of the urine and 100% of the faecal matter are collected at the households with 75% of the urine considered to be collected separately in urine collection container connected to urinals and the remaining urine is collected in TPS toilet mixed with faecal matter. The average household organic biowaste generation at Arba Minch is 600 g/d, and it was considered that all the organic biowaste is added to the TPS toilet to cover part of the sugar demand for lactic acid fermentation process. Addition of biochar during vermicomposting process is an integral part of the TPS system as biochar is considered to enhance the long-term effect of the produced soil amending material in terms of improving soil property for sustainable agriculture [2]. To meet this biochar demand wood gas cook stoves were planned to be used by the households and part of the agricultural waste from the local agriculture was considered to be carbonized.

Continuous vertical flow-through vermicomposting reactor was considered for further processing of the excreta collected in terra preta sanitation toilet. In vertical flow-through system organic material is fed to the reactor from the top in thin layers and the earthworms process the waste and move to the upper layer leaving the processed waste to pass through the base of the reactor. For the TPSAD system fixed dome anaerobic digester was considered due to its relatively lower construction cost and the good experience in Ethiopia for construction and operation.

Material flow analysis of the sanitation systems was conducted using STAN software. Economic evaluation was performed by determining the total annual cost of the sanitation systems which is the sum of the total annualized capital investment costs and the total annual operation and maintenance costs of the sanitation system components, and comparing it with annual revenues from the final sanitation products. The benefits considered for the cost evaluation include revenues in terms of monetary values of vermicompost, urine fertilizer and biogas energy. Initial capital investment costs were converted to annualized present values based on the life spans for the different components of the sanitation systems and the interest rate according to equation 1:

$$APV = I\frac{r.(r+1)^n}{(r+1)^n-1} \qquad \text{Eq. 1}$$

Where,
APV: annualized present value of investment cost [ETB y^{-1}], (ETB – Ethiopian Birr)
I: investment cost [ETB y^{-1}]
r: interest rate [-]
n: life span of different components of the sanitation systems [y]

5% interest rate was used for the calculations of APV based on the current value used by Commercial Bank of Ethiopia. The life span of the various components of the sanitation systems was assumed considering the material and the function of the component. Life span of 50 year was assumed for structural components made of concrete, 20 year for TPS toilet and 15 year for mechanical parts like vehicles and waste mixers [2].

3. Results and discussion

3.1 Material flow analysis

Organic matter and macro nutrients: nitrogen, phosphorous and potassium, recovery potential of the sanitation systems was evaluated taking into account transfer coefficients for transformations occurring through the different sanitation system components. The parameters used for the estimation of nutrient and organic matter flows are detailed in [2]. The annual quantities of nutrients and organic matter that can be potentially recovered by the application of TPS system in the case study area following the two pathways for further treatment and processing are shown in figure 2.

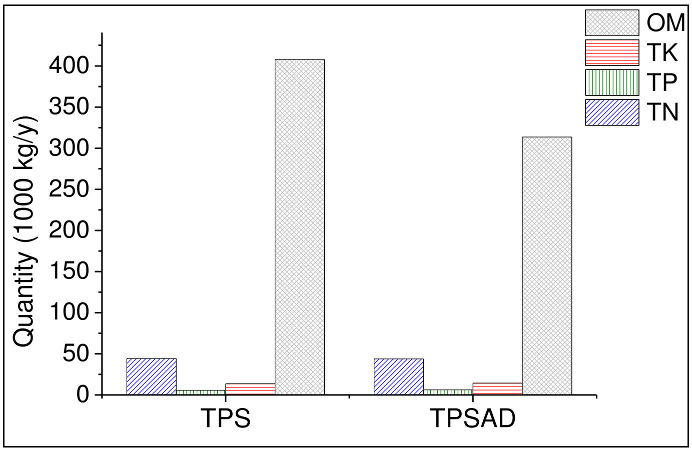

Figure 2: Annual nutrient and organic matter recovery potential of the sanitation systems. *OM – organic matter, TK – total potassium, TN – total nitrogen, TP – total phosphorous.*

Considerable potential was observed for recovering organic matter and nutrients by the implementation of the sanitation systems. 44.4 and 43.9 tons/y of nitrogen can be recovered for TPS and TPSAD systems, respectively. TPS system can recover 408 tons of organic matter which is more than that of the TPSAD which can recover 314 tons of organic matter. High efficiency of nitrogen and organic matter recovery is associated with the conservation effect of the lactic acid fermentation process [2] which was considered to be applied as human waste collection method in terra preta sanitation toilet and also the efficiency of vermicomposting process in conserving the nutrients available in the organic waste mixture. Lower quantity of organic matter from TPSAD system is due to the transformation of organic matter to biogas in anaerobic digesters.

Utilization of the final sanitation products in local agriculture: Considering the fertilizer application rate at the local state farm consisting of 820 hectare agricultural area and taking into account the nitrogen and phosphorus composition of the mineral fertilizers used, it was estimated that the nitrogen recovered can cover 100% of the fertilizer demand and the phosphorous recovered can cover 39% of the fertilizer demand of the state farm.

Utilization of biogas energy by the households: Assuming that a lamp consumes 0.11 to 0.15 m^3 of biogas per hour and operates 5 to 7 hours per day, the total biogas produced from anaerobic digesters will supply lighting to about 70% of the households using one lamp. Alternatively, considering per capita firewood and charcoal consumption of 120 kg/y at Arba Minch and the energy equivalence of 1 kg firewood and 1 kg charcoal to be 0.2 m^3 and 0.5 m^3 of biogas, respectively, the biogas produced can replace firewood and charcoal energy for the whole settlement area (assuming firewood and charcoal contribute to 75% and 25% of the household energy).

3.2 Economic Evaluation

Considering the capital investment as well as the operation and maintenance costs of the sanitation system components, the annual per capita cost of the sanitation systems was estimated to be 534 Ethiopian Birr (ETB) and 597 ETB for TPS and TPSAD systems, respectively. Compared to the cost of sewerage based system and septic tank system which have average per capita annual cost of 1,085 ETB and 769 ETB, respectively (based on data from the year 2004), the TPS systems offer low-cost alternative. Moreover, TPS systems produce high quality soil conditioner and fertilizer as well as biogas energy, therefore the revenue from the final sanitation products has to be included in the economic evaluation. In the study the revenues from the final sanitation products were estimated on the basis of market price of a similar product, i.e. price of organic compost, mineral fertilizer, electricity or firewood were used as a basis for assuming the value of vermicompost, urine fertilizer and biogas energy, respectively. Benefit to cost ratio was determined considering the values of the final sanitation products to evaluate the net annual cost of the sanitation systems which is shown in figure 3.

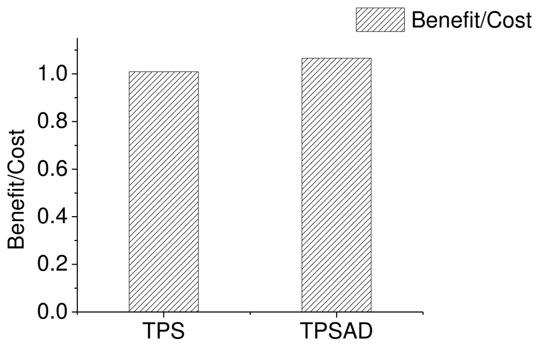

Figure 3: Benefit-cost ratio of the sanitation systems.

Both TPS and TPSAD systems generate net benefit as indicated by benefit to cost ratio of greater than 1. The revenue generated from biogas energy in the TPSAD system paid the cost of biogas digester system and created net benefit. It is further observed that the net cost of the sanitation systems is strongly affected by the assumed price of vermicompost.

4. Conclusion and outlook

The findings of the study suggest that the terra preta sanitation system enables the provision of economically self-sustaining sanitation service boosting local agricultural production through utilization of safely treated final sanitation products. Moreover, the integration of anaerobic digestion in terra preta sanitation as intermediate step between collection and further processing by vermicomposting was found to have overall economic advantage although the amount of organic material recovered for application of soil improvement was reduced by its conversion to biogas.

For an effective application of the terra preta sanitation system there must be parallel activities to create awareness in the society and to disseminate knowledge regarding benefits of recycling resources after appropriate treatment of human waste. Policies and regulations allowing safe recycling of resources from human waste should be promoted and activities for recycling should be supported by local authorities to develop market for the final sanitation products. Moreover, in order to avoid any health risk associated with recycling of nutrients in human excreta it is recommended to use the vermicompost first as fertilizer and soil conditioner for production of non-editable agricultural crops like cotton, moringa, fiber and biomass crops. The land can then be used for sustainable food production at later years.

Acknowledgements

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References

[1] Factura, H., Bettendorf, T., Buzie, C., Pieplow, H., Reckin, J. and Otterpohl, R. (2010), “Terra Preta Sanitation: rediscovered from an ancient Amazonian civilisation – integrating sanitation, bio-waste management and agriculture”, Water Science and Technology, Vol. 61, pp. 2673–2679.

[2] Yemaneh, A. (2015), “Evaluation of lactic acid fermentation process in terra preta sanitation system and application in Arba Minch, Ethiopia”, PhD Thesis, Institute of Wastewater Management and Water Protection, Hamburg University of Technology, Germany.

[3] Buzie-Fru, C.A. (2010), “Development of a Continuous Single Chamber Vermicomposting Toilet with Urine Diversion for On-site Application”, PhD Thesis, Institute of Wastewater Management and Water Protection, Hamburg University of Technology, Germany.

TE-O_18 Treatment of deinking sludge from wastepaper recycling by anaerobic digestion

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1. Objective

The use of recovered paper in paper production has increased steadily and has established itself as an important resource in many countries [1]. With 500 sheets of recycled paper about 79.4 Liter of water, 16.3 kWh of energy and 7.5kg of wood is saved when compared to paper produced from fresh pulp [2]. However, recycling operations produce large amounts of different wastewater streams among which is the deinking sludge (DS). According to Blanco et al. (2008), it represents 70% of waste produced by the European pulp and paper industry [3]. With the stringent European legislation on landfilling and the requirement of complementary fuel source for the incineration of such low calorific value sludges, the industry is faced with the challenge of finding a more ecological and economical approach of managing the produced DS. DS contains mainly water but also substantial amount of organic fiber fines as well as inorganic compounds. The objective of this study is therefore to investigate the application of DS as a substrate for the production biogas as an optimization approach for the waste paper recycling industries.

2. Methodology

2.1 Experimental set up

DSs from different sampling days, taken for a period of about 1.5 years, were provided from a waste paper recycling company in Northern Germany. DS was sampled after a predewatering step. DS was tested as a substrate for anaerobic digestion under mesophilic conditions in two different test systems.

- Firstly, a batch test comprising of 15 one-L Duran glass bottles immersed in a temperature controlled water bath was carried out following the guidelines of [4]. Biogas volumes were measured by displacement method using Eudiometers [5]. The investigated DS was sampled in April 2014 (sample number: S2W70_14.04.24). Three different Inocula collected from different operational anaerobic treatment plants were compared in the batch experiments. The first inoculum was taken from an anaerobic digestion plant that utilizes energy crops, manure and some commercial biowastes (Herling Group) and the others from conventional wastewater treatment plants (WWTP Köhlbrandhöft, WWTP Seevetal). They are described hereafter as Inoculum 1, Inoculum 2 and Inoculum 3 respectively. Cellulose was used as control substrate. The substrate to Inoculum organic dry mater (oDM) ratio was 0.5 for all the tests with DS and 0.13, 0.5 and 0.39 respectively for the tests with cellulose (control). The batch test did run for 22 days.
- Secondly, semi-continuous experiments using a test system comprising of three continuous stirred 10-L tank reactors (CSTR) made of polyvinyl chloride (PVC) were carried out. The reactors were immersed in a temperature controlled water bath. Stirrer velocity was set at about 70 rpm. The biogas volumes were continuously measured using millligascounters (MGC-1 PMMA, Dr.-Ing. Ritter Apparatebau GmbH & Co.KG). The gases were collected in aluminum bags for further analyses. The feeding of DS and removal of digestate from the CSTRs was done with a peristaltic pump. DSs from three different sampling days were step-wisely used (sample Numbers: S2W70_14.11.19; S2W70_14.18; S2W70_15.03.11). For the semi-continuous experiments inocula comprised digestates from preliminary semi-continuous tests (inoculum 4 and 5) and a digestate from a wastewater treatment plant (WWTP Seevetal; Inoculum 6). Since earlier investigation showed a deficiency in trace elements (TEs) compared to needed values given in [5], the influence of cobalt-Co, Molybdenum-Mo and Selenium-Se addition was investigated. A TE mixture containing Co-, Mo- and Se- solutions was prepared as follows: 82.5 vol. % Co-solution ($CoCl_2.6H_2O$), 9.7 vol. % Mo-solution ($(NH_4)_6Mo_7O_{24}.4H_{2O}O)$), 7.8 vol. % Se–solution (Na_2SeO_4).

- The semi-continuous experiments were started with three different inocula as follows: Reactor 1 with 79.5 vol.% inoculum 6 and 20.5 vol.% Inoculum 4; Reactor 2 with 79.5 vol. % inoculum 6 and 20.5 vol. % inoculum 5 ; Reactor 3 with 100 vol. % inoculum 6 to provide different types of microorganisms. The initial substrate to inoculum fresh mass ratio was 0.17 for Reactor 1 and 2 and 0.21 for Reactor 3. The experiments were operated in five phases depending on the subsequent feeding pattern as shown in Table 1. The feeding of the reactors was done two times in a week. For this article, only results from phases 4 and 5 were presented,

Table 1: Description of feeding method for the three reactors during the semi continuous experiments.

Phase	Reactor 1 & 3			Reactor 2		
	Number of feedings	Inoculum [ml]	DS [ml]	Inoculum [ml]	DS [ml]	TEs mixture [ml]
Phase1 (day 6-48)[1]	11	664	136	586	122	82
Phase 2 (day 52-87)[2]	11	1000	200	928	190	82
Phase 3 (day 90-104)[2]	5	0	200	0	118	82
Phase 4 (day 108-122)[2]	5	0	400	0	318	82
Phase 5 (day 125-139)[3]	6	0	600	0	518	82

[1] S2W70_14.11.19; [2] S2W70_14.12.18; [3] S2W70_15.03.11

2.2 Analysis

Characterization of DS samples was carried out by measuring following basic parameters Water content (WC), Organic dry matter content (oDM), Total organic carbon (TOC), Total inorganic Carbon (TIC), Total Nitrogen (TN) and pH following guidelines described in [6]. A Multi N/C 3000 device (Analytik Jena, Germany) was used for the TOC, TIC and TN analysis. Additionally, Acidity/ Alkalinity ratios (FOS/TAC) were measured for the monitoring of the stability of the process using a FOS/TAC device (FOS/TAC 2000, PRONOVA Analysentechnik GmbH & Co. KG). Samples were taken from the reactor before semi-continuous feeding. Normalized volumes (Temperature of 273K and Pressure of 1013 hpa) were computed according to [4]. Biogas qualities (CH_4, CO_2, O_2, H_2S and N_2) were measured using a portable gas analyzer (BIOGAS 5000, Geotech Environmental Equipment, Inc.) after sufficient biogas amounts were produced in the batch test while for the semi continuous test they were measured regularly before each feeding. Normalized methane yields were calculated using biogas volumes and methane concentrations in biogas.

3. Results and discussion

3.1 Characteristics of deinking sludge

Basic characteristic parameters of DS are shown in Table 2. The water contents of DSs (average: 88.4 %FM) were lower than that of some common high water content Anaerobic Digestion (AD) substrates such as sewage sludge (92-96 %FM), pig manure (94 %FM), cow manure(90 %FM) and blackwater from vacuum toilets (99.4 %FM) [7,8,9]. When the organic content of the DS (average: 30.3 %DM) is compared with the same substrates it can be observed that DS has lower organic matter content than sewage sludge (60-80 %DM), pig manure (80 %DM), cow manure (80 %FM) and blackwater from vacuum toilets (55.5 %DM) and [7,8,9].

The average oDM measured in this study for both non-dewatered DS and dewatered DS are 35.8 %DM and 30.3 %DM respectively (Table 2). The oDM of (non-dewatered) DSs reported by [10], is in the range of 33-64 %DM. [3] also reported 37 %DM, which is similar to the non-dewatered DSs investigated here. The dewatered DSs are slightly lower (Table 2), when compared with the non-dewatered DS. The range observed in [6] for DS in general can be explained presumably by different wastepaper grades and also different types of deinking processes adopted.

The TOC indicates sufficient amount of organics is contained in the substrate, which may be biodegraded in AD. The high TIC- can be attributed to the calcium carbonate content which is used in large amount as filler in paper making [11]. The ratio of the TOC to TN when computed from Table 2 is 73.8. This give the information that sludge is low in nitrogen and this could limit the bioactivity and hence process efficiency in AD. The pH value of about 7.5 of DS is in a neutral range and well suitable for AD. With these characteristics DS seems to be usable for wet AD, but biogas gains have to be investigated for a prove (Chapter 3.2).

Table 2: Characteristics of deinking sludge.

Basic parameter	Unit	Quantity	Number of Samples analyzed
Water Content (WC)	%FM	88.4±2.8	6
Organic dry matter content (oDM)	%DM	30.3±2.5 35.8±4.0*	6 4
Total Organic Carbon (TOC)	mg/l	7620±1034	3
Total inorganic Carbon (TIC)	mg/l	6852±812	3
Total Nitrogen (TN)	mg/l	196±47	3
pH	-	7.5±0.5	6

FM = Fresh Mass, DM = Dry Matter; *analysed from non-dewatered DS sample

It can be assumed that the analysed DSs are well representative for DS in general.

3.2 Anaerobic digestion in batch tests

In the batch experiments, the DS was tested with three different inocula (mixed cultures) from operational biogas plant. The methane yield curves as seen in both Fig. 1(a) and 1(b) showed a lag phase of about 2 days for all control (Cellulose) and DS reactors. All three reactors resulted in a similar methane yield of slightly above 300Ml/kgoDM for cellulose as substrate (Fig. 1a). The described cumulative methane yield curves show that all the three inocula were active with slightly quicker performance for Inoculum 1. Figure 1(b) shows that all the three inocula consumed also DS to produce methane. Inoculum 2 form the biogas plant in Seevetal resulted in the highest methane yield. However, the difference from the other two inocula is not particularly high. In all three inocula the necessary AD fermentation microorganisms were active. Comparing DS with cellulose, it can be seen that the cumulative methane yield of cellulose was about two times than of DS (Fig. 1(b), Fig. 1(a)).
The organics in the DS is composed of fibers and fines made up of hemicelluloses and cellulose which can be easily degraded by anaerobic bacteria. However, non-degradable or hardly degradable organic compounds seem to be present. This could limit the utilization of the organics present in the sludge. No clear AD inhibition due to specific chemical constituents of the DS was observed. .

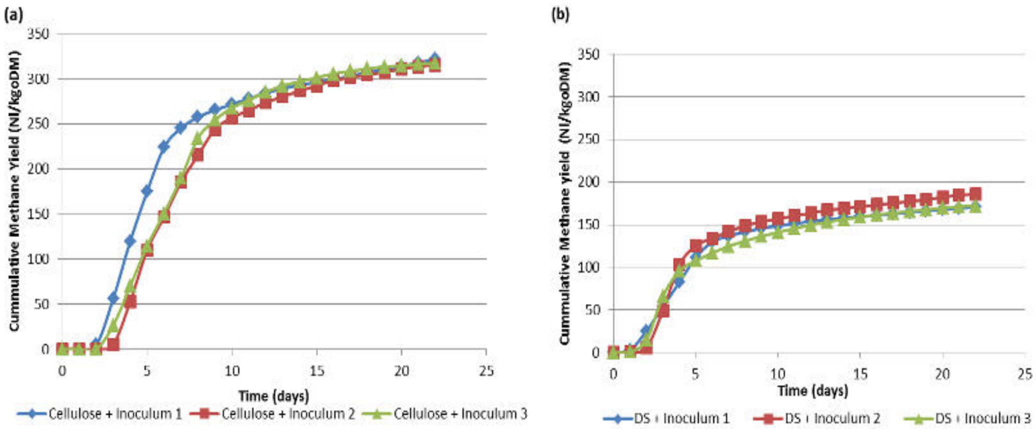

Figure1 (a.) Methane yield curve with cellulose as feedstock; (b.) Methane yield curve with deinking sludge as feedstock

For operating a biogas facility, the yield in relation to the fresh matter content is important. The biogas yields were 9.1 Nl/ kg FM, 11.5 Nl/ kg FM, 9.7 Nl/ kg FM respectively for the three inocula. The average biogas yield achieved in the batch test was 10.1 Nl/ kg FM. This is significantly higher than, for example the blackwater from vacuum toilets (2Nl/kg FM) [8] and from sewage sludge (5Nl/kg FM) [12], but also significantly lower than grease trap residues (30 Nl/kg FM) [8] or manure (pig manure (20 NL /kg FM, cow manure 30Nl/kg FM) [9]. This is owing mainly to the organics in the feedstock which is depending on the WC in combination with oDM.

3.3 Anaerobic digestion in semi-continuous experiments

The three reactors in the semi-continuous system were operated using slightly different starting inocula (chapter 2.1). Inocula 4 and 5 were digestates from a previous laboratory scale AD experiment. Inoculum 4 did not contain TEs, but Inoculum 5 did. Inoculum 6 was from WWTP Seevetal and was chosen, since it had the best performance in the batch test (Figure 1b). A further difference was regarding TEs supplementation during the process. While Reactor 1 and Reactor 3 had no TEs supplementation, Reactor 2 was supplemented with TES mixture (chapter 2.1).

In phase 1 and 2, feeding of DS was accompanied with inoculum. This was done so as to compare the result obtained with that in batch test. From phase 3 to 5 the feeding rate was increased step by step as described in Table 1. Only results of Phase 4 and phase 5 are described in the following.

The average Hydraulic Retention Times (HRT) and organic loading rates (OLR) were 72 days and 0.24 kg oDM/(m^3d) for phase 4 while 50 days and 0.79 kg oDM /(m^3d) for phase 5.

Figure 2 describes the biogas production from the DSs relative to reactor volume. All three reactors showed similar tendencies. However, Reactor 3 consistently showed the highest biogas production to the end of the investigation.

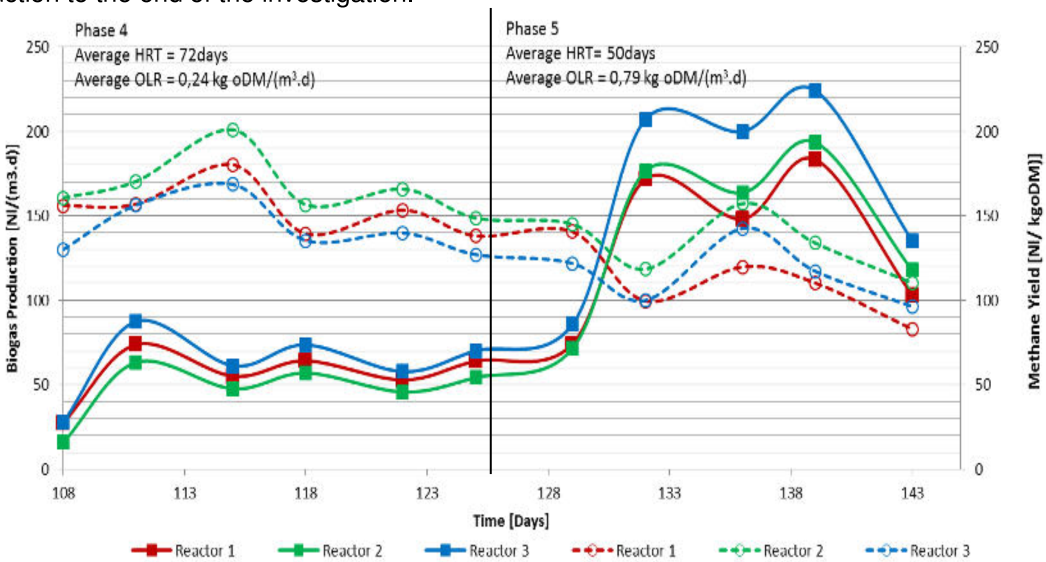

Figure 2: Biogas production related to reactor volume (pulled lines) and methane yield in relation to organic matter of DS (dashed lines) in the three semi –continuous experiments for phase 4 and 5

Furthermore, the biogas production curves indicate an increase in biogas production from phase 4 to phase 5. One reason is the different composition of the DSs used. In phase 4 were more biodegradables than phase 5 due to DS-composition: phase 4 had an oDM of 1.7 %FM, whereas the DS used in Phase 5 had an oDM of 4.5 %FM. The difference in the oDM and also in DM (2.8 %FM and 9.9 %FM respectively) was probably caused by variation in the settings of the pre-dewatering unit. Another reason is the increase of DS feeding from 400ml to 600ml (Table1) which increased the average OLR from 0.24 to 0.79kg oDM/(m^3.d) Typical OLRs of AD are between 2 -3 kg oDM/m^3d [13]. That means there is still room for improvement. Acidity-Alkalinity ratio (FOS/TAC) obtained in this phase were from 0.05 to 0.21. This show that no acid accumulation was encountered during phase 4 and phase 5 and also suggest that more potential for increasing OLR above 0.78 kg oDM/m^3d is possible.

Figure 2 describes also the methane yields related to oDM of the DS. For the three reactors (reactors 1,2,3), the average methane yields were 134Nl/kg oDM, 152 Nl/kg oDM and 130Nl /kgoDM respectively. These values are slightly lower when compared to the batch experiments (Figure 1(b)). Contrary to the trend described by the biogas production curves, a slight drop in the methane yield can be observed with the increase in OLR during the change from phase 4 to 5. This suggests that probably the DS of phase 5 contained more hardly biodegradable organics compared to the DS of phase 4. Comparing the methane yield curves of the three reactors, it is obvious that Reactor 2 delivered slightly high results. There exist a significant difference between the methane yield values of the reactor 2 (with TEs) and reactor 3 (without TEs) (two-tailed *t* test; $\alpha = 0.05$). This difference can be explained to be due to the TEs supplementation which improved the bioactivity of the methanogens. On the average the methane content observed in phase 4 and phase 5 were 62.0%, 63.35 and 53.8% for the respective reactors. Also Karlsson et al. (2011)

reported that TEs supplementation (0.5mg/l of Cobalt) in the anaerobic treatment of sludge from the pulp-and paper industry improved the transformation of acetate into methane and resulted in an improved biogas production [14].

3. Conclusion and outlook

This study confirms that deinking sludge is a bioresource for anaerobic digestion. Methane yields between 100 and 200Nl/kg oDM seem to be possible. Further optimization may increase yields. Options are pre-treatment by enzymatic hydrolysis, consideration of nitrogen deficiency and long-term studies to explore the microbial adaptation to substrate and supplemented TEs. Using biogas for heat generation to subsidies the high fossil heat demand of waste paper recycling process could be of economic benefit for waste paper facilities.

Acknowledgment:

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References

[1] Bajpai P. (2013). Recycling and Deinking of Recovered Paper. Elsevier Science, Amsterdam, ISBN: 978-0-12-416998-2

[2] Gromke U. and Detzel A. (2006). Ecological comparison of office papers in view of the fibrous raw material. IFEU Heidelberg: Institute for Energy and Environmental Research Heidelberg GmbH)

[3] Blanco A., Negro C., Fuente E. and Sánchez L.M. (2008). Alternative use of deinking sludge as a source of fibers in Fiber- cement manufacture. Cellulose chemistry and Technology. 42(1-3), pp 89-95

[4] VDI4630 (2006). Fermentation of Organic Materials: Characterization of the substrates, sampling, collection of materials data, Fermentation test. © Verein Deutscher Ingenieure, Düsseldorf, Germany

[5] Ezebuiro N.C (2014) Optimization of Anaerobic Digestion: Influence of trace Elements on Methanization Processes. Hamburger Berichte zur Siedlungswasserwirtschaft, Bd. 87., Hamburg University of Technology, Harburg. ISBN 978-3-942768-12-2

[6] Hertel, S., Nadal, J., Körner, I Biogas and fertilizer from lawn silage and lawn silage juice. Paper submitted for Ramiran 2015- 16th International conference on Rural –Urban Symbiosis

[7] Judith Olivia, Antonia Bernhardt, Hubert Reisinger, Manfred Domenig, Hans-Jörg Krammer (2009): Klärschlamm: Materialien zur Abfallwirtschaft ; Umweltbundesamt pp13

[8] Hertel, S.; Deegener, S. Körner, I. (2015): Biogas and nutrients from blackwater, lawn cuttings and grease trap residues – Experiments for Hamburg's Jenfelder Au district. Energy, Environment and Sustainability, in print.

[9] Karlsson A; Truong X; Gustavsson J; Svensson B. Nilsson F & Ejlertsson J. (2011): Anaerobic treatment of activated sludge from Swedish pulp and paper mills – biogas production potential and limitations, Environmental Technology, Volume 32, Issue 14, pages 1559-1571

[10] Kujala Asta (2012): Papermaking sludges and possibilities of utilization as material. Lappeenranta University of Technology, BH10A0300 Bachelor Seminar of Environmental Technology.

[11] Pöykiö R and Nurmesniemi H. (2008). Calcium carbonate waste from an integrated pulp and paper mill as apotential liming agent. Environmental Chemistry letters. Volume 6, Issue 1, pp 47-51

[12] Boxer99(2015):Biogaspotentialehttp://www.boxer99.de/biogas_biogaspotentiale.hm[accessed online 11/2014].

[13] Hertel, S.; Deegener, S. Körner, I. (2015): Biogas and nutrients from blackwater, lawn cuttings and grease trap residues – Experiments for Hamburg's Jenfelder Au district. Energy, Environment and Sustainability, in print.

[14] Karlsson A; Truong X; Gustavsson J; Svensson B. Nilsson F & Ejlertsson J. (2011): Anaerobic treatment of activated sludge from Swedish pulp and paper mills – biogas production potential and limitations, Environmental Technology, Volume 32, Issue 14, pages 1559-1571

[15] AbfAblV. *Verordnung über die umweltverträgliche Ablagerung von Siedlungsabfällen (Abfallablagerungsverordnung).* BGBl; 2001. I p. 305, modified 2006, modified 2006, BGBl. I S. 2860. http://www.gesetze-iminternet.de/bundesrecht/abfablv/gesamt.pdf; [accesed online 11/2014].

[16] Directive 1999/31/EC on the landfill of waste. Official J L 182; 42, ISSN 0378–6978. http://eurlex.europa.eu/legal-content/EN/TXT/?uri=OJ: L:1999:182:TOC; [accessed online 07/2015].

TE-O_19 Energy production and resource recovery on sewage plants in Austria

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1. Objectives

Regarding renewable energy production from sources other than digester gas, municipal waste water treatment plants (WWTPs) were almost ignored in Austria till now. As investigations on real world case studies have shown, WWTPs can not only treat waste water, there is also a high potential for the efficient use of so far wasted heat and other resources.

2. Methodology

The main tool used in this study is the Process Network Synthesis (PNS), which is based on the p-graph method, employing combinatorial rules for generating technology networks [1]. The PNS is used for economic optimisations of complex process networks, which are represented in a super structure. The super structure contains all possible technologies, raw materials and resources, intermediate products which are produced within the structure and used as inputs in other technologies, and products sold on the market. All energy and material flows, as well as other parameters as capacities and costs of materials and technologies, available amounts of raw materials, and demands and prices for products need to be defined. The super structure builds the basis for the PNS calculation, performed by the software tool PNS Studio [2]. The result of this optimisation is the optimum structure, i.e. the optimal technology network, maximising total revenues [3], as shown in Figure .

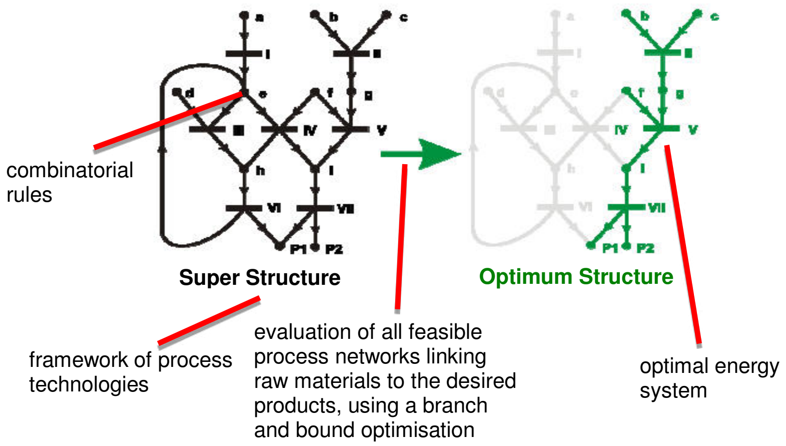

Figure 1: Super structure and optimum structure of a technology network [1], adapted.

3. Case study and results

3.1 Energy consumption and production at WWTPs

For decades, the main purpose of municipal WWTPs has been to remove certain substances from waste water (road grit, nutrients, etc.) for protecting receiving waters from pollution.

For the proper treatment of waste water, different treatment steps are applied (mechanical treatment, biological treatment, advanced treatment). All treatment related technologies as well as the common infrastructure of the WWTP itself have demands on electrical and thermal energy. In general, these energies are provided by external suppliers. However, certain amounts of WWTPs' heat and electricity demand are already covered from combustion and cogeneration of digester gas, which is being produced during anaerobic sludge treatment. In addition to digester gas, other (waste water based) sources of energy can be found at a WWTP as well.
Today, a shift in professional opinion can be observed from considering waste water as waste towards considering it as a resource. According to Stedman (2015) [4], WWTPs in the United

States now are seen and termed as waste water resource recovery facilities (WRRFs). Based on this approach, the authors of this article believe that WWTPs can also be integrated into local energy supply systems and thus serve as regional energy cells.

The following products can be produced at a WWTP:

Thermal energy:

- Low temperature heat (below 65 °C) by means of waste water heat pumps, solar heat, waste heat of pressure boosters
- High temperature heat (beyond 65 °C) by means of digester gas combustion, combined heat and power generation
- Cooling by means of waste water heat pumps

Electrical energy:

- Electricity by means of combined heat and power generation, gas turbines, turbines in the effluent of a WWTP, photovoltaic panels

Other resources:

- Treated waste water as service water
- Processed digester gas (biogas)
- Processed sewage sludge

Figure 2 gives an overview on the energy and resource consumption and production at a WWTP.

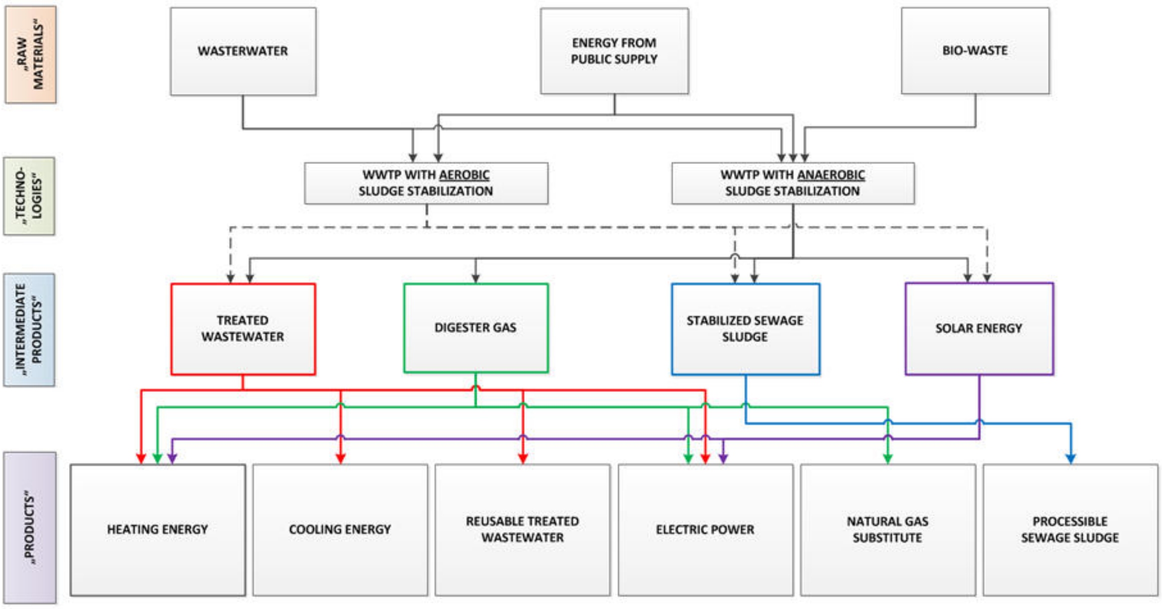

Figure 2: Energy and resource consumption and production at WWTPs [5]

3.2 Electricity and heat potentials at WWTPs in Austria

The annual amount of electricity generated from digester gas can be estimated based on the actual load entering the WWTP and an average energy supply of 15 kWh per population equivalent and year [6]. For the 158 Austrian municipal WWTPs equipped with digester towers [7], electricity in the amount of 115.5 GWh/a may be provided to cover energy consumption at the WWTPs themselves.

Moreover, thermal energy production from digester gas can be estimated in consideration of the actual load entering the WWTP and an average energy supply of 30 kWh per population equivalent and year [6]. Therefore, a thermal potential of 231 GWh/a arises from the above mentioned 158 WWTPs.

Concerning thermal energy from waste water, the available amount of treated waste water and the temperature level represent essential determining factors. Taking into account a 30 percent reduction of the wet weather flow, the annual runoff of about 1148 million cubic metres for 632 Austrian communal WWTPs with a capacity of more than 2000 population equivalents [8] results in an hourly dry weather discharge of about 91760 m³.

The annual amount of thermal energy from waste water can be estimated based on the following assumptions:

- The specific thermal capacity of waste water is calculated with the appropriate value of water (1.16 kWh/m³*K).
- The average waste water temperature in the heating period is estimated at 10 °C.
- The waste water in the effluent will be cooled down to 5 °C, so that 5 K can be extracted.
- Heat pumps with a performance factor of 4 are used for the energy generation in a dual mode system.

Assuming an application of the thermal energy in a mixture of functions (e.g. for residential, commercial and agricultural purposes), the annual duration of thermal extraction can be calculated in the order of 4500 hours, which is considerably higher than potential full load hours for the exclusive heat supply of residential buildings in the range of 1500 to 2200. Subsequently, the thermal energy potential from waste water can be estimated with an amount of 6386 GWh/a. This potential shall be contrasted for clarification of the magnitude with the heat generation in Austrian heating plants (without cogeneration of heat and power) of 8602 GWh in the year 2012 [9]. Table 1 presents the potential energy supply of Austria's WWTPs.

Table 1: Potential electric and thermal energy production at Austrian WWTPs.

Electricity / GWh_{el}	**Heat / GWh_{th}**	
Production from digester gas	Production from digester gas	Production from waste water
115.5	231	6386

3.3 The case study Freistadt

To make the results more visible on a smaller scale, three different municipal WWTPs (30000 - 150000 population equivalents) in Austria were chosen as case studies in a current national research project. The research is focused on an economic optimisation of the available energy producing technologies, by calculating an optimal energy supply system for the plant, using PNS Studio [10].

One of those case studies is the WWTP Freistadt with a population equivalent of 30000, located in a city of 7500 inhabitants in the province of Upper Austria. The average waste water load of nearly 1.7 million m³/a is treated mechanically and biologically. The result of the PNS Studio calculation shows that it is economically worthwhile to produce electricity on the plant location and cover parts of the in-house demand to reduce the need for net electricity. The remaining energy potential is converted into heat. In
Table, the potential of electrical and thermal consumption and production in the plant in Freistadt is illustrated. It is to be noted, that high heat production levels come at the cost of lower electricity production, since the high heat price in those scenarios favours heat production. These intermediate results are sensitive to changes in energy prices, external demands and distances to energy consumers.

To evaluate the possible thermal energy production on the plant, a sensitivity analysis was carried out using PNS Studio, considering different heat prices [11]. Up to a price of 35 €/MWh_{th}, which is considerably lower than the current district heating price, the WWTP Freistadt only covers its in-house heat demand of roughly 497 MWh_{th}/a. In this scenario the heat producing technologies would be a micro gas turbine and a gas burner. Taking higher heat prices as well as heat consumers in the surrounding area into consideration, more heat producing technologies were included in the optimum energy system solution within the PNS results, which are producing a much higher heat amount on the WWTP, as shown in Table 2.

.

Table 2: Potential annual energy production and consumption of the case study Freistadt.

Heat price / €/MWh		Electricity / MWh			Heat / MWh		
		Production	Consumption	Balance	Production	Consumption	Balance
35	WWTP[a]	292	671	-379	546	496	50
	Waste water heat pumps						
50	WWTP[b]	292	849	-3962	1812	497	12491
	Waste water heat pumps		3405		11176		
80	WWTP[c]	53	849	-4201	2085	497	12764
	Waste water heat pumps		3405		11176		

WWTP-waste water treatment plant

[a] including a micro gas turbine, a gas burner, and photovoltaic panels; [b] including a micro gas turbine, a gas burner, a heat pump recovering waste heat from boosters, solar thermal energy, and photovoltaic panels; [c] including a gas burner, a heat pump recovering waste heat from boosters, solar thermal energy, and photovoltaic panels

The results show that not only the in-house energy demand would be covered, it could moreover be economically reasonable to supply surrounding areas of the plant with additionally produced heat. At a heat price of 50 €/MWh, for instance, only 7% of the electric demand (including heat pumps) can be covered, but the heat produced is 26 times the internal demand.

4 Conclusion and outlook

The results of the PNS calculations for economically optimal energy producing technologies on an Austrian WWTP show that the internal thermal energy requirements could be covered by the plant itself and, under certain conditions, excess energy can be used to meet external demands of neighbouring consumers. First results show that especially the in-house heat demand on a plant could be covered by various technologies. For instance, heat exchangers and heat pumps using the warm waste water as source seem to have a high potential. Even parts of the internal electrical energy demands could be covered by the WWTPs themselves. Moreover, including local stakeholders helped to evaluate the market potentials of the produced resources like heat, electricity and biogas and integrate them into the energy network of the plant and the surroundings.

Acknowledgements

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References

[1] Friedler, F.; Varga, J.B.; Fan, L.T. (1995): Decision-mapping: a tool for consistent and complete decisions in process synthesis. Chemical Engineering Science Vol. 50, 1755-1768

[2] PNS Software Version 3.0.4 (2011): <www.p-graph.com>, accessed on 08/05/2015

[3] Maier S., Narodoslawsky M. (2014): Optimal Renewable Energy Systems for Smart Cities, Editors: Jiří Jaromír Klemeš, Petar Sabev Varbanov and Peng Yen Liew, Computer Aided Chemical Engineering, Elsevier, 24th European Symposium on Computer Aided Process Engineering Volume 33, pp 1849-1854, SN - 1570-7946

[4] Stedman, L. (2015): The wastewater resource recovery facility of the future. Water 21, Magazine of the International Water Association, 41-42

[5] Kretschmer, F., Weissenbacher, N., Ertl, T. (2015): Integration of Wastewater Treatment Plants into Regional Energy Supply Concepts, Sustainable Sanitation Practice, Issue 22, ISSN 2308-579, 4-9

[6] Lindtner, S. (2008): Leitfaden für die Erstellung eines Energiekonzeptes kommunaler Kläranlagen (Guideline for the Development of an energy concept for municipal wastewater treatment plants), Austrian Federal Ministry of Agriculture and Forestry, Environment and Water Management, Vienna, Austria

[7] Spatzierer, G. (2014): Nachbarschafts- und Kläranlagenliste (List of wastewater treatment plants and neighbourhoods). Kanal- und Kläranlagennachbarschaften, Folge 22, Austrian Water and Waste Management Association, Vienna

[8] BMLFUW, UBA – Austrian Federal Ministry of Agriculture and Forestry, Environment and Water Management, Environment Agency Austria (2014): Annual runoff of Austrian wasterwater treatment plants, data status Council Directive 91/271/EEC Concerning Urban Wastewater Treatment – Austrian Report 2014

[9] BMWFW - Austrian Federal Ministry of Science, Research and Economy (2014): Energiestatus Österreich 2014. Entwicklung bis 2012. (Austrian energy status 2014. Development up to 2012), Vienna

[10] Kollmann, R., Maier, S., Shahzad, K., Kretschmer, F., Neugebauer, G., Stoeglehner, G., Ertl, T., Narodoslawsky, M. (2014): Waste Water Treatment Plants as Regional Energy Cells - Evaluation of Economic and Ecologic Potentials in Austria, Chemical Engineering Transactions, Vol. 39, ISSN 1974-9791, 607-612, Prague, Czech Republic

[11] Kindermann, H., Kollmann, R. (2015): Optimisation of Regional Energy Systems Centred on Wastewater Treatment Plants, Sustainable Sanitation Practice, Issue 22, ISSN 2308-579, 10-14

TE-O_20 Estimation of design values for greywater treatment units

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1. Objectives

New sanitation systems are based on the separate collection and treatment of different wastewater flows. In Germany there are no design values for greywater treatment units available. This article elaborates characteristic values for greywater. Within the KREIS project a literature study, comprising more than 140 sources, was carried out to improve the data base. The literatures were reviewed and the greywater-data were evaluated statistically. To characterize the greywater and to improve the available several sampling campaigns were carried out at different demonstrations sites.

The HAMBURG WATER Cycle® (HWC) is an innovative integrated water and energy production concept based on a 2-material-flow system (blackwater and greywater), which will be implemented, evaluated and further developed within the Project KREIS for the new district "Jenfelder Au" in Hamburg (Germany) for approximately 2,000 inhabitants [1]. The Basis of the HWC is the separated collection, drainage and on site treatment of wastewater flows from households.

Greywater consists of discharges from washbasins, showers, baths, kitchen sinks, dishwashers and washing machines. Greywater excluding discharges from kitchen and washing machine is usually called light greywater, while greywater that includes only wastewater from washing machines, dishwashers or kitchen sinks is referred as dark greywater [1, 2, 3, 4]. Within the scope of this study greywater consists of discharges from washbasins, showers, baths, kitchen sinks, dishwashers and washing machines. The composition of the greywater highly depends on user's behavior, the equipment of the households (e.g. dishwasher), the used cleaning agents (e.g. detergents, soaps; etc.), the quality water supply and the type of water distribution [5, 6]. Greywater accounts for about 50 – 80 % of the total wastewater flow produced by households [5,7; 8, 9]. In Germany about 60 – 90 litre per capita and day (lpcd) of greywater is produced [1,10, 11].

The knowledge of loads and concentrations is required for the design of greywater treatment units and the assessment of negative environmental impacts. The objective of this study is the characterization of greywater and the improvement of the data base of greywater-values on the one hand and the estimation of design values for greywater treatment units on the other. Therefore as part of the KREIS project a literature-review was conducted, complemented by five sampling campaigns at the demonstration projects Berlin "Block 6" and Lübeck "Flintenbreite (Germany).

2. Methodology

The aim of the literature study is the estimation of design values using available data from a wide range of studies. The currently available design values are derived from studies in other countries and refer to specific values which depend on the particular framework conditions. Furthermore they are often based on small project areas or only few measurements [12].

More than 140 references were included into the desk study. The majority of the reviewed datasets with about 70 % are from Europe. In order to calculate standard values for the European context, the statistical analysis of the greywater characteristics is based on the data of European references with a focus on the complete household greywater flow. Most of the European data are from Germany (58 %), Sweden (16 %) and the Netherlands (11 %). Literature sources referring to mainly commercially objects were not taken into account. If no information on average values was given, these values were estimated based on the minimum and maximum values. Wherever the data could be converted into another unit, e.g. concentrations and volumes (flows) into loads it was done for the statistical analysis. Statistical values were used for the detailed description of the results. The data collection includes volumes, suspended solids (TSS), organic pollution (COD and BOD_5) and the nutrients nitrogen (TN, NH_4-N) and phosphorus (TP, PO_4-P).

The sampling campaigns were carried out by the Bauhaus-Universität Weimar in Berlin "Block 6" and the ecological housing estate Lübeck "Flintenbreite". A new sampling-method was tested, which allows the separate assessment of the liquid and the solid phase of the greywater flow. All liquid samples were collected proportional to the flow. The greywater-sludge was collected separately by a filter bag and dewatered by gravity for two hours before analysis. A detailed description of the sampling technique is given in Sievers et al. (2014) [13]. The samples were analysed in terms of total suspended solids (TSS), volatile suspended solids (VSS), chemical oxygen demand (COD), biological oxygen demand (BOD_5) and nutrients (total nitrogen (TN), ammonium nitrogen (NH_4-N), total phosphorus (TP), ortho-phosphorus (PO_4-P)). Analyses of TSS and VSS of the liquid phase as well as dry residue (TS) and the organic part of the dry residue (oTS) of the grey-water-sludge were performed according to German standard methods. COD, TN, NH_4-N, TP and PO_4-P were analysed with cuvette tests from Hach Lange and Hach Lange DR 3900 Spectrophotometer. BOD_5 measurements were performed using the respirometric method with OxiTop systems of WTW. All parameters were determined as repeat determination. COD, TN and TP analyses of the solids were performed by accredited laboratories in Weimar and Lübeck in accordance to German standard methods.

The demonstration project “Block 6” is located in the district Friedrichshain-Kreuzberg in Berlin, Germany. All in all 71 apartments for 207 residents are connected to the greywater system, which collects the wastewater from showers, bathtubs, washbasins, kitchens and washing machines. The greywater is recycled and used for toilet flushing [14]. The three sampling campaigns were performed in a multistory building of “Block 6” with 20 apartments and 51 residents. They were realized within the building for a period of 7 days in November 2012, 10 days in April 2013 and 10 days in March 2014. All samples were taken from a collection pipe inside the building.

The ecological housing estate “Flintenbreite” is located in Lübeck, Germany. “Flintenbreite” was designed for 350 inhabitants in around 100 residential units. Currently about 200 people are living in 57 housing units in “Flintenbreite”. The sampling campaigns were conducted for a period of 12 days in November 2013 and 11 days in May 2014. The measuring point was placed in the first sedimentation tank of the greywater treatment unit. Greywater was collected from 12 semi-detached houses with 46 residents.

3. Results

3.1 Literature Study

The composition of greywater varies widely depending on factors such as drinking water source, quality of water supply, water use, household activities, socio-economic, cultural factors and the country [5, 12]. Literature reviews on greywater composition usually show a great variability in terms of volumes, concentrations and loads. The major problem in this literature study was the inconsistent data quality. In particular the reviewed articles gave no information about the number of samples or showed a great variance in the number of samples and there was often to little information about the methods of sampling and analysis.

Table 1 shows the average values, standard deviation, median as well as minimum and maximum of the greywater volumes and the loads per capita and day (gpcd) based on literature data. Greywater volumes per capita and day (lpcd) were about 79 lcpd on average, ranging from 19 – 157 lcpd. The COD and BOD_5 loads are within 11 – 103 gpcd and 5 – 37 gpcd respectively with average values of 42 gpcd for COD and 18 gpcd for BOD_5. The COD/ BOD_5 -ratio was calculated to 2.3 which is slightly higher than for domestic wastewater. For the parameter TSS an average value of 20.6 gpcd was calculated. The data analyses of the TP load showed an average valuesof 0.4 gpcd with a minimum of around 0.1 gpcd and a maximum of 0.8 gpcd. TN and NH_4-N ranged between 0.4 – 3.1 gpcd and 0.03 – 1.1 gpcd respectively. Average values were 1.2 gpcd for TN and 0.3 gpcd for NH4-N. For aerobic treatment C:N:P -ratios between 100:10:1 and 100:5:1 are optimal [15]. Based on the literature study the average C:N:P- ratio is 100:2,9:1 indicating a nitrogen deficiency for aerobic treatment.

Table 1: Volumes and loads per capita and day based on the desk study

Parameter		n	Unit	Mean	SD	Median	P-85%	Range
Volume	Q	63	lpcd	79	28	71	110	19 - 157
	TSS	32	gpcd	20.6	21.2	11.4	46,00	1 - 71
org. Matter	BOD_5	39	gpcd	18	7	18	24	5 - 37
	COD	56	gpcd	42	18	42	57	11 - 103
Nutrients	TP	58	gpcd	0.4	0.17	0.5	0.6	0.07 - 0.8
	PO_4-P	16	gpcd	0.3	0.3	0.2	0.5	0.03 - 1.35
	TN	57	gpcd	1.2	0.6	1	1.7	0.4 - 3.1
	NH_4-N	21	gpcd	0.3	0.3	0.2	0.65	0.03 - 1.1

3.1 Sampling campaigns

The Results of the sampling campaigns in „Block 6“ and Lübeck “Flintenbreite” show on average higher loads for the organic parameters with 36 – 105 gpcd for the COD and 20 – 54 gpcd for the BOD_5. COD and BOD_5 mean values were around 61 gcpd and 33 gpcd in “Block 6” and 44.8 gcpd and 26 gpcd in Lübeck. The COD/BOD_5-Ratios can be calculated to 1.84 and 1.72 respectively and indicates a good aerobic biodegradability. For TSS and VSS loads in a Range of 1.7 – 15.5 gpcd and 1.2 – 14.3 gpcd were found. The TSS and VSS loads in Berlin were significantly higher on average with about 8.2 and 6.7 gpcd than in Lübeck with 4.9 and 3.6 gpcd.

Inhabitant specific loads of total Nitrogen (TN) and total Phosphorus (TP) were in a similar range like values found in the literature. The average loads of TN were about 1.2 gpcd (Berlin) and slightly lower with 0.84 gpcd in Lübeck. Average NH_4-N loads in Berlin and Lübeck were in similar ranges with 0.15 gpcd and 0.13 gpcd respectively. The overall Ranges for TN and NH_4-N loads were between 0.5 – 2.1 gpcd and 0.03 – 0.4 gpcd. Total Phosphorus TP showed range from 0.19 – 0.76 gpcd, but the average loads per capita and day for the TP values in Berlin and Lübeck were equal with 0.4 gpcd. Table 2 shows the inhabitant specific loads along with the volumes for Berlin “Block 6 and Lübeck “Flintenbreite”.

Table 2: Average loads per capita and day in Berlin “block 6” and Lübeck “Flintenbreite”.

Parameter		Berlin						Lübeck				
		n	Unit	Mean	SD	Median	Range	n	Mean	SD	Median	Range
Volume	Q	27	lpcd	78	14	77	61 - 114	23	57,10	6.1	56,00	48 - 69
	TSS	26	gpcd	8.2	2.8	7.4	2.9 - 15.5	22	4.9	1.5	5.2	1.7 - 7.4
org. Matter	oTS	26	gpcd	6.7	2.7	6.1	2.1 - 14.3	22	3.6	1.1	3.8	1.2 - 5.5
	BOD_5	22	gpcd	33	11	30	20 - 54.4	22	26	6	26	18.5 - 42.2
	COD	26	gpcd	60	20	53	36 - 105	23	45	7	43	35.4 - 65.8
Nutrients	TP	27	gpcd	0.4	0.14	0.3	0.19 - 0.76	23	0.4	0.11	0.4	0.2 - 0.63
	PO_4-P	27	gpcd	0.1	0.07	0.1	0.05 - 0.33	23	0.1	0.03	0.1	0.04 - 0.18
	TN	27	gpcd	1.2	0.3	1.1	0.7 - 2.1	23	0.84	0.22	0.83	0.5 - 1.1.7
	NH_4-N	27	gpcd	0.15	0.07	0.13	0.01 - 0.4	23	0.13	0.05	0.13	0.08 - 0.25

The values of all measurements are summarized in Table 3. It shows the overall average values, standard deviation, median, 85%-Percentile and the minimum and maximum of the greywater volumes and the inhabitant specific loads based on all measurements. The results of the measurement campaigns are compared with the data of the literature study and the values of the DWA-A 272. DWA-A 272 gives inhabitants specific values for greywater for the German context, which are based on a relatively small number of data-sets [1].

The comparison of the different data shows that the average value of greywater volumes determined during the sampling campaigns is about 10 % lower than volumes given in the literature and DWA-A 272 respectively. Particularly COD and BOD_5 mean values of 54 gpcd and 29 gpcd are significantly higher than the literature values which are around 42 gpcd for COD and 18 gpcd BOD_5. On the other hand the TSS loads given in the literature are considerably larger ranging from 13 – 20.6 gpcd than TSS loads of around 7 gpcd found during the sampling. The inhabitant specific loads for TN and TP measured in Berlin and Lübeck show a good correlation with the values given in the literature and are about 1 gpcd and 0.5 gpcd.

Table 3: Average loads per capita and day of the sampling campaigns, the literature study and the DWA-A 272 (2014).

Parameter		Sampling campaigns							Literature		DWA
		n	Unit	Mean	SD	Median	P-85%	Range	Mean	P-85%	Median
Volume	Q	50	lpcd	68	15.2	67	84	48 - 114	76	110	75
	TSS	48	gpcd	6.7	2.8	6.4	9.4	1.7 - 15.5	20.6	46	13
org. Matter	oTS	48	gpcd	5.2	2.8	4.8	7.1	1.2 - 14.3	—	—	—
	BOD_5	44	gpcd	29	9.2	27	38	18.5 - 54.4	18	24	18
	COD	49	gpcd	54	17.1	48	68	35.4 - 105	42	57	47
Nutrients	TP	50	gpcd	0.4	0.13	0.4	0.5	0.2 - 0.8	0.4	0.6	0.5
	PO_4-P	50	gpcd	0.1	0.05	0.1	0.14	0.04 - 0.33	0.3	0.5	—
	TN	50	gpcd	1	0.3	1	1.3	0.52 - 2.1	1.2	1.7	1
	NH_4-N	50	gpcd	0.14	0.08	0.13	0.21	0.01 - 0.4	0.3	0.65	—

4. Conclusion

The compilation of greywater literature data and values from sampling campaigns show a large variability concerning the parameters TSS, COD and BOD_5. The average COD and BOD_5 loads found during the measurements are particularly higher than values from literature and contain around half of the organic load that can be found in mixed domestic wastewater. The resulting TSS loads based on the measurements on the other hand were notably lower compared to the values found in the literature. Based on the measurements future design values for COD and BOD_5 are probably higher and for TSS slightly lower than values given in the literature. Regarding the TN and TP inhabitant specific loads the literature and the DWA values mesh with the results of the sampling campaigns. Probably future design values for TN and TP may be in same order of magnitude. Nevertheless further investigations are required to gather a higher data density and more detailed information on greywater characteristics before design values can be conclusively established.

References

[1] (DWA 2014): Arbeitsblatt DWA-A 272. Grundsätze für die Planung und Implementierung Neuartiger Sanitärsys-teme (NASS), June 2014 (Standard DWA-A 272 Principles for the planning and implementation of ecological sanitation Systems). Ed.: Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall e. V. (German Association for water, wastewater and waste), Hennef, Germany.

[2] Birks R., Hills S. (2001): Characterisation of Indicator Organisms and Pathogens in Domestic Greywater for Recycling. Environmental Monitoring and Assessment, Vol., 129 (2007).

[3] Chaillou K., Gerente C., Andres Y., Wolbert D. (2011): Bathroom Greywater Characterization and Potential Treatments for Reuse. Water Air Soil Pollution, Vol. 215; No. 1 -4; pp. 31–42.

[4] Friedler E. Hadari M. (2006): Economic feasibility of on-site greywater reuse in multi-storey buildings. Desalination 190 (2006), pp.221–234.

[5] Eriksson, E.; Auffarth, K.; Eilersen, A.-M.; Henze, M.; Ledin, A. (2002): Characteristics of grey water. Urban Water, Vol. 4 (2002), pp. 85 -104.

[6] Oldenburg M., Meinzinger F., Nisipeanu P., Schneider C. (2008): Begrifflichkeiten und Kennzahlen zur Zusammensetzung von Teilströmen neuartiger Sanitärsysteme. KA Korrespondenz Abwasser, Abfall 2008 (55) Nr. 10. pp. 1113 – 1119.

[7] Friedler E. (2004): Quality of individual domestic greywater streams and its implication for on-site treatment and reuse possibilities. Environmental Technology, Vol. 25, pp. 997-1008.

[8] Lazarova V., Hills S., Birks R. (2003): Using recycled water for non·potable, urban uses: a review with particular reference to toilet flushing. Water Science & Technology, Vol. 3 No. 4, pp. 69- 77. IWA publishing 2003, London.

[9] Sheikh, B. (2010): White Paper on Graywater. American Water Works Association (AWWA), Water Environment Federation (WEF), and the WateReuse Association (Hrsg.). URL: http://www.watereuse.org/files/s/docs/Graywater_White_Paper.pdf [04.01.2015].

[10] fbr H 201 (2005): Grauwasser-Recycling. Planungsgrundlagen und Betriebshinweise. Hrsg.: Fachvereinigung Betriebs- und regenwassernutzung e.V., Havelstr. 7, Darmstadt.

[11] Otterpohl R., Oldenburg M., Büttner S. (1999): Alternative Entwässerungskonzepte zum Stoffstrommanagement. Editor: Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall e. V. (DWA). Hennef, Germany. Korrespondenz Abwasser, Abfall·1999 (46) Nr. 2, pp. 204 - 212.

[12] Meinzinger, F.; Oldenburg, M. (2009): Characteristics of source-separated household wastewater flows: a statistical assessment. Water Science & Technology, Vol. 59, No. 9, pp. 1785–1791.

[13] Sievers, J. C.; Londong, J.; Albold, A.; Oldenburg, M. (2014): Characterisation of Greywater – Estimation of Design Values. In J. Lohaus, ed. Proceedings of 17th International EWA Symposium "WatEnergyResources – Water, Energy and Resources: Innovative Options and Sustainable Solutions" during IFAT, 5-9 May 2014, Munich, Gemany. Hennef: European Water Association.

[14] Nolde, E. (2012): Personal Information from 11.11.2012. Nolde & Partner – Innovative Wasserkonzepte, Berlin.[15] Mudrack K., Kunst, S. (1994) Biologie der Abwasserreinigung. Ed.: Gustav-Fischer- Verlag, 4. Edition, 1994.

TE-O_21 Interfacing urban water management and agriculture – Transformation steps towards new alternative sanitation systems

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1. Objectives

The Water Framework Directive (WFD) of the European Union requires the member states to ensure a good qualitative and quantitative status of all water bodies by 2015. Hence there is a sense of urgency: National legislature is required to implement the political will and the requirements of the European Union in federal and state law. These legislations put pressure especially on the wastewater management associations in regions where no collection or treatment systems are installed yet. The German federal state Thuringia is an example: Ten years ago 50 % of the rural areas have not been equipped with an appropriate wastewater infrastructure [1]. Since most of the investment was done in settlements with more than 2000 inhabitants, this elicitation can still be considered as a current guide value.

Additionally, the conditions in rural areas, which are affecting urban water management, are becoming more and more volatile. The scientific community names in particular demographic and climate change, increasing energy prices and nutrient scarcity as main challenges [2]. Almost all components of conventional grid-bound water infrastructure systems are in use for very long period of time and are associated with high investment and maintenance costs [3]. Furthermore, due to a decreasing population future prospects are deteriorating. Building a conventional system with high fix costs in an area with a declining population will lead to high and even rising fees.

More flexible solutions, which seek to implement an integrated approach and a local value-added chain, are an appropriate countermeasure. Therefore we want to discuss in this paper, how a New Alternative Sanitation System (NASS) can counteract the crucial development by establishing a new partnerships between urban water management and agriculture.

2. Methodology

In the context of the research project TWIST++ wastewater systems for a rural model region in Germany are designed. The goal of the project is to adapt water infrastructure to future challenges. Beside conventional systems, a NASS was identified to be an appropriate measure, too. The key objective of implementing NASS is to establish a circular, reuse orientated system (Figure 1). Reusing water and extracting nutrients and organic matter from wastewater becomes feasible by separation of the different wastewater streams from toilet (blackwater) and bathroom and kitchen (greywater) [2]. Furthermore, the technology can be adapted to the particular pollutants and to the quality requirements on reused water. Consequently NASS can not only respond to technical requirements but also to socio-economic developments [4].

A specific NASS was designed for the examined model region. A co-fermentation of blackwater and organic residues from agriculture (Figure 2) closes the local value-added circle of nutrients and energy. This aspiration is fulfilled by using the digestate as fertilizer on the fields of the co-operating farm. Additionally, the produced energy can be used in an appropriate manner. A central treatment plant is intended to process the residual greywater and the rainwater. Depending on the level of pollution and the structural condition, the existing gravity sewer can drain the greywater [5]. Otherwise a (partly) restoration of the sewer system is necessary.

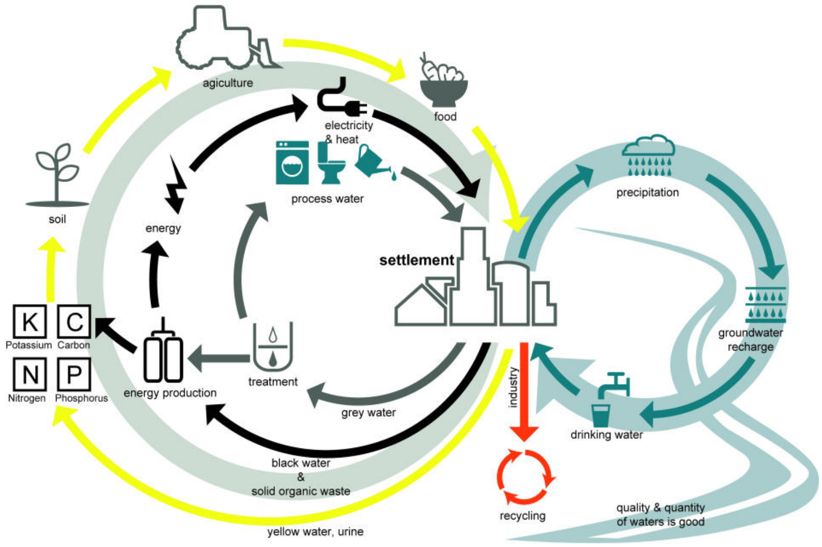

Figure 1: Concept for future urban material flows: New Alternative Sanitation Systems (NASS).

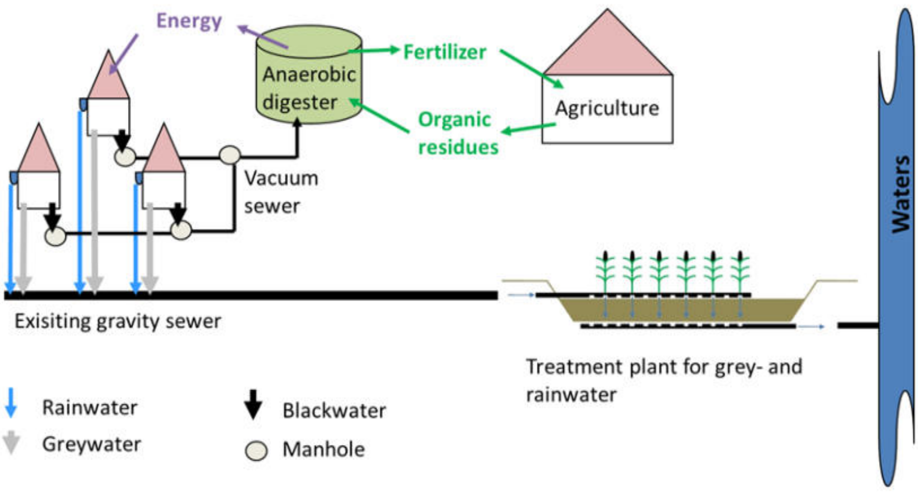

Figure 2: System overview of the designed New Alternative Sanitation System for the model area.

Besides the scope of the research project, the responsible wastewater disposer of our model region is now aiming to implement this concept in its associated area. Regarding the local boundary conditions almost 50 % of the 9.000 inhabitants in 24 villages are not connected to a biological wastewater treatment. Most of these villages are supplied only with a sewage system. It means in effect that the pre-treated wastewater is directly discharged into the water bodies. A first survey in two villages reveals that around 70 % of the population are satisfied with this situation. Furthermore 94 % reject higher fees to improve water infrastructure. Moreover the examined area is afflicted with demographic change with a decreasing population.

3. Results and discussion

This paper emphasizes a conceptual and planning approach in order to implement the core idea. The integration of transformation and management strategies helps to accomplish a more flexible system, which would be able to adapt to future changes. During the planning phase we observed legal barriers which have to be overcome to implement a NASS in cooperation with agriculture.

3.1 Transformation Steps

Since the introduction of NASS requires various changes in the existing system, many different aspects like legal requirements and issues of acceptance need to be considered [4], [6], [7]. Our approach is to transform the infrastructure from the current status over a probably long period of time into a future-oriented, target system. Among the main requirements of such an innovative system implementation during a long-lasting process are:

- During the process of transforming existing infrastructure facilities the service should be available continuously. This necessity requires a mindful combination of transformation steps.
- An efficient use of financial resources is important especially in rural regions with decreasing population. Thus the transformation steps have to be assessed to what extent they can avoid sunk cost. The commercial service life of each component should be exploited as far as possible.
- The boundary conditions, which determine the target system, may change. Therefore alternative transformation steps are preferable in order to achieve a higher flexibility.

After having worked now for two years on the project we have determined the following transformation steps for the selected NASS (Table 1).

Table 1: Technical transformation steps to implement a NASS.

	1. Central Treatment Plant	2. Vacuum Sewer System	3. Biogas plant	4. In-house separation	5. Sewer Restoration
Description	Build a central wastewater treatment plant at the outlet of the existing sewer to clean the pre-treated wastewater and the rainwater.	Install a vacuum sewer system, which drain the wastewater to a central point.	Establish cooperation with agriculture to build a co-fermentation of high polluted wastewater and organic residues.	Offer incentives for all homeowners to install in-house separation of black- and greywater.	Restore existing gravity sewer system.
Requirement	Preservation of settling pits on private property.	Treatment of this stream must be ensured all along.	Legal barriers have to be overcome.	Organisational framework has to be established.	-
Advantage	Takes load off the water bodies.	Demographic resistant system.	Exploitation of wastewater-bound energy and nutrients .	Increases organic concentration of wastewater stream (input of biogas plant).	Prevents the immission of groundwater and the emission of wastewater.
Alternative	-	Restoration of the existing sewer system and cancellation of the NASS approach.	If an agricultural biogas plant is already existing locally, making use of this has to be considered.	Using other co-substrate to increase the concentration.	Building a total new sewer system, which allows rainwater separation.

3.2 Legal Framework

In general the institutional context evolves together with the technical innovations frequently over a long time period. This interdependency helps to stabilize both processes, but prevents change, also [4]. An example for a legal barrier of the designed NASS is the currently forbidden use of wasterwater digestate as fertilizer.

Londong & Hartmann (2014) suggest politics to become rather an enabler than a developer [8]. Through a temporary state of exception, open spaces for improvisation, provocation, experiments and conflicts are accomplished [8]. Thus the advancement of the accepted rules of engineering shall not be prevented, but some well-defined boundaries should be set.

According to these considerations the responsible wastewater disposer decided to submit the project idea to the "Internationale Bauausstellung (IBA) Thüringen". A main objective of the IBA Thüringen is to create realms of freedom off the beaten paths. Using their network we have been able to enter into a dialogue with the law-making body in Thuringia even at an early stage of the IBA.

4. Conclusion and discussion

New Alternative Sanitation Systems (NASS) are an adequate measure to establish a valueadded, resource oriented circular economy on a regional level. Hereby future prospects of rural areas will be enhanced, although implementing innovative infrastructure systems requires a careful planning. We suggest transition pathes, which step by step transform existing systems into NASS. The current state of the project is to overcome legal barriers in order to increase acceptance of all stakeholders. Additionally, the basic design of the first transformation step is in progress. At the moment, an organizational model is developed, too.

Acknowledgements

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References

[1] W. Kämpfer, M. Berndt, and J. Londong, “Zustandserfassung von Kleinkläranlagen in Thüringen und Vorschläge für die technische Umsetzung der Anforderungen der Abwasserverordnung,” Erfurt, 2005.

[2] DWA, “Do we need New Alternative Sanitation Systems in Germany ?,” Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall e. V. (DWA), Hennef, Germany, 2010.

[3] T. Hillenbrand, J. Niederste-Hollenberg, E. Menger-Krug, and S. Klug, “Demografischer Wandel als Herausforderung für die Sicherung und Entwicklung einer kosten- und ressourceneffizienten Abwasserinfrastruktur,” Dessau-Roßlau, 2010.

[4] S. Panebianco and C. Pahl-Wostl, “Modelling socio-technical transformations in wastewater treatment—A methodological proposal,” *Technovation*, vol. 26, no. 9, pp. 1090–1100, Sep. 2006.

[5] H. J. Temann, *Teilortskanalisationen - Möglichkeiten der Integration eines etablierten Provisoriums in zukunftsfähige Systeme*. Dissertation. Schriftenreihe des Bauhaus-Instituts für zukunftsweisende Infrastruktursysteme an der Bauhaus-Universität Weimar: Rhombos-Verlag Berlin, 2014.

[6] A. Poplin, “Playful public participation in urban planning: A case study for online serious games,” *Computers, Environment and Urban Systems*, vol. 36, no. 3, pp. 195–206, May 2012.

[7] H. Söbke and J. Londong, “Promoting innovative water infrastructure systems: Simulation games as virtual prototypes,” in *Proceedings of 17th International EWA Symposium “WatEnergyResources – Water, Energy and Resources: Innovative Options and Sustainable Solutions””*, 2014, vol. 3, no. 3.

[8] J. Londong and M. Hartmann, “Angepasste Einzelfalllösungen unter wegweisenden politischen Rahmenbedingungen statt detaillierter administrativer Vorgaben - ein Plädoyer für die Ingenieur- und Rechtskunst Problemstellung und Ausgangslage,” in *47. ESSENER TAGUNG für Wasser- und Abfallwirtschaft “Ist unsere Wasserwirtschaft zukunftsfähig?,”* 2014.

Thematic Area TE – Sustainable regions (Poster presentations)

Ina Körner: Wood, processed photograph

TE-P_01 Agriculture as a mean of sustainable resettlements – A proposal of symbiotic agricultural and micro-economic projects for internal displaced persons (IDPs) in Darfur, Sudan

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1. Introduction

Darfur suffers from ongoing conflict for more than 9 years. It is an evident that water scarcity is the root of most conflicts of farmers and nomadic communities. The partial peace agreement emphasised the need for the return and resettlement of war victims to their place of origin in order to pave the road for a complete peace and resolution. This resulted many initiatives concerned of repatriation. The poster reveals the suitability of onsite treatment and nutrient recovery as a possible solution to ensure more sustainable repatriation of the internal displaced persons –war victims- and illustrates the challenges to provoke the technology transition.

2. General background

Drafur is located in the west of Sudan and consists of 5 states. It has a population of 7.5 million, 2.2 millions of them are registered as internal displaced persons (IDP's). Out of those 155,000 are verified voluntarily returned to their place of origin (IOM, 2014). The significant resources for Darfurian livelihood are water, fodder, wood and soil (Jeremy and Alan, 2012).The resources vulnerability are considered as one of the primary factors contributing to the conflict. Although the political context is complex and the sources of conflict are not always easy to define, the socio-economic environment of nomads and farmers competing over scares water source is considered as a trigger for many conflicts between local communities**.** The largest IDP camps are located in the surroundings of the capitals of the states Nayala, Elfasher, Zalengei, Elgeneina creating slum-like situations. The water quality levels have very low attention and the presence of water contamination is largely neglected. While the war still ongoing in some parts of the region, a partial peace agreement between the government of Sudan and the liberation and justice movement highlighted supporting IDP's return to the safe places of origin as core objective. As a result many initiatives have formed to contribute to this objective.

3. Peace and resolution program

The Darfur Regional Government (DRA) is coordinating the implementation of a return program co-lead by humanitarian partners (UNHCR and D. mission, 2012). As a result, 8 initiatives were formed to facilitate and monitor return adequacy. The initiatives are working platforms launched by the different governmental bodies on the state level and the humanitarian partners (Ibid). Although the government institution are taking the lead of the projects implementation, the international community has the bigger stake of the project financing and monitoring mainly via Darfur community and a stability fund.

The donor conference promised US$3.6 billion for early recovery activities in Darfur. The estimated January to December 2013 fund was $94 per beneficiary (UN, 2013). The fund is divided into different objectives; food security accounts for 51% of the total donation, other objectives such water sanitation and hygiene, return and reintegration support. These objectives only gained a small fraction of the fund granted. The provision of the service in the IDP’s camp is regarded as priority objective (Ibid). Additionally, the main activities for all projects rely on non-tangible outputs. For instance, the UK Trust Fund reported activities for 2011-2012 programs were mainly empowerment workshops for communities, awareness campaigns (eg. hygiene, gender equity), conflict resolutions, etc. Only a few tangible activities of building and rehabilitation of 26 school and 80 water sources account for a total expenditure of £900,000. The annum evaluation report highlights that the outputs did not meet 3/5 substantially and 2/5 moderately the planed expectations (DFID Sudan, 2013).

4. The challenges

The major difficulties identified by the humanitarian agencies regarding projects implementation and monitoring of fund were very much the same (DFID Sudan, 2013; UNHCR and D. mission, 2012; IOM, 2014):

- The vulnerability of the resource –water- and provision of basic service (education, and health) remain a major threat for return sustainability.
- The handover of projects remained a big challenge; the government and community abilities to operate the project is questioned.
- Reporting of activities by partners (the governmental bodies) is very weak, no evidence of project implementations in some activities and the driving cost for projects is mostly unknown.
- Conflicts of interest between different stakeholders are not fully defined.

5. The current water, sanitation condition

The access to improved water and sanitation in the IDPs camps is estimated by 90% and 32% in the rural areas, see camp coverage figure C (WES, 2013). Ground water is directly supplied with addition of chlorine residuals. However, water camps ground water reveals high levels of pollution intermittently. The topic is barely discussed due to the emergency and war context in the region; such topic is regarded as a trigger of conflict between civilians/IDPs and the government. The research in ground water contamination is considered as a threat by the local authorities, thus the water quality research rather scarce. The only source which addresses water quality in great detail -(Abdelrahman and Eltahir, 2010)- reveals high inconsistency in the final results presentation. However it confirms the presence of fecal contamination in all water sources examined in 7 camps.
The design of the latrine is unimproved pits, the pit sized according to WHO guideline (WES, 2013). However, since the pit latrine is constructed by the household as community contribution, there is no full confirmation of meeting the specification. The general practice is to abandon the pit after being filled and to move the latrine elsewhere in the house. The ground water varies from shallow alluvial aquifer and a complex basement of around 2-55m water depth to deep a sandstone aquifer of 35-110m water depth; the complex basement is the dominant aquifer (UNEP, 2007). Accordingly it is believed that the water sources in the region is highly contaminated.

6. The pit latrine assessment

Assuming the pit latrines are hygienically safe is faulty, unless the sludge is gathered, collected and treated for reuse or disposal. Although the WHO guideline considers the effect of leaching, contaminates are very low if the water table is deep enough to allow sufficient travel distance of at least 2 meter below the pit; permitting acceptable level of microorganism deactivation. However, the microorganism deactivation, filtration of micropollutants and organic matters rely on the soil properties (eg. Grain shape, diameter) and geological conditions (eg. uniformity coefficient). Graham and Polizzotto (2013) review paper confirms the pit latrines (VIP; with lining walls and without) impact on ground water contamination is underestimated and recommendations for reducing the effect of leached contaminants are greatly variable between different sites. Furthermore, the study shows shallow ground water is contaminated in all sites where pit latrines are extensively applied (eg. eastern Botswana, Zimbabwe, Siberia and Kosova) regardless whether the guidelines have been strictly or vaguely followed.

7. The proposed solution

The onsite conventional wastewater treatment systems are probably the most convenient solution for water reuse in the IDP camps or the resettlement villages which are considered non-permanent settlement either due the building material or government legislations. As well as, the onsite -household level- treatment system will rely on individual interest on implementing a mean for providing food security for their family; the grey water generated per household could be roughly estimated by 0.122 m3/day 70% out of consumption which sufficient to irrigate 13.5 m2 of maize (FAO, 1986). Furthermore, the nutrient recovery would be an attractive choice for the target group which are already familiar with fertilizers (organic and chemical) to increase crop production, nevertheless the challenge is to maintain constant quality of fertilizers. In the literature they are several
studies identifies the efficiency of compost and urine for growing specific crops. For example the (Sangare, and Funamizu, 2014) study defines the optimum ratios of urine and compost pile for Okra cultivation and provides a practical guidance for using human waste as natural fertilizer in okra production. Such systems are not just increasing sustainability of projects by ensuring physical existence of the target group in the settlement area, as well it solve vital problem of water scarcity specially in dry season 9 month, and strengthen the food security within targeted community.

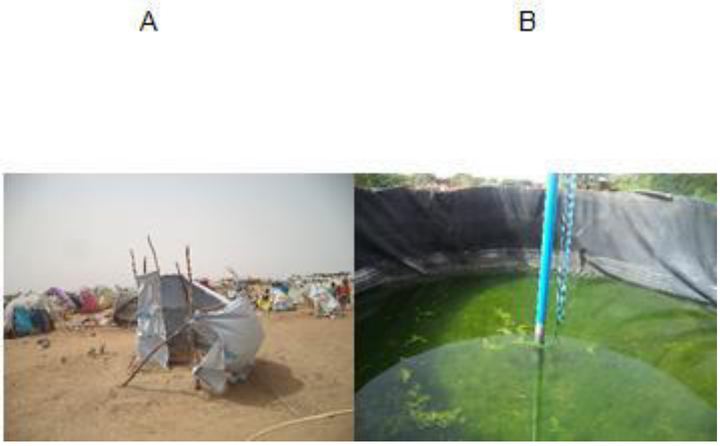

Figure 1: A: Pit latrine and B: Storage tank Alslam camp funded by Unicef.

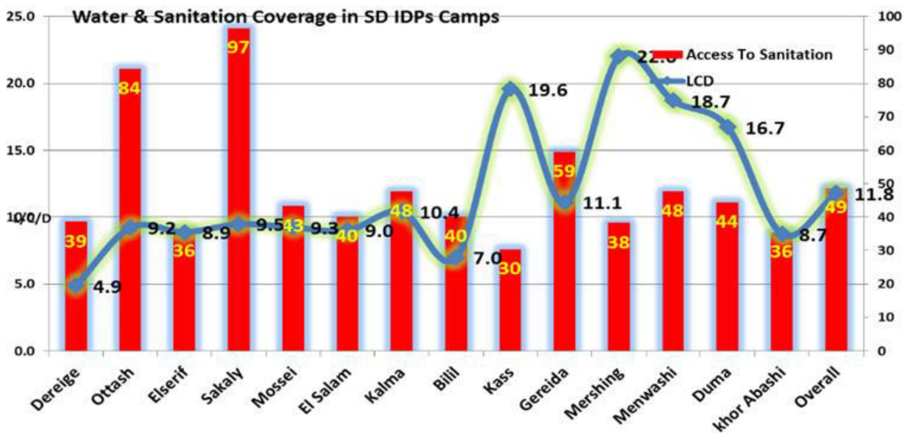

Figure 2: The water and sanitation coverage South Dafur State IDP's Camp 2013.

8. Onsite treatment

Furthermore, they are several techniques which are developed for safe onsite treatment in developing countries such as urine diverting and mixed toilets with compositing in order to produce fertilizers (Katukiza, and Lens, 2012). Such devices are installed by NGOs, government institutions, and individuals in several peri-urban and rural areas, many of them show a good performance and sustainable use but only few are assessed to come with practical correlations between different factors affecting compost process and pathogens inactivation. Thus they remain very site specific and the quality of the final product depends on the external environment in term of moisture content, temperature, rain intensity and quality of the influent.

On the other hand, grey water treatment has being applied in slums using fed soil or sand filters; well graded with uniformity coefficient smaller than 5, the effluent used for irrigation purposes(Ibid). the technology is very cheap though it depends on the availability of raw material for infrastructure, examples of such systems are Mulch towers in South Africa (Zuma et al., 2009), grey water towers with nutrient recovery in East Africa (Kulabako et al., 2009). The major problems associated with these systems are breakthrough of contaminates, varying compositions of grey water, higher risk of diseases associated with raw vegetable consumption (Siobhan et al., 2006). Less common, more robust systems have been applied on household level such as, the horizontal flow trench system, intermittent sand filters, and the crate soakaway (Katukiza and Lens, 2012).

9. Obstacles facing onsite treatment

The public or the target group compliance with the use guidance seem to be a major threat, since every household is responsible for operating and maintaining such systems, thus the final product quality remains uncertain either due to poor understanding of the operating process or underestimation of the health risk associated with non-advisable operating procedures. For example the Viet-

namese farmers use the compost after 3-4 months instead of 6 months – the recommended time for safe use of pile compost- because it is the only time available for production of composite between two seasons without giving much concern about whether this period of time is enough for producing hygienically acceptable final product (Jensen, and Dalsgaard, 2009). However, this give an indication that the problem is quite complex is not just about implementing the right technology which can cope with the technological level, economic and social context, the designer of such system should realize why and how to make the beneficiaries interested following their guidelines or either design system which can fulfil the beneficiaries requirements on the first place.
On other the hand, the researcher/executor full understanding of the site context for system selection and implementation. In Malawi experience on implementing composting toilets, the hygiene level of the final product remain minor threat due to the climate effect since the temperature is high and the open space of the farm land allows quick dewatering of the compost (Morgan, and Mekonnen, 2013). Although, this observation advocate that the desiccation process was governing the composting, the mixed pit latrine (no urine separation) had the higher implementation rate. In contrast to Vietnam experience where farmers were the major threat for latrines without nutrient recovery; several projects were failed due to damage of latrines in order to recover the compost (Jensen, and Dalsgaard, 2009). The Malawi experience shows farmers were not in favour of using composite nutrient apart from those who are very poor and mainly construct composite toilets as aid for food security.
Further challenges were the beneficiary dissatisfaction of the system final design will remain major barrier for further implementations and the affordability of the system is questioned. The Eco-San system major scepticism was concerning the design of the latrine; the depth of the latrine was too shallow compare to the traditionally used pit latrines creates negative impact of Eco-San *acceptability*. Similar case challenged Eco-San in Zimbabwe concerning the volume of the latrine 1.17 m3 is considered small by the user and modification have been made to 0.282 m3 and evaluated by other researchers but recommendation for adopting the modified design was never reality (Morgan, and Mekonnen, 2013). On the other hand, the Malawian Eco-San affordability is based on providing subsides and loans to the beneficiaries which failed to provide equal treatment for the stakeholders, thus it has a high negative impact on the implementation rate.
Many researchers emphasis the need for context specific solutions wastewater treatment system but meaning mainly technical applicability, social acceptability, socio-economic aspects (affordability, and running cost), but hardly any look in depth on the technology transition phase after implementation or discuss the manner of integration of the new technology as part of the common practice in the site. Although, this might be the only option to maintain the existence of such system and for the technology to develop carrying the site context and following the objective which such is technology made for.

10. Conclusion and recommendations

The Darfur situation of accelerating conflicts around vulnerable resources, the international aid regarded as major support to the government for increasing livelihood conditions in order to reach peace and resolution in the region. The result of the programs activities are mostly intangible due to the nature of activities implemented is very general, the lack of transparency between government and partners agencies, high corruption level in the government institutions. Thus, from the region political and economic context it is clear that the micro projects with tangible outputs are needed especially if it has a direct impact in reducing water scarcity and supporting food security. The proposed onsite treatment targets beneficiaries directly and shift their role from war victims waiting for aid to productive citizens. The onsite treatment could be a very good solution for reducing ground water pollution and increasing peace. The technology is well researched in term of hygienically safe, applicability, functionality and the results are so far acceptable. Nevertheless, it can be clearly seen from previous experiences of such systems following bottom-up approach in reflecting the target group opinions on the final system design and make the target group realize the beneficial impacts that will gain by implementing onsite treatment in their houses. Along with, reducing uncertainties regarding continues implementation are key success parameters for shifting the technology from imposed solution to popular and common application at site.

11. Key parameters for suggested research

- Bottom-up approach; investigate people consideration around the application and carry their concerns to the design phase, and reduce the sense of inferiority by adoption and modification of indigenous or favored deign by the community.
- Technology transition; consider the technological level of the place eg. The technology exists on the surrounding markets, technician level of education.
- Find entrepreneur; study the availability of local prospect beneficiaries out of selling, constructing, and provide maintenance for the application

- Reduce uncertainty by relying only on beneficiaries for the system continues implementation, and optimize the capital cost of the system according to the beneficiaries' economic level without considering any subsidies.
- Ensure that crucial needs and benefits of the system are very clear for the target group, and consider high involvement of the community in system development.

References

[1] Abdelrahman, A.A. & Eltahir, Y.M., 2011. Bacteriological quality of drinking water in Nyala, South Darfur, Sudan. Environmental Mosnitoring and Assessment, 175(1-4), pp.37–43.

[2] DFID Sudan Darfur Community Peace and Stability Fund (DCPSF II), 2013, Annual review Project ID: 202302, http://devtracker.dfid.gov.uk/projects/GB-1-202302/documents/

[3] FAO, 1986 Irrigation Water Management: Irrigation Water Needs CHAPTER 2: CROP WATER NEEDS. Available at: http://www.fao.org/docrep/s2022e/s2022e02.htm [Accessed May 20, 2015].

[4] Graham JP, Polizzotto ML. 2013. Pit Latrines and Their Impacts on Groundwater Quality: A Systematic Review. Environ Health Perspect 121:521–530; http://dx.doi.org/10.1289/ehp.1206028.

[5] IOM Sudan Mid-Year Report January – June 2014. (n.d.). Retrieved May 15, 2014, from http://www.iom.int/files/live/sites/iom/files/Country/docs/IOM-Sudan-Mid-Year-Report-2014.pdf.

[6] Jensen, P.K.M., Phuc, P.D., Konradsen, F., Klank, L.T. & Dalsgaard, A. 2009, "Survival of Ascaris eggs and hygienic quality of human excreta in Vietnamese composting latrines", Environmental health : a global access science source, vol. 8, no. 1, pp. 57-57.

[7] Jeremy and Alan, (2012), Programming for secure livelihoods amid uncertainty: trends and directions in livelihoods and food security in Darfur. Available at: http://www.fao.org/fileadmin/user_upload/emergencies/docs/Darfur_Livelihoods_Study_2012.pdf [Accessed May 20, 2015]

[8] Katukiza, A.Y., Ronteltap, M., Niwagaba, C.B., Foppen, J.W.A., Kansiime, F. & Lens, P.N.L. 2012, "Sustainable sanitation technology options for urban slums", Biotechnology Advances, vol. 30, no. 5, pp. 964.

[9] Kulabako, R., J. Kinobe, J. Mujunga, S. Olwenyi, K. Sleytr. (2009). Greywater use in peri-urban households in Kitgum, Uganda. Retrieved September 27, 2010 from, http://www.ecosan.at/ssp/issue-01/issue-1-1

[10] Morgan, P. & Mekonnen, A.T., 2013. Paving the way to scaling-up: Factors contributing to the adoption of Eco-San toilets and safety of humanure in Malawi. , p.27 pp. Available at: http://r4d.dfid.gov.uk/Output/192824/Default.aspx [Accessed May 20, 2015]

[11] United Nations, 2013. Mid-Year Review of the UN and Partners Work Plan for Sudan 2013. Available at: https://docs.unocha.org/sites/dms/CAP/MYR_2013_Sudan_Workplan.pdf [Accessed May 20, 2015].

[12] UNEP, 2007. Darfur: water supply in a vulnerable environment. Available at: http://www.unep.org/disastersandconflicts/portals/155/countries/Sudan/pdf/dafur_water/Darfur Water Resources TF.pdf [Accessed June 1, 2015].

[13] UNHCR, and D.D.J.A.M., 2012. Return, Reintegration, and Urbanisation Situational Analysis. Available at: http://www.darfurconference.com/sites/default/files/files/5 - Thematic Working Group Returns, Reintegration and Urbanisation Report%281%29.pdf [Accessed May 22, 2015].

[14] Sangare, D. et al., 2014. Toilet compost and human urine used in agriculture: fertilizer value assessment and effect on cultivated soil properties. Environmental Technology, 36(10), pp.1291–1298. Available at: http://dx.doi.org/10.1080/09593330.2014.984774.

[15] WES; Water Environment and Sanitation program South Darfur, 2013, camp profile report September 2013. Nyala. Zuma BM, Tandlich R, Whittington-Jones KJ, Burgess JE, 2009, Mulch tower treatment system; part I: overall performance in grey water treatment. Desalination; 242:38–56.

TE-P_02 Sustainable urban rural proposal in Malinalco, Mexico – Case study

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1. Objective

Design a Rural Development – Sustainable Urban composed of six small businesses that use appropriate technologies and human and natural resources of the region.

2. Comments

Malinalco is a rural village with a population of about 10,000 inhabitants. It is located 110 km. by road from Mexico City. It has a semi-tropical climate that ranges between 28 and 32 ° C. There is a monolithic pyramid was ceremonial center of the ancient Aztecs.

The main activity is tourism during weekends. There is a particular thriving business for breeding and sale of trout, also 20 bakeries that use wood as fuel. About 20 years ago some families had coffee trees for their own consumption. The nearest place coffee producer is 400km. from México City.

3. Description of the case study

The proposal is to train people who want to participate in the Project, so they can exploit the natural resources of the region using Appropriate Technology for their welfare, improving their diets and family income. The by products can be used also.

It is proposed to build two greenhouses, one for growing vegetables and another one for flowers. In the first one the space can be maximize up the floor, placing several trays, leaving space between them for different varieties as lettuce, cabbage and tomatoes. Products can be used by producers to improve their diets and also market them to have surplus income.

Figure 1: The other greenhouse for growing flowers which can be marketed.

Figure 2: In an area next to the greenhouses they can grow carrots, potatoes and peas.

The byproducts such as stems and leaves mix with other ingredients can be used to make balanced food for chickens and rabbits.

It also intends to build three galleys with local materials such as stone and adobe.

One of the galleys used for breeding hens for eggs and also to sell them and to have income improving health and economical situation.

Another galley for fattening chickens and they can eat that meat and over commercialization.

The third gallery breeding rabbits and they and their families can improve their diet with this meat and sell what they do not consume as well as the skins.

The balanced feed for these two species is partly what it is not used for human consumption of cultivated vegetables as well as heads and chickens feathers with other ingredients of the region.

The feces of chickens and rabbits can be used to make organic fertilizer for vegetables.

Another small business can also build with local materials a pond to grow tilapia that is a fish that does not require special conditions for their growth as trout. The meat produced can be consume by the workers and sell the surplus.

Heads, skeletons and queues can be utilized as inputs for balanced feed.

Also take advantage of conditions in the region to grow and process coffee, using the dry process that does not require as much water as wet way. Solar energy can be used for drying coffee beans. This will be another income they had to sell coffee.

Lemon, melon and grapes are fruits that can be grown for consumption and sale.
It can be captured rainwater in a cistern to use this natural resource.

The tomato is an easy vegetable grown in the greenhouse. Its seeds contain some protein so by food consuming improve their health because also help the prostate of men. This product can be combined with other like potatoes to make nutritious biscuits for consumption and also sold for income.

It is possible to build an anaerobic digester to generate methane gas with organic waste and use it as fuel instead of wood.
These six technical-economic activities may be located in one spot so that there is interaction among workers.

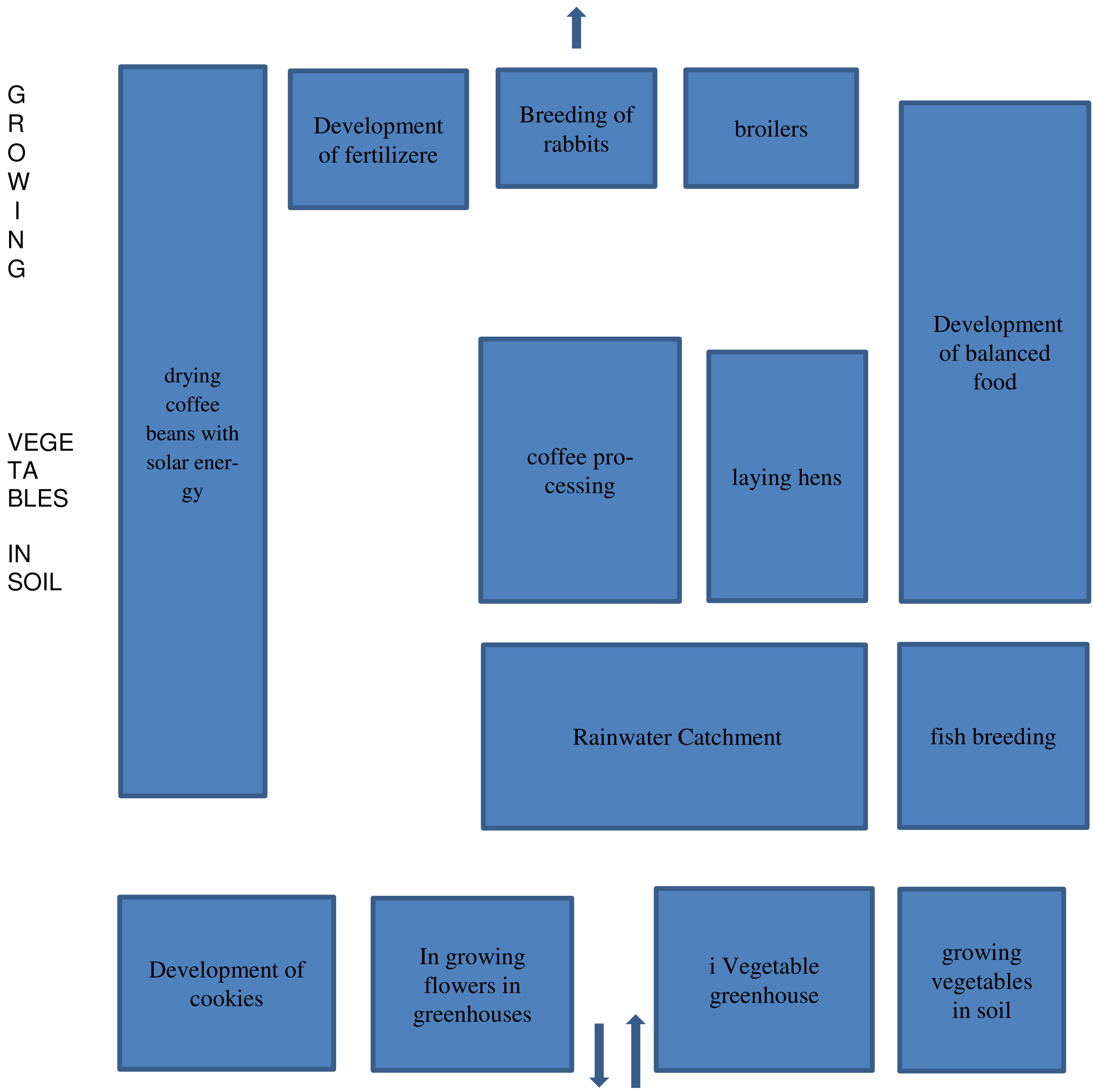

Figure 3: Sketch project distribution.

4. Conclusion

It can be possible to create a cooperative with the working groups in each of the six technical-economic proposals activities:

1. A group for the greenhouse and vegetable growing in the ground and one for the greenhouse of flowers.
2. Another group for breeding and fattening of chickens and selling. One more for chicken breeding selling position and tilefish.
3. A further group for breeding and selling rabbits.
4. Another for growing and sale of tilapia.
5. One more for the cultivation and processing of coffee.
6. And the last for the production and sale of nutritional biscuits.

Participants in these activities people can learn Technologies not be dependent on these or others, as well as improving their diet and family economy.
This proposal can be achieved by rural communities improving their feeding, health and ecosystems and environmental sustainability.

The number of chickens, rabbits and fish depend on the number of people wishing to participate in this project.

References

[1] R. Castañeda analysis and simulation of a physical model of the greenhouse effect.Agro Science vol.41, No. 3, 2007.
[2] National Commission of Natural Protected Areas. 2008, Mexico, DF
[3] Biodiversitas No.110, CONABIO, 2013 ISSN: 1870-1760. Mexico DF.
[4] Ecosystems and Human Well-Being: Synthesis. Island Press, Washington, D.C.USA, 2005.

TE-P_03 Systemic approach applied to a Mexican rural area, in order to improve the quality of life and economic well-being of people

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1. Objectives

How could a Systemic Design project support and encourage sustainable rural development?
Can we, as designers, pull for the development of agro-food systems, contributing to the sustainable, inclusive and economic growth in rural regions?
Is it possible to promote economic diversification combining traditional agricultural skills and new technical and technological know-how?
Which strategies can be implemented to increase sustainable food production, to ensure food security regionally, to improve human health, and to reduce poverty?
How the Systemic Design can be the right approach to encourage social innovation processes for the improvement of the quality of life and the economic wellbeing of people in rural areas?

2. Methodology

This project follows the desk and field research design methods [1]. The combination of desk and field methods guarantees a rich and deep understanding of the facts to define an original framework. The desk phase is composed by the group of knowledge produced in the past on the same issue and with similar approaches. It is required in order to meet necessary background to deal with the theme of the project. The field phase is about the direct observation of the reality and of the case-study. Despite it takes a lot of time and it is crucial for the project, cannot be said that it is more relevant than the first one. The success of the project comes precisely from the ability of the researcher/designer to coordinate these two phases and make the right contaminations to proceed in a parallel and alternatively way. The desk phase without that field one is sterile; the field phase without that desk one is weak.
That methodology aims to turn the theoretical knowledge into pathways of change suitable to the needs and specific capacities of regions or localities, and to define the right level of replicability.

3. Case-study description

To answer at all the initial questions we applied the Systemic Design approach (SDA) [2] at the rural area of Ahuacuotzingo (Mexico, State of Guerrero). This approach allows us to have a clear definition of the steps to plans open systems strongly connected with the territory. The place was chosen because of its particular features related to food, both the production and the consumption. Take action on these aspects means managing environmental, social, economic and health consequences. The area is characterized by low population and enterprises density, high unemployment and emigration, especially to the United States. Since 1980, Mexicans have been the largest immigrant group in the United States. In 2013, approximately 11.6 million Mexican immigrants resided in the United States (up from 2.2 million in 1980) and they accounted for the 28% of the country's foreign born (41.3 million) [3]. This situation generates a radical change in lifestyles, food consumption, and a loss in material culture, because people try to imitate other cultures losing totally its own local and traditional know-how. In recent years, someone seeks to improve the quality of life and well-being returning to farm the land in their own hometown. The population of this rural area, rather isolated, reveals to be intimately and intensely linked to the territory and to have a strong sense of belonging and aggregation. In addition, the farmers of the cooperative Ahuehuetla, with which we are working, are very motivated for a substantial changing towards sustainable rural development.
Apply the Systemic Design Approach is a complex process consisting of several stages, managing a lot of variables and it implies the active participation of all actors involved. This means that the project needs many years to be implemented, but even if it is not finished yet and complete, the first results can be experienced in the context.

4. Research steps and preliminary outcomes

The SDA generates benefits for the whole community: from a better environmental quality, to the creation of new job placements. The first guideline of this approach is that the waste (output) of a system must become resources (input) of another system; in this way the system generates for itself resources, contents and meanings, by updating and developing independently [2].

In this project, SDA is applied to the activities of five farmers in Ahuacuotzingo who want to be able to improve their quality of life, starting from the resources of their own territory.

The project consists of three phases:

- The first stage is the analysis of the territory,–called the *Holistic State of the Art*. It does not just research geographical and territorial aspects, but also characteristics related to the cultural and social aspects. The Holistic State of the Art, in that specific project, was the deep analysis of the activities of these five farmers in order to identify inputs, outputs, and processes in terms of quantity and quality. This initial analysis is necessary to define strengthens and weaknesses of the activities at the actual situation, in order to figure out what kind of actions are needed to improve the entire system. From this analysis, we noticed that the critical aspects are focused on food, both at the production and consumption level.

  Preliminary outcomes: At this stage of analysis, we defined the amount of input and output and their characteristics.

  It is important to underline that the activities of the five farmers from which we started are family run at micro-dimension with the simple and essential goal of self-subsistence. The productions are often managed in a very random and disorganized way. For this reason, the data from which we started the following phases are subject to many variations.

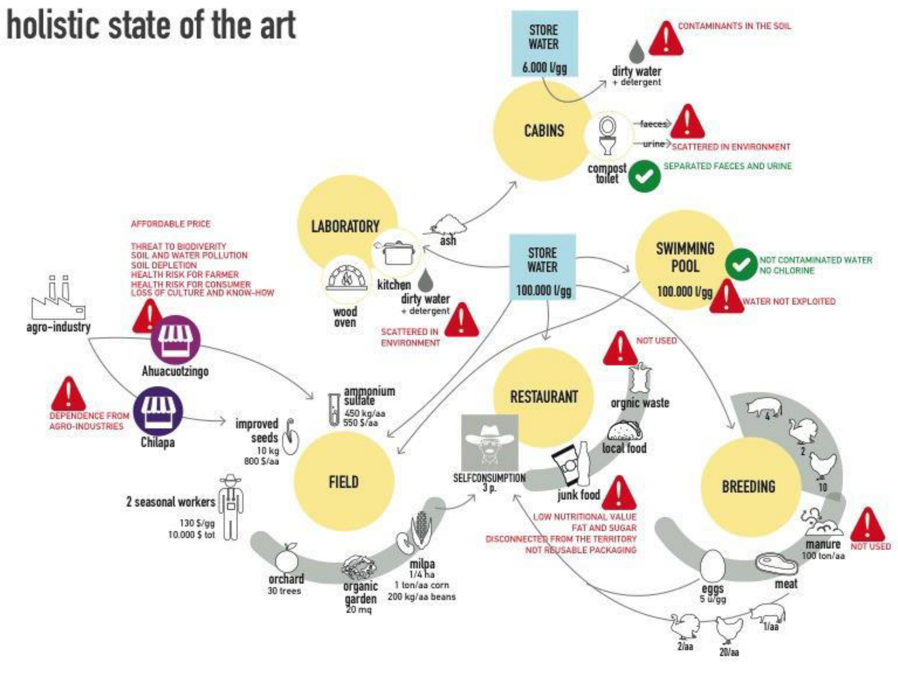

Figure 1: First phase, the Holistic State of the Art of Ahuacuotzingo (Cavideco community space).

- The second phase is the development of the *Local Agricultural Cooperative* (LAC) composed by five farmers, that are involved from the beginning, and by two other interesting realities on the territory: a group of five women producing panela (a typical local product, a sweetener resulted from the transformation of sugar cane juice) and another one of women who own and manage an organic greenhouse [4]. The aim of the LAC, called Ahuehuetla, is to cooperate and collaborate on the basis of some shared values to address common challenges and provide mutual benefits.

  Preliminary outcomes: The LAC seems to be the ideal framework for the implementation of a Systemic Design project. The goal is to work with local resources, and show that this rural area is not poor, redefining and designing new flows of matter and energy. In this way, the farmers could become stronger under different perspectives: the support in investments that alone could never face; the sharing of equipment, tools and spaces.

Furthermore, there are also intangible aspects that are very important: they are supporting each other, sharing problems and solutions. The creation of the Ahuehuetla Cooperative also allows the farmers broadens their perspectives: if currently they are limited to produce what is enough for their own outliving and obtain a minimum gain; all together they can act with a vision closer to a small-micro enterprise [5]. This perspective starts from the increasing of production, because, right now, farmers grow only a small part of the land they have, despite the availability of water and staff. In this way it is possible to achieve quantitative results: the increased number of products, and the new-born relationships among the farmers themselves, the consumers, and the locals.

- The third step is the *designing of the flows* according to the intention of tending to zero waste and emissions. The objective is to reach the satisfaction of the agro-food demand of the people that live in Ahuacuotzingo, by changing the production but also the dietary habits. The project aims to produce healthy, local and clean food, linked to the rural Mexican tradition. The educational and social aspects are equally important: the Ahuehuetla Cooperative owns a community space (Cavideco), where it is possible to transform and sell food, as well as to be a meeting point for seminars and workshops. It could be the link between the Cooperative and the local population, so that both can benefit from the systemic project.

  The comparison between the current diet and a proper and balanced diet figures out how many people could be satisfied with the cultivation of the land in the cooperative. One of the goals of this project is to improve the health of the inhabitants in this rural area, because actually they have a high percentage of cardiovascular diseases mainly due to improper diet.

  The Cooperative has been designed as if it is a living being; each part has its function, chosen according to the geography and the resources, and it contributes to the success of the whole system.

  Preliminary outcomes: The conversion of the waste of the area into resources generates many new products and related production activities, such as fish farming, compost and worm compost, worm farming, seeds self-production, natural filtering of water, biogas, bio detergents and natural soaps, panela (which was the typical and most important production of this area until it has been replaced by the refined white sugar), honey. In order to guarantee the right dimension in production is necessary to resume land and activities now almost totally abandoned. Sugar cane, for example, now is used almost exclusively for animal feed, but this is a serious loss because even from sugar cane is easy to obtain many products, such as panela, juice, vinegar, alcohol, but also interesting outputs, such as bagasse, cachaza, vinaza, ...

5. Conclusion

Expected results regard social, economic, environmental and health aspects. The farmers of the cooperative Ahuehuetla can become not only self-sufficient in terms of energy and food production, improving the quality of life, but also increasing the supply of food products, both unprocessed and processed.

Within the project of Ahuehuetla Cooperative, we also dealt with the design of the logo, which is the central visual element that helps to identify and remember the brand. As happen in every company, this icon is a real symbol: the main purpose of it is to summarize and underline shared concepts and values from farmers. It is not only a graphic action but it is a way to define and promote a strong and precisely identity for the farmers and the entire community.

During the Field research some important needs from the farmers came up: for example, they need to have counseling and contacts with experts and technicians in the field of organic farming and appropriate technology [6]. This is why we designed and managed also the educational part of the project, mainly divided into two parts. On one side, some practical courses held by technicians and addressed to farmers can be organised in Ahuacuotzingo at Cavideco. On the other side, Systemic Design seminars addressed primarily to university students can be organized, with the purpose of developing the competences in designing and organizing flows in a system. This is the reason why we are working together with Nuria Costa Leonardo, one of the 1000 women proposed for the Nobel Peace Price 2005, involved in different projects with the Red Mexicana de Mujeres (rememur) on socially responsible business related to rural development. This also helps us to relate to a distant reality from ours, though often so similar to many other rural areas all around the world.

One of the most important results was the definition of the necessary skills to make up the team working in similar rural development project. It is important to define the skills needed both during the design, and especially during the implementation phase, because this means to bring together engineers, experienced technicians on organic farming and integrated pest managers, systemic designers, farmers, producers and consumers.

Another important result is the definition of pathways and frameworks that lies under the systemic project. This is what makes it scalable and replicable.

As mentioned above, the starting data, defined during the preliminary phases of analysis of the Holistic State of the Art, are subject to different variations. It depends on many factors, especially as the impossibility of farmers themselves to quantify their own inputs and outputs. This limit turns out to be insignificant precisely because the main objective is not the quantitative result, but the identification of a framework that can be reused and exported to other rural realities in order to foster sustainable development.

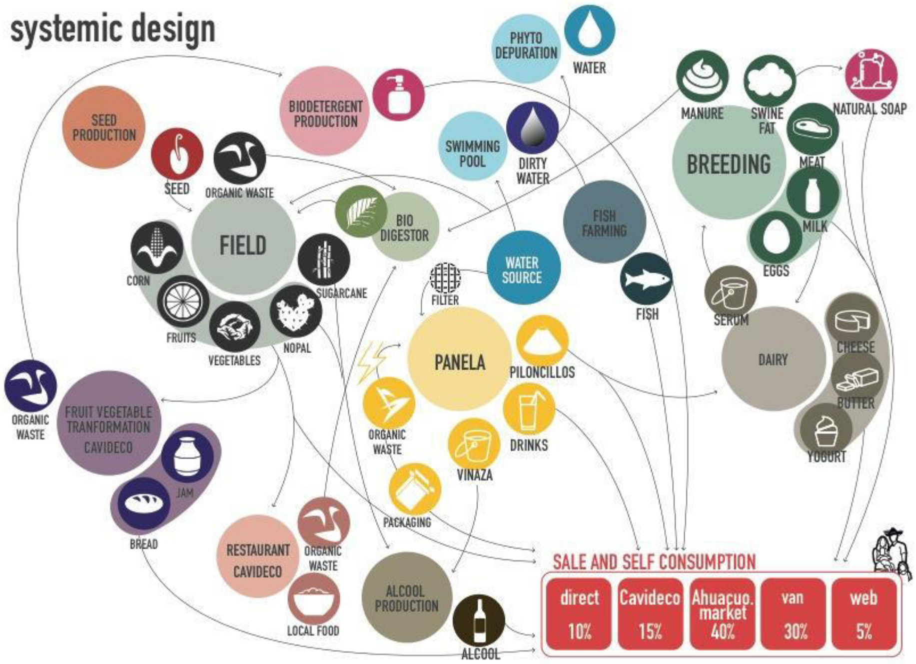

Figure 2: The third phase, Systemic Design Approach applied to the Cooperative Ahuehuetla in Ahuacuotzingo..

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References

[1] Celaschi, F., and Deserti, A. (2007) Design e Innovazione. Strumenti e Pratiche per la Ricerca Applicata. Roma:Carocci Editore

[2] Bistagnino, L. (2011): Systemic Design. Designing productive and environmental sustainability. Bra, Slow Food Editore

[3] *Source*: MPI tabulation of data from the U.S. Census Bureau 2013 ACS. www.census.gov/population/www/documentation/twps0029/twps0029.html

[4] Zahedi, A.; Otterpohl, R. (2015): Towards Sustainable Development by Creation of Green Social Entrepreneur's Communities. Procedia CIRP,Volume 26, 196–201 (12th Global Conference on Sustainable Manufacturing – Emerging Potentials)

[5] Bistagnino, L. (2014): MicroMACRO. Micro relazioni come rete vitale del sistema economico e produttivo. Milano, Edizioni ambiente

[6] Pauli, G. (2010): Blue Economy. 10 years, 100 innovations, 100 milion Jobs, Taos, Paradigm Pubns

TE-P_04 Unemployed young people and periurban agriculture – Case of Madrid

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1. Introduction

Agriculture lost importance around the urban areas some decades ago. Periurban family farming has been one of the main victims. Since 2007, Spain is immersed in a financial and socio-economic crisis that has also created some opportunities to preserve nature (Santos-Martín et al. 2013) and to reconsider and revalue its relevance related to our quality of life. In this way, unemployed young people are considering other options. These circumstances might provide a range of opportunities to promote agriculture as a professional choice for the young population in Madrid´s periurban municipalities.

With this purpose, the main goal of this research is to systematize and to analyse young people´s discourses about agriculture in Madrid´s periurban area. In order to achieve this, we have designed two specific goals: (1) a characterization of social perceptions on agriculture based on its capacity to change the socio-economic situation on this area, (2) an identification of the main motivations and barriers to promote entrepreneurship on this sector with a special emphasis on young and unemployed inhabitants.

The studied area selected consisted of a flood plain located in the south-east of Madrid Province, involving three main rivers (Tajo, Tajuña and Jarama). The area covers 1500 km^2 (19% of Madrid region surface) and 21 municipalities, characterized by one of the most fertile valleys of the region. Nineteen of these municipalities have more than 5000 inhabitants, while six have less than 1000 inhabitants; to sum up the population consist on 151.179 inhabitants (Table 1). Its agrarian character has remarked its culture, nevertheless the activity in these days remains anecdotic. In addition, the unemployment rate is 12%, higher than the average rate in this region (10%). The average distance to Madrid city is 51 km, which guarantee a good connection among rural and urban areas. All these characteristics made this area suitable for the study developed.

2. Methods

The data sampling has included different sources of information: official databases to collect socio-demographic information about each municipality, semi-structured interviews, and focus groups.

On the whole, 21 interviews were conducted with key informants, including: farmers (n=10), local government staff (politicians and managers; n=6), as well as businessmen (n=5). The interview was structured in different sections related to: current perception of the rural population in the region, current perception of the farming activity in the region, policies and public/private support of the agricultural sector, entrepreneurship vocation and professional skills and training.

After that, 6 focus groups were created from unemployed young people (younger than 40 years old) and young farmers stablished in the region. Structural sample considered agrarian family tradition, sex and age variables. The sampling was conducted during the period 2013-2014.

3. Results and discussion

From decades ago agriculture has been losing importance (Guzman et al. 2013). Young people, women especially, have moved from rural areas to urban areas looking for better opportunities.

Table 1: Municipalities' data.

Municipalities	Inhabitants	Sex ratio	% employment in agriculture 2001	% employment in agriculture 2010	2001- 2010
Ambite	591	102	9.86	17.65	7.79
Aranjuez	56.877	95	1.22	1.7	0.48
Belmonte de Tajo	1.580	105	5.32	8.93	3.61
Brea de Tajo	572	132	8.18	8.8	0.62
Carabaña	2.052	102	2.85	3.06	0.21
Chinchón	5.404	103	14.44	16.73	2.29
Ciempozuelos	23.716	101	1.02	1.1	0.08
Colmenar de Oreja	8.432	105	5.04	8.06	3.02
Estremera	1.486	103	5.96	5.32	-0.64
Fuentidueña de Tajo	2.077	104	9.95	13.45	3.5
Morata de Tajuña	7.515	102	3.99	6.19	2.2
Orusco de Tajuña	1.300	102	1.95	2.02	0.07
Perales de Tajuña	2.877	103	0.85	1.65	0.8
San Martín de la Vega	19.615	109	1.59	2.16	0.57
Tielmes	2.616	106	1.67	3.26	1.59
Titulcia	1.206	108	8.42	8.21	-0.21
Valdaracete	665	102	22.14	32.89	10.75
Valdelaguna	863	102	18.54	26.51	7.97
Villaconejos	3.488	104	14.08	23.85	9.77
Villamanrique de Tajo	795	109	22.37	19.88	-2.49
Villarejo de Salvanés	7.452	103	2.44	3.98	1.54

The mindset that rural parents have tradicionally translated to their children was: you have to study and move forward to Madrid instead of working as a farmer, search for a job in an office or in a factory. However, the situation in periurban areas was different. Young people who lived in Madrid periurban area could study in Madrid and come back every day at home; maintaining a higher link with their hometown. These areas didn't lose population but people were not interested in conducting farming activities.
In the next lines we are going to show some of the main verbatim reported by the participants:

Verbatim 1. Interview	What did a family used to say to the son? Look for life stability and security, study computer engineering and establish yourself in the city. What did a family used to say to the daughter? Go away as fast as possible. Nowadays, female participation in agriculture is just anecdotic.
Verbatim 2. Interview	It is normal that young people wanted to study, go to university and leave the farm… We have tried to provide an education for our children in order to avoid them working at the farm.
Verbatim 3. Focusgroup	I want my children to study and to have the freedom to choose and decide what they want to do. If, in the future, they are not able to find anything better, they still could work here.
Verbatim 4. Interview	When I said "My father is a farmer" people told me "you must study, in order to change your occupation".

The current economic, financial and social crisis is changing these discourses. People are **lo**oking for a job and have changed their perception about agriculture and farming; seeing both as a source of new chances. Madrid is a huge market with a wide range of diverse consumers, who buy agrarian products from Spain; looking for different characteristics such as food quality products, environmental friendly or social justice. As held by previous authors (i.e. Iniesta-Arandia et al. 2014) a generational change is emerging, and the motivations of young people towards farming activities are increasing together with an interest **in** organic agriculture, agroecology and conservation principles. Nevertheless, Madrid is surrounded by neglected meadows.

Verbatim 5. Interview Nowadays young people want to work in the countryside. They want to run a farm.

But, what kind of barriers and stimuli do they find? Firstly they talk about economic barriers (initial investment, access to land, difficult negotiation with large distributors). Basically, they consider that agriculture is not profitable and is very hard work and demanding, facing similar problems as family farming around the world (FAO 2011).

Verbatim 6. Focus group If you want to undertake in agriculture, you must do a huge investment, otherwise it is impossible.

Verbatim 7. Focus group There is a problem, renting a land is very expensive. Nobody wants to rent their land. Some owners distrust and they do not rent their land. They prefer to leave their lands abandoned.

Verbatim 8. Interview Another issue to take into account is commercialization. I can grow tomatoes, but how can I get a fair price?.

In conclusion, the most relevant barrier identified in the study following their discourses is that the social perception of agriculture is very poor and farmers wish their children would not work in agriculture: “study and work outside agriculture”.

Verbatim 9. Interview Being a Farmer is an outcast in spanish society.

Current socioeconomic context is overthrowing this barrier. Some young unemployed are considering agriculture as a professional alternative. In fact, collective and local agroecological initiatives are increasing since 2011; with more than 30 agroecological networks across spanish municipalities (Llobera and Redondo 2014). Innovation and tradition are working together and we can find some entrepreneur in these meadows.

Verbatim 10. Focusgroup Nowadays you can not work as a builder and young people are considering being agricultures. But It is a bit sad because the main cause is that there is nothing else to work at.

They expressed new demands in terms of obtaining more information, training, as well as improving advisory services, and the initial public support. In the short term, they are demanding extension services. The need of more knowledge and innovation transfer has been suggested by recent studies in central Spain, in addition, it has been detected that young farmers are willing to change this practice (Marques et al. 2015).
A new kind of revitalizing extension services are necessary (Cristóvão et al. 2012). Madrid does not have a broad agriculture formal education offer or a professional training; you only can learn agriculture at the University level or doing a lot of short trainings.

Verbatim 11. Interview Agriculture education and training should be more structured. I mean, It is difficult for us to learn correctly about agriculture only attending to short courses.

Verbatim 12. Interview Theory is perfect but we need training. If you want to learn about agriculture you need to practise.

Periurban agriculture is emerging like a professional alternative in Madrid. The most important barrier is perception about agriculture and farming but socioeconomic context has changed this situation . At this moment, family farming is in a crucial shift point of change. The current socioeconomic context obliges to overcome psico-social barriers and revalue the land. Some young people, including female, are starting agrarian projects in the periurban areas where they live. Under these circumstances, new requirements emerge, such as the need of a new model on extension services, social innovation and nature-based solutions that enhance organic agriculture, short food supply chains, and a higher collaboration between different stakeholders connecting consumers and producers.

4. Conclusions

Some young unemployed are looking for their professional opportunities in agriculture. People are rediscovering, recovering and revaluing the agricultural tradition in the periurban area of Madrid. Financial and social crisis has overthrown the most important barrier: agriculture perception. Young people are demanding information, training, as well as the improvement of advisory services. Maintaining farmer family tradition is a key point for this entrepreneurial profile, since they could get knowledge and initial investment easily.

References

[1] Cristóvão A, Koutsouris A, Kügler M. 2012. Extension systems and change facilitation for agricultural and rural development. Farming Systems Research into the 21st Century: The New Dynamic. Darnhofer, I., Gibbon, D., Dedieu, B., Springer Netherlands, pp. 201–227.

[2] FAO. 2011. Save and grow. A policymakers's guide to the sustainable intensification of smallholder crop production. Rome, Italy.

[3] Guzmán G, López D, Román LL, Alonso AM. 2013. Participatory action research in agroecology: Building local organic food networks in Spain. Agroecology& Sustainable Food Systems 37, 127–146.

[4] Iniesta-Arandia I, Piñeiro C, Montes C, Martín-López B. 2014. Women and the conservation of agroecosystems: an experiential analysis in the Río Nacimiento region of Almería (Spain). Psyecology 5, 214-251

[5] Llobera F, Redondo M. 2014. Dinamización de iniciativas locales agroecológicas: el método TERRAE. Consumo gusto TERRAE. Diseñando estrategias de transición agroecológica desde la iniciativa de las administraciones locales. Red Terrae, MAGRAMA, y F. Biodiversidad.

[6] Marques MJ, Bienes R, Cuadrado J, Ruiz-Colmenero M, Barbero-Sierra C, Velasco A. 2015. Analysing perceptions attitudes and responses of winegrowers about sustainable land management in Central Spain. Land Degradation & Development 26, 458-467.

[7] Santos-Martín F, Martín-López B, García-Llorente M, Aguado M,Benayas J, Montes C. (2013) Unraveling the relationships between ecosystems and human wellbeing in Spain. PLoS ONE 8(9): e73249

TE-P_05 Ecological restoration approaches for degraded forests in landscape scale – Functional roles of corridors

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1. Objectives

The ecological status of the water catchment around a city is often the key to water and food security. The main objective of this study is to apply a new methodology, that complements the availability of data and status of degraded forests, to reduce forest fragmentation by emphasize on roles of corridors.

2. Research framework

2.1 Study area

To apply the ecological restoration approach, Shafarud watershed is selected, which is located in the northern part of Iran (see Figure 1). This zone, with about 39,800 ha of surface area, has a lot of fragmented forests, which are in the danger of degradation. Among the regional ecosystem, the Shafarud River, with approximately 34-35 km of length, is placed. Figure 1 demonstrates the location of the study area for this investigation.

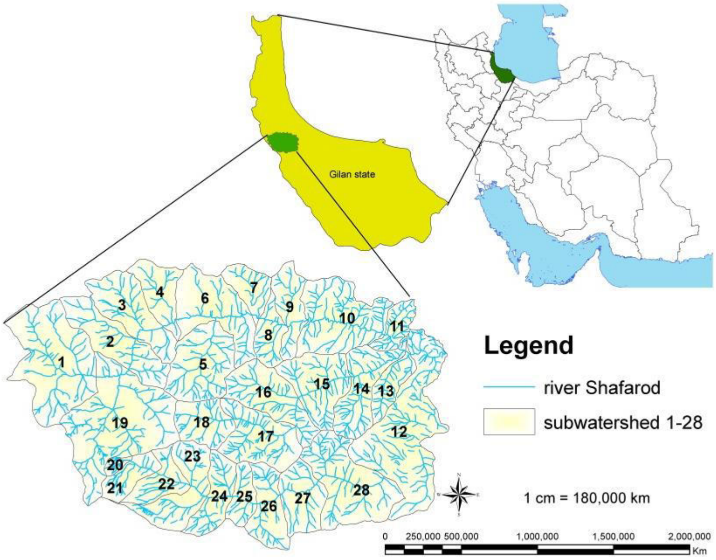

Figure 1: The location of the Shafarud watershed within the Gilan state of Iran.

2.2 Applied approaches

The method outlined here follows the conceptual “corridor” functionality for defragmenting in Shafarud watershed, with approximately 39,800 ha of forest. Data collection is conducted through the extracted satellite images from *Indian Remote Sensing (IRS), and digital maps* including land use, roads, and rivers atlas. For computing fragmentation, all of these maps are overlaid in Shafarud landscape. Priorities of restoration are evaluated by taking into account patches’ shape, surface area (i.e. more than 1 ha is recognized as a patch in this study), distance to river, and distance to roads. As acknowledged, a row of stepping stones is a non-continuous way to connect two patches [1, 2], which can greatly increase species’ mobility and ability to disperse in a fragmented landscape [3]**.** Thus, we recognized stepping stones as the corridors in this survey. Classifications of patches are carried out according to Forman, 1995 [4], which are based on convolution of edges (number of lobes) and elongation (the ratio of length to width), see Figure 2.

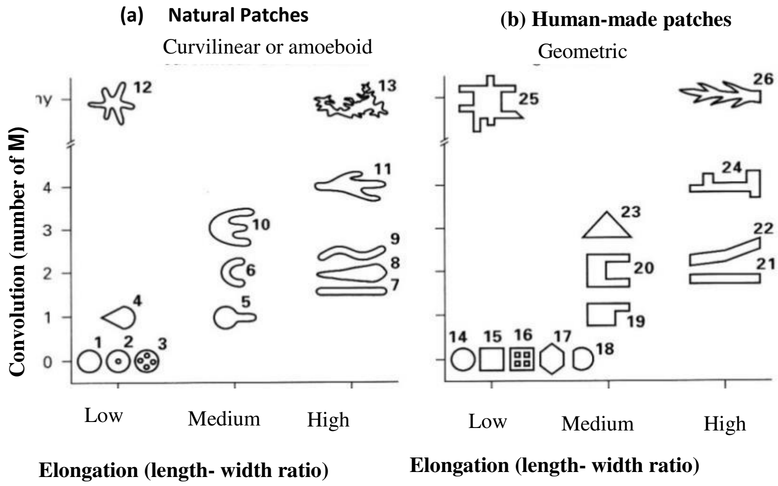

Figure 2: Applied classification of landscape patches in Shafaroud landscape
(a) Natural patches: **1** Bog, pino, tarn, hummock in wetland. **2** Slope surrounding dry hilltop, wetland in karst terrain. **3** Gaps within a patch. **4** Delta, alluvial fan. **5** Landslide, avalanche, woods extending along an island in a river. **6** Oxbow lake, barchans. **7** Glacial wetland or lake. **8** Drumlin, dune eyot. **9** Riparian woods. **10** Mountaintop with lava flows, headland around fjord. **11** Riparian woods along stream with tributaries. **12** Mountaintop vegetation extending along ridges, disturbance by mammal trampling around a waterhole. **13** Fire disturbance, pest outbreak disturbance. **(b) Human-made patches**: **14** Village around well or fort, central pivot irrigation. **15** City block, logged clear-cut in a chequer-board forest. **16** Woods with internal clear-cuts. **17** Geographical pattern of land use surrounding a central village. **18** Farm pond with a dam. **19** woodlot in an agricultural area, city park. **20** Suburban woods with an encroaching housing estate. **21** Cultivated field. **22** Golf course fairway, ski slope. **23** Field cut diagonally by a later road. **24** Woodlot lying within the intersection of several farms. **25** Town or city with development along roads and railways. **26** Dammed reservoir.

3. Results and discussion

Ecological restoration is defined as the process of assisting to recover an ecosystem that has been degraded, damaged, or destroyed. Different stages for ecological restoration can be considered, as shown in Figure 3, including naturalization, creation, reclamation, rehabilitation, and restoration in an ecosystem [5] . Here we assumed that by replacing the physical environment (i.e. corridors), the natural process can retain its biological capacities.

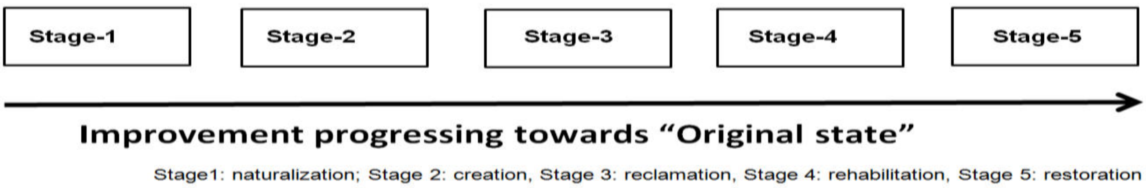

Figure 3: Multi stages process for ecological restoration in a region (Adapted from [5])

The functional role of corridors in a landscape planning is viewed as a tool for protecting ecological populations in a fragmented landscape, which can recover and support the ecosystem by connecting interior patches [9]. Connectivity is also used by scholars to describe the extent of flora and fauna movement in patches. As explained in Section 2-2, we adapted a more comprehensive approach for the functionality of a corridor, rather than being a linear element in an ecosystem. The results of several studies indicate that there is a link between the size of a corridor, and the tendency of species to stay in that. For instance, Kubes, 1996 [6] reported that an appropriate corridor size is in the range from 0.5 to 5 ha in length, and with a minimum width of 10 to 20 m. Thus, 1 ha is considered as an appropriate dimension (with a minimum width of 10 to 20 m) in this study.

For defining priorities in natural and man-made corridors, Core-Satellite configuration is used in this study, which is reported as an important parameter for selecting patches. In this approach, it is assumed that satellite and core region can be considered as an integrated area by establishing passage ways. Therefore, patches are classified, in this study, through their distance from rivers and roads. As follows, a corresponding code is allocated to each patch (see Table 1). According to the priority scale, number 1 is defined as the best state for forest restoration, while number 2 and 3 are recognized as moderate and worst state, respectively.

Table 1: Applied priority for selecting corridors (based on geographical situation).

Priority	Roads	Rivers	Restoration
1 (Best state)	Impassable via patches	Passable via patches	Available forest patches between two patches
2 (Moderate state)	Passable via patches	Passable via patches	Flowing river between two patches
3 (Worst state)	Passable via patches	Impassable via patches	Flowing river and passable road between two patches

After the extraction of the satellite images (i.e. *LISS-III* (Linear Imaging Self Scanning Sensor) of *IRS* (Indian Remote Sensing) satellite, in 2005, and with precision of 30 meter), the priority of restoration is judged for patches by the developed methodology (see Section 2-2). Table 2 shows the calculated priorities in Shafarud landscape for selected subwatersheds (i.e. numbers of 1-6, and 19-23 in Figure 1) according to the defined categories (see Table 1). These priorities are also applied in satellite image in order to have a better view of the region for restoration purposes. In this direction, the software of ENVI 4.0 is also applied for decoding and integration of digital maps and then, Majority Minority Analysis is adapted for filtering the integrated map. Finally, *Arc GIS 8.3* software is employed for evaluation the restoration priority. The graphical results are presented in Figure 4.

Table 2: Calculated forest patches priorities in the Shafarud landscape for selected subwatersheds (numbers of 1-6, and 19-23 in Figure 1).

Row number	Priority of river	Priority of road	Shape code*	area (ha)	Priority of restoration
1	3	1	20	1.5	1
2	3	1	13	1.3	1
3	3	1	13	6.3	1
4	3	1	13	6.5	1
5	3	1	8	1.0	3
6	3	1	21	1.9	1
7	3	2	13	17.4	3
8	3	1	24	2.8	1
9	3	3	19	1.3	3
10	1	1	17	1.6	1
11	1	3	19	2.5	3
12	1	3	20	2.1	3
13	1	1	17	1.4	1
14	1	1	13	4.1	1
15	3	1	21	1.5	1
16	2	2	22	1.2	2
17	3	2	8	1.0	3
18	3	1	13	5.3	3
19	1	3	8	1.7	1
20	3	3	16	2.8	3

*Based on Forman [7]

According to the applied classification of landscape patches (see Figure 2), the obtained mean abundance of each shape code in Shafarud watershed is reported as follows:

23=15>13>21=17>19>22>24>8>9>12=16

The entire watershed has about 683 patches (i.e. natural and man-made) with a surface area of 36,200 ha which can be used for ecological restoration. According to the developed methodology, 56 patches with a total area of 228 ha, are identified for establishing corridors in the Shafarud watershed (see Figure 4). From this amount, and based on the defined restoration priority, around 109 ha with priority of 1 (i.e. highest rank) are recognized as a suitable option for establishing the corridors, rather than 51 ha with priority of 2 (i.e. moderate state), and 68 ha with priority of 3 (i.e. worst state). The total corridor area that is needed for restoration of the Shafarud landscape is evaluated around 62 ha.

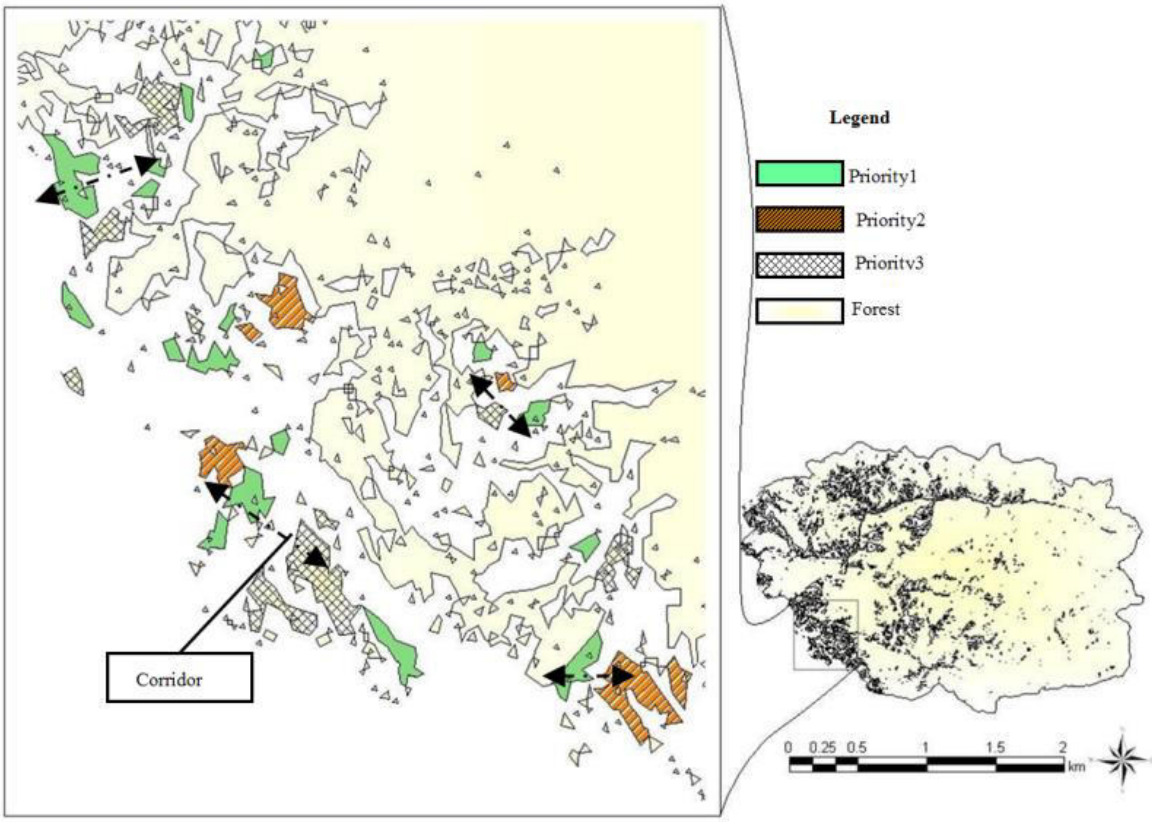

Figure 4: Obtained priorities of patches in the Shafarud landscape.

Despite the importance of a holistic landscape plan, very few studies were conducted in this direction. The main stream of scholars has focused on local scale application and biological conservation, commonly in wetland ecosystem. Marginnis and Jackson in 2003 [7], reported the important role of planted forest for ecological restoration, however no significant investigations can be found in the literature especially within the study area, which is recognized as one of the most important forest in the entire country. Our proposed methodology can be adapted in other degraded forests areas, and because of connections which are provided by the corridors, different species can move easily between the fragmented forests and this will prevent the isolation of species. As a result, natural functions inherent in the forest landscapes could be maintained.

4. Conclusions

Maintaining the integrity and sustainability of an ecosystem are crucial issues in a regional development plan. Regarding the integrated environmental model, forest is considered as a valuable and important part of an ecological structure. Ecological restoration in degraded forests can be started with naturalization of fragmented forests by appropriate corridors. The results of the proposed methodology indicate that about 56 patches with a total area of 228 ha, are suitable for establishing corridors in the Shafarud watershed.

References

[1] Hilty, J.A.; LidickerJr, W.Z.; Merenlender, A.M. (2006): Corridor Ecology: The Science and Practice of Linking Landscapes for Biodiversity Conservation. Washington D.C: Island press 324pp.

[2] Dramstad, W.E.; Olson, JD.; Forman, R.T.T. (1996): Landscape Ecology Principles in Landscape Architecture and Land use Planning. Island Press, USA, 80pp.

[3] Sondgerath, D.; Schroder. B. (2001): Population dynamics and habitat connectivity affecting the spatial spread of populations: A simulation study, Landscape Ecology, 17, 57-70.

[4] Forman, RTT. (1995), Land Mosaics: The Ecology of Landscapes and Regions, Cambridge University press.

[5] Brookes, A.; Shields, FD. (1996), River Channel Restoration: Guiding Principles for Sustainable Projects, John Wiley & Sons, USA, 458pp.

[6] Kubes, J. (1996), Biocentres and corridors in a cultural landscape: A critical assessment of the territorial system of ecological stability, Landscape and Urban Planning 35, 231-240.

[7] Marginnis, S.; Jackson, W. (2003): The role of planted forest in forest landscape restoration. In: UNFF Intersessional Experts Meeting on the Role of Planted Forests in Sustainable Forest Management, 25-27 March, 2003, New Zealand.

TE-P_06 Biosolid recycling in agriculture – A case study, Cairo sludge disposal study

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1. Objective

The Cairo Sludge Disposal Study is being funded within the Mediterranean Environmental Technical Assistance Programme (METAP) through the European Investment Bank (EIB). The study is a demonstration programme to show how the problems of sludge disposal from large rapidly developing cities, such as Cairo, can be solved through the establishment of a practical system for the safe reuse of sludge in agriculture.

2. Case study description

The implementation of wastewater projects in the major cities of Egypt Cairo will result in large quantities of bio-solids (raw sludge, digested sludge and composted sludge) being produced and requiring disposal. The Greater Cairo Wastewater Project will eventually result in about 0.4 million tonnes dry solids per year within the next 10 years or so (CSDS,1995). Disposal routes must be environmentally and socially acceptable, and cost-effective. Agriculture may offer the most sustainable and beneficial outlet for bio-solids, but there are concerns about protecting the environment and human health, and its practicality .The principal environmental concerns are due to the inevitable presence of potentially toxic elements (PTEs - mainly heavy metals) and human pathogens(Smith,1996).

Bio-solids should be regarded as a natural resource to be conserved and reused, rather than discarded. Its use in agriculture is widely regarded as the newly reclaimed soils in Egypt are characterised by low fertility, high salt content and poor moisture retention. Several investigators indicated the efficiency of different bio-solids in improving soil characters or increase the productivity of such soils (Abd El Lateef2003).

The Cairo Sludge Disposal Study has been initiated under the Mediterranean Environmental Technical Assistance Program, funded by the European Investment Bank and promoted by the Cairo Wastewater Organisation, in order to resolve at least part of the difficult problem of bio-solids disposal in the Cairo, which otherwise will become an overwhelming problem once all of the wastewater treatment plants (WWTPs) become operational with the full capacity. The objective is to demonstrate the practical and safe reuse of bio-solids produced by Cairo, and thus serve as a demonstration programme and information source for similar towns and cities in Egypt and warm climates beyond.

3. Methods

A 3 year programme of sludge sampling and analysis has been completed on all Cairo's WWTPs in order to characterise the quality of the sludges. The sludges have been analysed for a wide range of determinands, including nutrients, heavy metals and pathogens. The Cairo Study established 30 field trials, covering about 200 feddans (83 hectares). A wide range of arable crops have been grown over successive winter and summer seasons, including wheat, barley, fodder and grain maize, berseem, soya bean, faba bean, sesame, sorghum and cotton. Vegetables and root crops were excluded. A number of long-term fruit trials were also established and included grapevine, citrus, olive, banana, peach, apple, pear, persimmon and custard apple. The primary objectives of the trials were to evaluate sludge as a replacement for farmyard manure, to show that sludge performed at least as well as traditional methods, and that it can be used safely. Farms were selected as trial sites to evaluate the agricultural use of sewage sludge, encompassing the major soil types and crop husbandry practices within the nominal sludge 'marketing' area around Cairo. Six main sites were selected around Cairo, and three are being established near Alexandria. A total of 17 demonstration field trials were established with composted sludge from Alexandria involving several crops.

Cairo Wastewater Treatment Plants currently produce air dried raw or digested sludge which is pushed to drying beds and stocked pile for six months prior usage. Under Egyptian conditions to maintain a relatively high organic matter content in the product for its soil conditioning value, so complete stabilization is not necessary.

Table 1: Average nutrient and other elemental additions to soil in composted sludge estimated on volumetric basis(1).

Parameter	Units	Composted sludge	Farmyard manure
Ds	kg/m^3	614.8	563.9
VS	kg/m^3	208.5	223.1
OM	kg/m^3	170.5	165.1
N	kg/m^3	13.33	8.88
P	kg/m^3	2.38	1.65
K	kg/m^3	1.84	7.55
Fe	kg/m^3	10.32	2.77
Mn	g/m^3	113.2	63.2
Zn	g/m^3	207.7	71.0
Cu	g/m^3	126.0	51.8
Ni	g/m^3	25.5	15.9
Cd	g/m^3	1.3	1.3
Pb	g/m^3	33.3	2.8
Cr	g/m^3	24.2	6.7
Co	g/m^3	7.2	5.8

Note: (1) Average density of composted sludge was 0.7 t/m^3 (range: 0.65 – 0.8 t/m^3)

4. Results

4.1 Sludge Quality

The sludge quality surveys have shown the relative consistency of sludge quality at each WWTP, compared with the very variable quality of farmyard manure). However, the management and control of the sludge treatment and distribution operations need to be improved to ensure that this management strategy is sustainable, environmentally acceptable, cost-effective and minimises the potential risks to human health.

4.2 Hygiene

The study indicated that solar drying reduces the numbers of faecal coliform bacteria to ≤1000 MPN g-1 ds(dry solids) and this is indicative of the effective removal of disease-causing enteric pathogens. In keeping with this highly precautionary approach to the safe use of sludge, applications to fruit grown close to the soil (e.g. strawberries, melon) and vegetables which may be eaten raw, are not permitted since these may be in direct contact with the ground and also possibly with sludge cultivated into the soil. Field workers should also be trained to maintain good levels of personal hygiene when handling sludge

4.3 Nutrient content of bio-solids

Sewage sludge contains ergonomically valuable amounts of N and P, in addition to other major and minor elements required for plant growth, including Fe, Mn, Zn and Cu, which frequently limit crop yields in Egyptian soils, especially on the reclaimed lands (Table 1). Typically, solar-dried sludge from Cairo's WWTPs contains 1.7 % and 0.8 % of total N and P, respectively. The total K content is relatively small in comparison and approximately 0.3 %. Organic manures are applied to land on a volumetric basis in Egypt and 1 m3 of solar-dried sludge typically supplies 11.5 kg N and 6.0 kg P (19 kg N t-1 and 10 kg P t-1 on a dry solids basis).

4.4 Heavy metals

The detailed investigations on heavy metals in sludge carried out by the Studies have shown that the quality of sludge produced in Cairo and Alexandria are similar that of other industrialized countries. Also the heavy metal concentrations are well within the standards required for agricultural use and are not a barrier to reuse and do not limit rates of sludge application on farmland.

4.5 Effect of bio-solids in field crop production

The field trials programme demonstrated the effectiveness of solar-dried raw, digested, composted sewage sludges and farmyard manure as fertilizers and soil conditioners for arable crop production on alluvial clay soils of the Nile delta, and sandy soils on reclaimed land. The main factors influencing crop responses to the applied sludges were rapid and slow N release characteristics of the sludge; improved fertility of soil from cumulative applications of sludge or manures; crop requirement and sensitivity to N; soil type; and irrigation method .On clay soil, digested sludge gave the largest initial fertilizer response compared with other manures tested. As a general recommendation, the suggested optimum rates of application of raw digested sludge and farmyard manure on reclaimed desert soil are 20 m^3 fd^{-1} 10 m^3 fd^{-1} and 20 m^3 fd^{-1}, respectively. These rates should be reduced where specialist crops are grown with high N sensitivity (e.g. sesame), depending **on** the fertility of the soil after frequent additions of sludge or farmyard manure (Table2).

Table 2: Mean values showing the main effects of manurial, fertiliser and irrigation treatments on the yield of different crops in a six season rotation on clay soil (Nile alluvium).

Treatment	Cropping season and type					
	Winter(1) Wheat (t fd^{-1} grain)	**Summer Soya bean (kg fd^{-1} seed)**	**Winter Berseem (t DM fd^{-1})**	**Summer Cotton (kg fd^{-1} seed-cotton)**	**Winter Faba bean (kg fd^{-1} seed)**	**Summer Maize (t fd^{-1} grain)**
Manure (n = 16)						
No sludge	1.79c	568d	1.17b	509c	0.63c	2.23c
Digested sludge W	2.50a	816ab	1.28b	976a	1.13a	2.63b
Digested sludge S	No sludge	700bc	1.26b	950a	1.11ab	2.99a
Digested sludge W+S	W applied	870a	1.69a	742b	0.91b	3.21a
Farmyard manure W+S	2.29b	615cd	1.29b	510c	1.02ab	3.04a

(1) Note In the first season only winter applications are supplied
W is winter applied; S is summer applied
fd^{-1} =(feddan)= 4200m^2
Values followed by the same letter are not significantly different at P=0.05

4.6 Residual effect of bio-solids:

Sewage sludge has important cumulative and residual value for field crop production on reclaimed desert soils. Yields of crops grown on land treated previously with sludge may be increased by 10 - 20 % compared with normal farmer practice. Frequent applications of sludge to reclaimed soil increase soil fertility and can reduce inorganic N fertilizer requirements (Table 3).

Table 3: Cumulative effects of sludge on the yields of field crops relative to normal farmer practice.

Number of consecutive applications of 10 m^3 fd^{-1}	Crop	Increase above normal farmer practice (%)
2	Barley grain	22
3	Forage maize	36
4	Wheat grain (4.	100

4.7 Chemical composition of crops on sludge-treated soil

Extensive chemical analysis of field and fruit crops from the trials (25,000 individual plant tissue analyses were reported by the Study) did not reveal any significant relationships between heavy metal additions from sludge to soil and the concentrations present in plant tissues, which were in the normal ranges reported for crop plants in Egypt. In contrast to the heavy metals, significant and consistent effects of sludge and farmyard manure (FYM) on NPK content of crops were evident. The relative effectiveness of the different organic materials and N fertilizer value, based on observed patterns in crop N content, is as follows:

Digested sludge > FYM > N fertilizer > Raw sludge

5. Conclusions

A comprehensive scientific analysis of the potential value and safety of using Cairo sludge on agricultural land as a fertilizer and soil conditioner provided the assurance that, many climatic, soil, operational, agricultural and economic factors favour agricultural use of sludges under Egyptian conditions and warm climates. Climatic and soil conditions in Egypt strongly favour a reuse option because calcareous and clay soils limit crop uptake of heavy metals and potential toxicity. Also, the reclaimed land and clay soils are deficient in Zn and Cu, as well as other essential elements which are present in sludge and required for plant growth; and the extensive sunshine exposure, high temperatures and dry conditions provide aggressive and unfavourable conditions for the survival of microbial pathogens.

In addition to these beneficial climatic, soil and operational aspects, a further major advantage favouring reuse as a means of sludge management in Egypt and warm climates is the extensive and constant demand of agriculture for bulky organic manures as fertilizers and soil conditioners. Although these justifications favours the use of bio-solids in the agricultural land , it is recommended to treat the sludges to higher standards to assure higher levels of security and safety

References

[1] Abd El-Lateef, E.M. 2003. The Effects of Sewage Sludge Application on the Yield and Quality of Berseem and Forage Maize in Newly Reclaimed Soil. 2 nd I international Symposium of Ecological Sanitation. April 7-11 Lubeck ,Germany651-654.

[2] CSDS: Cairo Sludge Disposal Study. 1995. The agricultural use of sewage sludge in warm climates, with special reference to Egypt. Final Report Phase 1, Volume 1. WRc plc, Medmenham, Marlow, UK.

[3] Smith, S.R. 1996. Agricultural Recycling of Sewage Sludge and the Environment. CAB International, Wallingford, UK.

TE-P_07 Methodology for geo-based bioresource inventory shown on a case study of the town of Beočin, Serbia

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1. Objectives

When comparing large centralized energy systems with local energy communities, the latter may have advantages in becoming self-sustainable. The main goal of this work was to show that urban and suburban bioresources can have substantial energy potential.
The specific objectives of this work were:

- To classify selected urban and suburban, residue-based bioresources regarding biogas potential,
- To make an spatial evaluation of the collection, and the utilization of such bioresources in Beočin, Serbia, an sourounding,
- To evaluate the selected bioresources regarding usability for urban biogas generation based on bioresource quantities and spatial position.

The methodology consists of five steps. The results will be displayed in the same manner.

2. Methodology

2.1 Selecting bioresources and defining factors for calculations

The selection focused on residue-based bioresources, which are available in many regions and therefore are realistically obtainable also for Beočin. Since data for residue-based bioresources range widely in literature (e.g. regarding generation rates) and studies display various units (e.g. kilogram, litre, per square meter, per capita) a definition of most probable values with standardized units was made based on literature results. Before implementation of a concept into practice it is recommendable to verify and adapt the data to specific regional conditions.

2.2 Creating an GIS-based bioresource inventory including logistical networks

This part of the methodology was the core of the analysis. An overview on the bioresources in the community in question was done efficiently and relatively quickly by means of GIS-Inventory with "point-" and "area-resources". The software used was ArcGIS.
Different layers used in the GIS-inventory were downloaded from Open Street Maps depots (Geofabrik - www.geofabrik.de; Trimble Data - data.trimble.com). They were used as a base to portray the use of land, natural surfaces, man-made entities (e.g. roads, rail-, waterways, buildings). The administrative division layers were also used. It has to be mentioned, that the maps did not cover the complete area around Beočin and were also not 100% exact. Census and statistical values were taken from the statistical office. The layers had to be pre-processed in order to use them for the bioresource inventories.
For the logistical part the "Network Analyst" from ArcMap was used. It basically creates a spatial analysis based on networks of roads. It can help for example in determining the bioresource potentials within certain drive distances or time ranges. The network created via the "Network Analyst" is made out of points (e.g. bioresource sources and destinations) and lines (e.g. streets). Junctions were defined specially. Since urban bioresources can be transported to a point of usage via different ways, the step of defining junction could be crucial in the analysis.
The collection of the bioresources will consume a substantial amount of fuel. In respect to the energy content of the bioresource itself and the transport distances, it can make no sense to collect and transport them at all on the energetic point of view.

2.3 Analyzing the potential CO_2 savings

The bioresource itself is considered as CO_{2eq} neutral. The usage of fuels to power the transporting machinery was considered. First the number of trucks needed for bioresource transport was calculated via Eq. 1:

$$number\ of\ trucks = \frac{fresh\ mass\ of\ the\ bioresources}{load\ of\ truck} \qquad \text{Eq. 1}$$

In [2] the trucks specific emission was estimated with about 0.79 kg CO_2/km in town conditions while loading and transporting. With the average distance per tour, the number of trucks, and the CO_{2eq} emissions per truck the total yearly CO_{2eq} emission were calculated via Eq.2:

$$CO_2\ emission = truck\ amount \cdot av.distance \cdot spec.truck\ emission \qquad \text{Eq. 2}$$

Following assumptions for biogas utilization were made [2]:

- Producing one kWh of electricity saves around 0.55 CO_{2eq},
- Producing one kWh of heat saves around 0.38 CO_{2eq}.

Upon these assumptions CO_{2eq} emissions by transport and the CO_2 savings by biogas utilization were compared. The emission comparison was based on the conservative assumption that all conventional heating came from natural gas.

2.4 Determining the primary energy substitution

In this step, a preliminary estimation for the energy use of the households was made using statistical data for Serbia [1]:

- inhabitants per household (average),
- electricity usage 4,200 kWh per household (average).

This makes around 1,400 kWh of electricity usage per person in the average household. In the same manner heat energy consumption was estimated [2]. Using this data it can be determined which percentage of these needs could be replaced by utilizing biogas from the bioresources in question.

2.5 Determining potential areas for digestate utilization

A good option for removing the digestate from the biogas unit is by using it as a fertilizer. Using GIS it could calculated in which radius the digestate can be applied. However, it should be noted that at the time being there is no knowledge for Beočin which agricultural areas are used for growing crops and how their nutritional status is.
The main issue when determining the needed area for digestate application is that the amount of digestate applied on a certain area is depending on the digestate characteristics, namely on the nitrogen content, and on the recommendations for soil fertilizing, which should not be surpassed. According to these limitations an overview of the possible digestate receiving areas was given.

3. Results and Discussion

3.1 Bioresource selection and factors for calculations

The selected residue-based bioresources that were considered in the GIS-Inventory were:

- green waste from public green areas: 0.03 m^3_{biogas}/m^2 [2],
- green waste from private green areas: 0.073 m^3_{biogas}/m^2 [2],
- private kitchen waste: 112 kg of waste/ year & capita (6.38 m^3_{biogas}/ year &capita [2].

3.2 Overview of the GIS-Inventory

"Point-resources" such as household kitchen waste were relatively easily to evaluate and to map. The green wastes are "area-resources" and were more demanding to map, as they cover a certain surface. One example is shown in Figure 1. The data behind the maps can be used to gain numerical parameters for the bioresource potential calculations depending on the area.

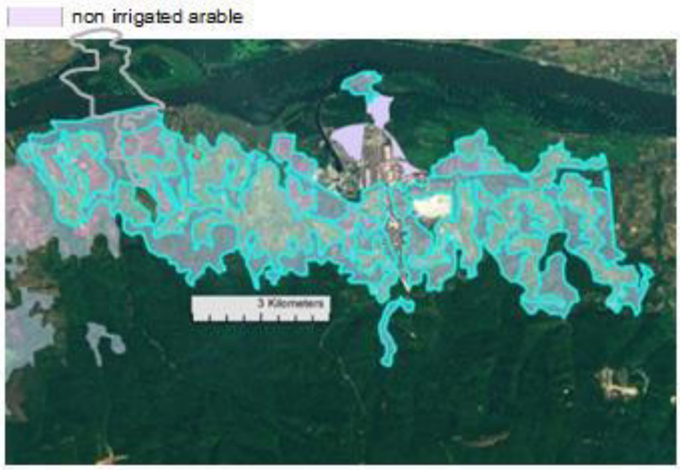

Figure 1: An overview of urban and surrounding agricultural areas in the 5000m distance from Beočin (selected in bright blue) (town highlighted in orange) (own depiction from ArcGIS).

The evaluation of the considered bioresources delivered following results for Beočin:

- public green areas (several meadows and a park): up to about 1.72 mio. m^2,
- private green urban areas: up to 2.15 mio. m^2,
- 1100 mapped housing objects.

In Table 1 an overview of the bioresource characteristics is shown. The organic matter of the mix was about 796 Mg/a and the dry matter content 14 % [2].

Table 1: Bioresource characteristics for the Beočin case

Substrate	Amount (Mg/a)	Mass Share (%)	Dry Matter (%)	Biogas yield (m^3/a)
Private kitchen waste	840	14.7	16	47800
Private green waste	3655	64.1	18	156950
Public green waste	1204	21.1	18	51600
Total	5699	100%	17.7	256350

The following results have been obtained by using Corine Land Cover data, Geofabrik data, data from the Statistical Office of the Republic of Serbia, as well as manual cartography of the private homes in the municipality of Beočin.
It is visible that the private green waste can potentially yield the biggest amount of biogas, which is logical, since most of the residential areas in the town are single-family houses.

3.3 CO_2 Savings

The abovementioned biogas amount could supply the following amount of energy for the town of Beočin:

- electrical energy in amount of about 600,000 kWh, saving about 330 Mg CO_2/a emissions,
- heat energy in amount of about 704,000 kWh, saving about 268 Mg CO_2/a emissions.

The calculated emission to transport all of the substrates totals up to 0.503 Mg/a. This value is almost negligible in comparison to the savings which can be made.

3.4 Primary Energy Substitution

The abovementioned produced energy amount can replace around 2% of electrical energy, and about 8% of the heat energy consumed by the Beočin within the limits of the analysis.

3.5 Digestate Management

The potentially liable areas that can receive the digestate are the marked in blue in Figure 1. These areas have 310 ha as potential surfaces for digestate application. The following table shows an approximation of the digestate characteristics with respect to the bioresource input into the biogas unit as explained in the chapter 3.2 (Table 1).

Table 2: Calculated digestate parameters in the Beočin case [3].

Parameter	Value
Degradation of the Organic Portion (%)	79.9
Dry Matter of the Digestate (%)	5.2
Nitrogen Content of the Digestate (%)	0.57
Amount of digestate (t/a)	5029

According to the before mentioned nitrogen input limit in the soil, we get a yearly value of around 36 Mg digestate /ha that can be used of around 139 ha of land where eventually a digestate application is possible. However it is not known which of this land is used to grow food or non-food cultures. The most likely utilization area must be determined in future phases of planning.

4. Conclusion

The results show that there are substantial, in this case practically unused, amounts of bioresources within the urban communities, which can be identified and dimensioned using GIS processing. The results shown were compiled based on several assumptions, so they may vary from case to case or from season to season. Still, the good side of this type of GIS-inventories is once the model of the inventory is made, it can be easily upgraded or modified in order to achieve a greater precision. The quality of the results is analogue to the input details.
Even though both proposed biogas plants cannot fulfill the complete energy needs, it is still a good example of integral bioresource usage. It can show that how much a cities can do with residue-based bioresources to become sustainable.

Acknowledgment

The foundations of this type of GIS research and assessment were laid in the frame of collaboration with the French "National Research Institute of Science and Technology for Environment and Agriculture" in Rennes, France (IRSTEA), during a study visit in the GIS research-team in November and December 2013. Especially we thank Thierry Bioteau for his guidance and valuable help.

References

[1] Statistical Office of the Republic of Serbia. Statistical Yearbook of the Republic of Serbia. Belgrade : s.n., 2010.
[2] Lazic, D. (2014): Methodology for Geo-Based bioresource inventories shown on a case study of the town Beočin, Serbia; Master thesis at the Hamburg University of Technology, Institute of Wasewater Management and Water Protection, Bioconversion and Emission Control Group.

TE-P_08 The effect of long term storage of dogs´ excrements with "Enviro" lime on the survival of helminth eggs

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1. Objectives

Regarding the spread of helminthoses, domestic animals (dogs, cats) are of great importance because they live in a close contact with man. Through faeces of infected animals the germs of parasitozoonoses spread into the environment. The most serious problem is the sanitation of the faeces. The aim of the study was to monitor the effect of long-term aerobic storage (73 days) of dogs´ excrements with or without "Enviro" lime on the survival of model helminth eggs under laboratory conditions.

2. Methodology

Dogs´ excrements mixed with hay in the ratio of 1:5 and mixed with "Enviro" lime in a concentration of 20 $g.kg^{-1}$ and 70 $g.kg^{-1}$ mixture were used in the experiment. General characteristics of tested lime were as follows: CaO + MgO - min. 82.0 %; MgO - max. 3.5 %; CO_2 - max. 11.0 %; Granularity **-** 0-1.0 mm.
The "artificial contamination of organic wastes" with unembryonated *Ascaris suum* eggs was used approach to make sure that there was a sufficient number of positive samples in our observations. *A. suum* eggs were isolated by dissection of a distal uterine part of female pig ascaris. The distal uterine ends were then removed to a glass homogenizer and processed. The water suspension of eggs was stored in an Erlenmayer flask in a refrigerator at 4°C. The controls with eggs were incubated in distilled water for the same time as the exposure time. Eggs were inoculated at a dose of 1000 eggs per carrier into polyurethane carriers and introduced into the wastes. The methods used for monitoring physical and chemical properties of organic wastes corresponded to the STN 465 735. The C content was calculated according to the content of OM by the method of Navarro et al. (1993), and the C:N ratio was calculated. Samples were collected after 0, 1, 2, 3, 7, 8, 9, 10, 14 and 73 days of the storage. Three samples were taken and analysed at the given sampling intervals.

3. Results and discussion

Utilisation of wastes resulting from domestic animal has been the subject of many investigations with regard to contamination of the environment by emissions, to the plant toxicity of treated wastes but also to the potential of survival and spreading of pathogenic agents. An important factor of the risk of infection transmission is also possibility of animal to move outside its housing (yard, move in nature), or the use of a dog (hunting or social). The most frequent way of transmission of parasitic diseases is through the contact (free-living animals with a domestic ones), or through contaminated environment with developmental stages (oocysts, eggs, larvae). Through faeces of infected dogs and cats the cysts of intestinal parasitic protozoa (*Entamoeba histolytica, Giardia intestinalis, Balantidium coli, Toxoplasma gondii*), the eggs of tapeworms (*Dipylidium* sp., *Echinococcus* sp.) and parasitic nematodes (*Toxocara canis)* can spread into the environment. Amongst the nematodes eggs*,* eggs of *A. suum* are reported to be the most resistant against environmental factors. Their egg shell is enveloped with an outer layer formed by acid polysaccharides and proteins and a central layer consisting of proteins (25%) and lipids (75%, particularly alpha glycosides). This protects *A. suum* eggs against effects of chemicals and drying and is s the reason why they have been chosen as model eggs.
Infection and way of transmission of the diseases depends on the biological properties of the involved pathogens as well as the utilization of the farmland and of the resulting crops. For the sani-

tation of animal excrements, the use of dust rejects from lime production, at more affordable price than quality lime, is very suitable.

The following changes in physical and chemical properties of the dogs´ excrements mixed with hay with or without “Enviro” lime were monitored: the pH, dry matter (DM), ammonium ions (NH_4^+) and C:N ratio. The physical and chemical properties of the treated material are given in Tables 1-3.

An application of “Enviro” lime to the mixed dogs’ excrements at a concentration of 20 $g.kg^{-1}$ of organic wastes, resulted in a devitalisation of 65.6 ± 2.8 % and at a concentration of 70 $g.kg^{-1}$ 77.0 ± 2.4 % of model unembryonated *A.* eggs within 24 hours. *A. suum* eggs were totally devitalised as early as within 8 days in dogs’ excrements after application of “Enviro” lime at a concentration of 70 $g.kg^{-1}$ and within 21 days after application of lime at a concentration of 20 $g.kg^{-1}$. 57.2±3.2 % of eggs were devitalised in the control without lime in the end of experiment (Table 4). The most important physico-chemical factors affecting viability of helminth eggs include pH and ammonia. We observed the highest pH and ammonia content especially in the organic wastes treated with tested “Enviro” lime.

Table 1: Physico-chemical properties of the dog excrements during long term storage.

Storage (days)	pH	DM (%)	NH_4^+ ($mg.kg^{-1}$ DM)	C:N
0	9.1±0.1	35.7±1.8	219.1±55.7	11.2:1
1	8.6±0.1	34.7±0.1	232.1±23.6	11.2:1
2	9.6±0.1	35.2±4.2	395.7±2.5	10.1:1
3	9.8±0.1	37.6±1.9	309.8±95.0	9.1:1
7	9.0±0.1	37.2±0.3	370.9±8.2	18.1:1
8	9.4±0.2	33.2±0.2	82.2±2.5	32.7:1
9	9.5±0.2	29.8±3.0	132.6±72.9	54.9:1
10	9.5±0.1	31.7±1.1	124.1±0.2	57.3:1
14	9.3±0.1	53.9±4.3	138.0±7.4	76.4:1
73	8.5±0.1	86.3±0.2	28.1±3.3	46.3:1

DM - dry matter; NH_4^+ - ammonium ions

Table 2: Physico-chemical properties of the dog excrements mixed with “Enviro” lime in a concentration of 20 $g.kg^{-1}$ during long term storage.

Storage (days)	pH	DM (%)	NH_4^+ ($mg.kg^{-1}$ DM)	C:N
0	8.4±0.1	37.2±0.1	400.6±47.8	9.6:1
1	11.2±0.2	44.5±0.1	12.5±8.6	9.7:1
2	9.3±0.3	56.4±15.1	36.2±7.8	12.4:1
3	8.6±0.1	57.3±33.2	645.1±362.6	4.2:1
7	9.1±0.1	45.1±6.8	225.4±91.0	15.4:1
8	9.1±0.1	43.9±2.8	439.8±141.3	16.7:1
9	9.27±0.1	68.7±1.3	398.6±2.5	20.9:1
10	9.1±0.1	64.1±0.1	349.0±10.1	17.7:1
14	8.9±0.1	60.0±0.9	338.5±24,9	14.7:1
73	8.7±0.1	89.1±0.1	74.5±26.6	22.6:1

DM - dry matter; NH_4^+ - ammonium ions

Table 3: Physico-chemical properties of the dog excrements mixed with “Enviro” lime in a concentration of 70 g.kg^{-1} during long term storage.

Storage (days)	pH	DM (%)	NH_4^+ (mg.kg^{-1} DM)	C:N
0	9.1±0.1	35.7±1.8	219.1±55.7	11.1:1
1	12.6±0.1	43.1±1.5	41.0±42.8	16.2:1
2	12.7±0.1	46.8±0.2	10.19	42.7:1
3	12.6±0.1	44.1±1.9	20.6±14.4	32.7:1
7	12.4±0.1	45.5±0.2	140.3±9.8	7.7:1
8	10.6±0.1	45.2±0.5	131.3±2.9	15.0:1
9	10.1±0.1	45.6±1.2	82.59	16.2:1
10	10.1±0.1	46.7±1.1	85.6±10.6	14.1:1
14	9.8±0.1	52.2±1.1	14.28	9.2:1
73	8.9±0.2	87.2±0.5	2.2±0.1	24.4:1

DM - dry matter; NH_4^+ - ammonium ions

Table 4: Survival of *A. suum* eggs during long term storage of the dog excrements with or without “Enviro” lime.

Storage (days)	Demaged *A. suum* eggs (x%±SD)		
	Control	**20 g.kg^{-1}**	**70 g.kg^{-1}**
0	12.6±1.1	12.6±1.1	12.6±1.1
1	35.7±2.5	65.6±2.8**	77.0±2.4***
2	54.4±10.7*	68.6±3.9**	82.3±4.8***
3	67.0±2.6**	75.1±1.2**	87.6±3.9***
7	62.6±4.0**	76.2±5.4**	97.1±3.9***
8	59.8±2.7*	76.9±2.7***	100***
9	61.9±2.9*	82.3±4.8***	100***
10	62.8±4.0*	85.7±1.4***	100***
14	61.9±3.3*	95.7±6.3***	100***
21	55.6±2.4*	100***	100***
73	57.2±3.2*	100***	100***

* Significance at the level P<0.05; ** Significance at the level P<0.01; *** Significance at the level P<0.001

4. Conclusion and outlook

The issues of safe sanitation and waste management are highly topical as it has been universally acknowledged that the majority of endoparasitic germs is able to cause infection in animals and humans even a year or two later. For the sanitation of dogs’ excrements, the use of “Enviro” lime, is very suitable. Model helminth eggs were devitalised within 8 days storage in dog droppings mixed with hay in the ratio of 1:5 and “Enviro” lime.

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References

[1] Navarro, A. F., Cegarra, J., Roig, A., Garcia, D. (1993): Relationships between organic matter and carbon contents of organic wastes. Biores. Tech., 44, 203-207.
[2] STN 465 735: Industrial composts, 1991

TE-P_09 Integrated management of water resources with an innovative aquaponic system under greenhouse conditions in Spain

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1. Objectives

The objective of this study is to design, develop and validate a prototype of aquaponic system under greenhouse conditions which is a combined production system of tomatoes and tenchs with a sustainable and integrated management of re-used wastewater as a water resource for the crop and potable water for the canals where tench fishes are being fattened.

2. Methodology

2.1 Description of prototype of aquaponic system

One prototype of greenhouse has been designed and built in the Experimental Centre of the Tecnova Foundation (in the province of Almeria, Spain) to make possible the combination of two production systems: the production of vegetables and fresh water fishes in one aquaponic system. This prototype of greenhouse had a surface area of 380 m^2 and an ogive multispan structure made with steel tubes, with film plastic cover and passive control of climatic conditions.

This prototype has been provided with five canals of 18 m long and 0.5 m width and height (with a total volume of water stored of 22,5 m^3) to fatten fresh water fishes. Ten hanging growing canals of a hydroponic pure system (NGS®) have been installed in this prototype made with a scrap metal structure and a film plastic multi-layer system, with three different film plastic layers located above this structure to grow vegetables provided with a drip irrigation system and with the recirculation of the nutrient solution. Every hanging growing canal was placed at 0.6 m from the soil surface of the greenhouse and had 18 m long, with a separation of 1.6 m between growing canals and of 0.4 m between drip irrigation emitters of the same drip line.

Three different filter systems have been installed in the head irrigation to apply three different treatments to the wastewater collected and recycled from the canals of fishes: (1) a sand filter used to retain solid stains, (2) an ultraviolet filter to eliminate the microbial load and, (3) a biologic filter to promote the nitrification process of the wastewater collected.

2.2 Experiments

Two combined production systems have been evaluated during a period of time of one year and a half: tomatoes and tench fishes (*Tinca tinca*) have been produced, with a sustainable and integrated management of re-used wastewater as water resource for the crop and potable water for the canals where tench fishes have been fattened.

Two consecutive tomato crops have been evaluated during this experiment and during one fattening cycle of tench. The first tomato crop was transplanted in January 2014 and finished in June 2014. The second tomato crop was transplanted in September 2014 and finished in May 2015. The tench fishes were introduced in this prototype with one year old and its fattening period has lasted twelve months. The canals of fishes were covered with plastic grilles and white mesh to reduce the amount of incident radiation and to favour the maintenance of the activity of these nocturnal fishes during the daytime. The tench fishes have been fed with fish flour fodders.

During the first tomato crop, three different strategies of water and nutrient management have been applied and evaluated in the aquaponic system: (A) an independent management of both production systems, (B) a mixed management and, (C) a closed management (pure aquaponic). During the second tomato crop, an economical assessment has been developed with the application of a mixed management of water and nutrients to validate this management system and to compare the economical results obtained with the conventional production system of tomatoes under greenhouse in southeastern of Spain.

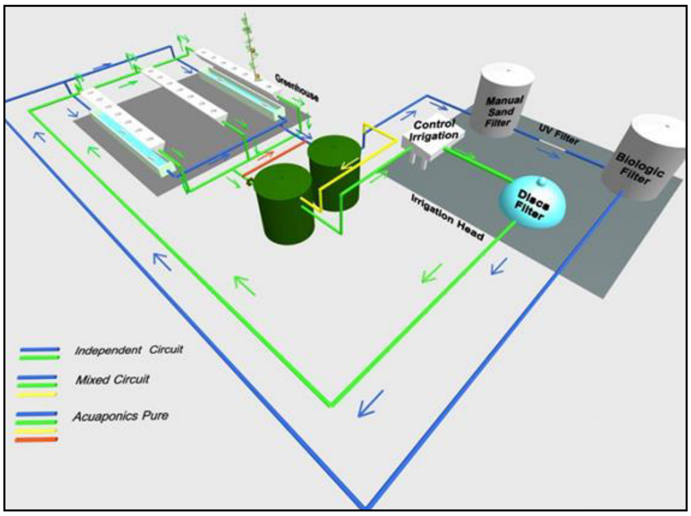

Figure 1: Sketch of the prototype of aquaponic system under greenhouse.

2.3 Water and nutrient management

During the phase of vegetative growth of the first tomato crop (two months of duration), an independent management of water and nutrients has been applied in the vegetable production system and in the aquaculture system. During this period of time, the water from the canals of fishes has been continuously recycled using one circuit. The nutrient solution of the pure hydroponic system has been recycled using another independent circuit and replacing this nutrient solution before reaching values of electrical conductivity of 3,5-4 dSm^{-1} and concentrations of 100 mg l^{-1} of nitrates and 1 mg l^{-1} of ammoniacal nitrogen.

A mixed management of water and nutrients has been applied during the phase of fruit set and fruit growth (two months of duration). During this period of time and every 48 hours, a volume of wastewater that ranged between 2 to 3 m^3 has been sent from the canals of fishes to the pure hydroponic system, to refresh the nutrient solution of the tomato crop. The nutrient solution for the tomato crop was adjusted taking into account the nutrients provided by the wastewater of the canals of fishes.

During the phase of harvest (last two months of the tomato crop cycle) a closed management of water and nutrients has been applied with the total combination of this two production systems and without the application of mineral fertilizers to the tomato crop. In that way, the amount and the size of fishes in the canals have determined the amount of nutrients that contains the wastewater for the canals of fishes.

Weekly measurements of concentration of nitrites, nitrates and ammoniacal nitrogen in the water of the canals of fishes have been developed using portable colorimetric tests.

3 Results and discussion

3.1 Evaluation of water and nutrient management

The independent management of water and nutrients in these two production systems has produced two kinds of advantages in comparison with the conventional production system of tomato in soil and under greenhouse in southeastern of Spain: a reduction of the amount of water and nutrients applied to the tomato crop (that ranged between 20-30%, depending of the quality of the irrigation water) and the increase of the benefits obtained per area unit.

The mixed management of water and nutrients between these two production systems has allowed the recycle of wastewater from the canals of fishes to refresh the nutrient solution of the tomato crop, with a reduction of the amount of mineral fertilizers applied to the tomato crop (this reduction depends on the number and the size of the fishes fattened in the canals). This mixed management of water and nutrients has produced a suitable development of the tomato crop and a suitable growth of tench fishes.

The closed management of water and nutrients has produced deficiency of nutrients in the tomato crop. The population density recommended for the tench fishes (7-15 kg m^{-3}) seems not to be enough to produce the amount of nutrients needed for the tomato crop in the wastewater of canals.Other fattening fish with high population densities (like red tilapia) could be more appropriate for pure aquaponic systems with tomato crop.

3.2 Validation of the aquaponic system

The species of fish selected (*Tinca tinca*) has been fattened properly in canals under greenhouse without active control systems of climatic and water conditions. The necessary fatten cycle of tench species to reach a commercial weight has lasted twelve months under Mediterranean conditions, and it is completely compatible with the crop cycle of tomato crop. The use of hanging growing canals to develop the tomato crop has enabled the combination of these two production systems (tomato crop production and tench fishes fatten) under the same greenhouse structure, and it has increased the benefits obtained per area unit (between 8 to 10 times, depending on the market price of tench).

The application of a mixed management of water and nutrients has demonstrated to be the best option for the combination of the production systems evaluated. It has produced a suitable growth and yield of the tomato crop and it has improved the use of resources (water and nutrients). The re-use of effluents produced in the canals to the NGS system has provided to the tomato crop a source of nutrients rich in ammonium, nitrates, phosphates and organic substances, reducing the amount of mineral fertilizers that must be used periodically to refresh the nutrient solution of this closed hydroponic system. The amount of wastewater (2 to 3 m^3) derived every 48 hours from the canals of fishes to the NGS system has been enough to attend the water needs of the tomato crop.

The direct costs that have increased with the combination of these two production systems are the energy and the labour costs. The use of photovoltaic solar collectors could be a good option to self-supply of energy to this prototype in this region (which receives the highest number of sun hours per year of Europe). The increase of labour cost has been little and it has been due to maintenance labours. The cost of this prototype of greenhouse is higher than the cost of the traditional greenhouse widely used in the Mediterranean region. The increase of the net benefits obtained per area unit (from 3 to 8 euros m^{-2}) allows facing the amortization of this equipment easily by farmers.

4. Conclusion and outlook

This innovative aquaponic system is a model of sustainable production system in which urban sewage water and effluents from canals for fatten tench fishes are re-used to growth a tomato crop, reducing the use of inputs (water and fertilizers) and increasing the benefits obtained per area unit.

The management of water and nutrients more suitable for the combination of the production systems evaluated in this experiment (tomato crop and tench fishes) has been the mixed management, deriving an amount of wastewater from the canals of fishes every 48 hours to the NGS system. It has produced a suitable growth of the tomato crop and the tench fishes, the total reuse of the water resources and the reduction of the amount of fertilizers applied to the tomato crop.

The advisable population density of tench fishes has demonstrated not to be enough to provide the needs of nutrients of the tomato crop with a pure aquaponic system. Additional combinations of production systems should be evaluated with pure aquaponic systems in Mediterranean climatic conditions with the integration of renewable energy systems.

Acknowledgements

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References

[1] Martan, E. (2008). Poyculture of fishes in aquaponics and recirculating aquaculture, Aquaponics Journal 48: 28-33.
[2] Murray, F.; Bostock, J.; Fletcher, D. (2014). Review of recirculation aquaculture system technologies and their commercial application. Highlands and Islands Enterprise. University of Stirling Aquaculture.

TE-P_10 Changes in farming production strategies with wastewater in Sacaba Valley, Bolivia

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Objectives

This study identifies farming production strategies with wastewater in Sacaba Valley to contribute to assessment of their risks. It has two objectives. The first one was to identify characteristics of production with wastewater. The second one was to know the changes shown in such characteristics during the last ten years to identify the factors that caused these changes.

Methodology

It consisted in transect walks alongside the river that receives wastewater, on-farm enquires to 19 farmers, semi-structured interviews, field observation and revision of secondary information. Identified production strategies and practices have allowed classifying plots according to intensity, type of crops and agricultural practices in production systems. Comparing former and current characteristics have served as a basis to identify main factors that originated such changes.

Results

Three production systems have been identified: a) horticultural semi-intensive, b) forage semi-intensive and c) horticultural intensive. Two main changes have been identified in production strategies on these systems: the search for alternative sources of water –which has resulted in one type of on-site treatment by filtration– and change of produced crops. These changes have been triggered by six main factors, four of which have already been mentioned by Raschid-Sally et al (2004) –i.e. reliable flow availability, farm size, village conformity, market access– and two are news: social pressure and organization level of farmers.

Conclusion

Understanding of the main factors that explain the situation of production systems can be achieved during the process of risk assessment, and information that they provide may be useful as a part of a long term planning for sustainable use of wastewater. On the other hand, it is a need to get reliable data about the resulting risk of identified production practices, strategies and the on-site treatment.

References

[1] Raschid-Sally, L.; Bradford, A.; Endamana, D. (2004): Productive use of wastewater by poor urban and peri-urban farmers: Asian and African case studies in the context of the Hyderabad Declaration on Wastewater Use, in: Moriarty, P.; Butterworth, J.; van Koppen, B.: Beyond Domestic: Case studies on poverty and productive uses of water at the household level. Delft, the Netherlands, IRC International Water and Sanitation Centre (Technical Paper Series; no 41)

TE-P_11 The forces of globalizations spin the resource cycles – Australia's water tradeoffs and the water-energy-food-nexus

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1. Introduction

Worldwide more and more areas suffer from water scarcity. Basins get exploited faster than their replenishing speed and each collapsed basin is one natural water storage system less and, especially, in the case of aquifers difficult to recover once overexploited and a loss of high quality drinking water.

The nexus discourse, which reached the international arena with the Conference "The Water Energy and Food Security Nexus – Solutions for the Green Economy" picks up the interconnectivity of resource cycles (like water, energy or soil) and some political, economical and social dimensions. However, only some first steps are made to analyze local water scarcity caused by ample interconnected factors with multiple synergies by the forces of globalizations_[11]. This work tries to develop an analytical framework for understanding the forces pushing local water users to unsustainable water usage, which effects the global water storage capacity.
The hypothesis is that resource cycles as a nexus connected are heavily influenced by the forces of globalization and hegemonic struggles on quantities and ways of resource exploitation pushed by competing interests. These forces and synergetic links are an interconnected global network of factors socially and territorially wise disconnected from the basins, however, affect them heavily locally. Each basin is embedded in a different social, cultural and political framework shifting the factors, buffering or increasing the affects of some. Therefore, more dimensions have to be included into the nexus framework to make the global anthropogenic link more visible and analyzable.

This article connects the nexus framework with theories of power and the concept of hegemony as an important dimension to analyze cases of global water losses. A contribution in the development of theorising water exploitation regimes by using exemplary cases derived from Australias energy nexus discourse.

2. The water-energy-food nexus and struggles for hegemony

Local water scarcity is affecting the global water balance in form of less water stored in easily accessible systems like surface water systems, aquifers or soil. Most surface water systems are more and more polluted, which makes aquifers in terms of drinking water supply even more valuable, even though more effort is required to reach the resource. Several cities in this world switched in their history from surface to underground supply due to water quality issues and as one of the first measurements before installing treatment plants. However, currently, more and more cases occur, where the drinking water supply switches toward desalination plants, which is another step to the less easily accessible resources like in Australia.
Each step from easy accessible water resources to salt water or even ice caps requires a resource input with energy as one part of it, e.g. pumping, treatment or desalination plants. Therefore, the link between energy and water is getting closer and closer already just by looking at the drinking water supply dimension.

Apart from direct drinking water supply is food needed for our most basic survival and irrigation is needed for proper yields in most areas of the world even when not situated in arid regions. Food production, especially, since the green revolution, when using chemical fertilizers and pesticides became conventional farming methods got highly draining on water quantities, quality and soil availability in high quality with its also important water storage capacity of soil.

[11] Globalizations instead of globalization as the process takes different forms and shapes well described by Bob Jessop (2003) from the political economy perspective as the forces of globalizations.

On most land in dry regions like Australia is farming not practiced without irrigation and the majority is conventional farming. This does not mean that it is not possible without irrigation and organic farming is on the rise in Australia with 2015 10-15% of the overall farming (Australian Bureau of Statistics 2012). However, the effects of conventional farming are severe and the water requirements of plants do not play a major role in the choice of crops if irrigation is still possible (Hoekstra 2010). The world price or trade links are much bigger pushing factors in the decision making process of what is planted than planting crops, which are most suitable for the local climate with considering the replenishing speed of the basins irrigation is tapped into, if not otherwise regulated. Conventional farming methods intensify the situation even further as fertilizers pollute the water and soil quality directly. And again by shifting towards less accessible and technological intense methods for water treatment, reuse or desalination more capital and energy is required, even though innovations could reduce in some areas these expenditures. Especially, developing countries without the capacity for strong regulative water management frameworks are endangered of draining their water resources without being able to shift their supply to less accessible stored systems. A global phenomenon is that valuable basins as easy accessible water storage systems are getting destroyed with the difference that industrialized countries have more opportunities to buffer the effects and to reverse some.

The energy component is getting more and more important in regards to water, but is only part of a world society with rising energy requirements. The Australian economy pictures this development with rising exports in the energy carrier sector and by trying to tap more and more into unconventional energy carrier exploitation. At the same time creates the fuel and mining industry severe environmental problems and most fossil energy carriers have apart from a big carbon also a big water footprint with unconventional ones an even bigger one (Gerbens-Leenes 2008, EPA 2011, Government of Alberta 2011*)*. A case for close synergetic links of the water-energy-food nexus spiraling into water scarcity used to develop a new nexus framework by focusing on international trade and hegemonic struggles as one of the driving forces.

In order to analyse these often competing interests and alliances establishing hegemony is seen as a process of power struggles between different interest groups shaping the discourse on issues influencing resource cycles.

Max Webers (1922) definition of power: that power means each chance in social relationships to set your will above another person will, where it does not matter on which ground this chance is based, has not been challenged, however, more sophisticated frameworks have been developed. For this work Power is captured by the concept of hegemony. The concept of Hegemony goes back to Antonio Gramsci. He argued that hegemony stands on three pillars and that there are more dimensions of power then just the institutionalized materializations. International relations challenges his approach, especially, early ideas of measuring power by counting the military equipment of different parties or to transfer this to another context to just compare how financially equipped different protagonists are. His three dimensions of hegemony are: intellectual, moral and political, which are important for establishing hegemony. This approach does not neglect the institutionalized side, but can explain that other components are necessary for establishing hegemony and that there are different avenues for groups becoming hegemonic. These dimensions already imply that other tools are needed to analyze hegemony as for example moral shifts are more difficult to capture as counting money or tanks.

In general, establishing hegemony is seen as a process of power struggles between different interest groups shaping the discourse on issues. Hegemony in tradition of gramscian thought includes not just the domination of a political debate and ability of a hegemon to quiet public dissent by force, more so, the capacity to shape interests of antagonists in order to reach consensus in favour of the hegemons interests (Wullweber 2009). The hegemon is, therefore, the party which was able to shape the discourse in its favor at a certain point of time, however, as these power struggles are a process discourses are getting shaped by ample interest groups which form coalitions on certain issues and others on others. Furthermore, different actors forming these groups shape discourses differently in different structures of their historically grown environments. These temporary coalitions can be hegemonic for a short or longer period as hegemony has to be always reconstituted.

Therefore, is hegemony established and can be analyzed through discourses. Discourses are seen in this work in the tradition of Michel Foucault (1969): A discourse is not just what different protagonists of interest groups say or write. The character of a discourse is the ability to establish connections between institutions, economical or social processes, models of behavior, normative systems, techniques, types of classification and characterization. A discourse can be established and characterized in different forms as there are ample of materializations. Discourses are the interactions, but also the material what is left due to these interactions and are seen as an ongoing process.

Therefore, resource cycles as a nexus connected are heavily influenced by constantly reestablished discourses, hegemonic struggles on quantities and ways of resource exploitation. More dimensions have to be included into the nexus framework to make the anthropogenic link more visible and analyzable. The following overview shows discourses impacting directly or indirectly resource cycles with the innermiddle spheres buffering or increasing the impacts dependable on the regulative framework. The outer spheres impact the inner ones and vice versa.

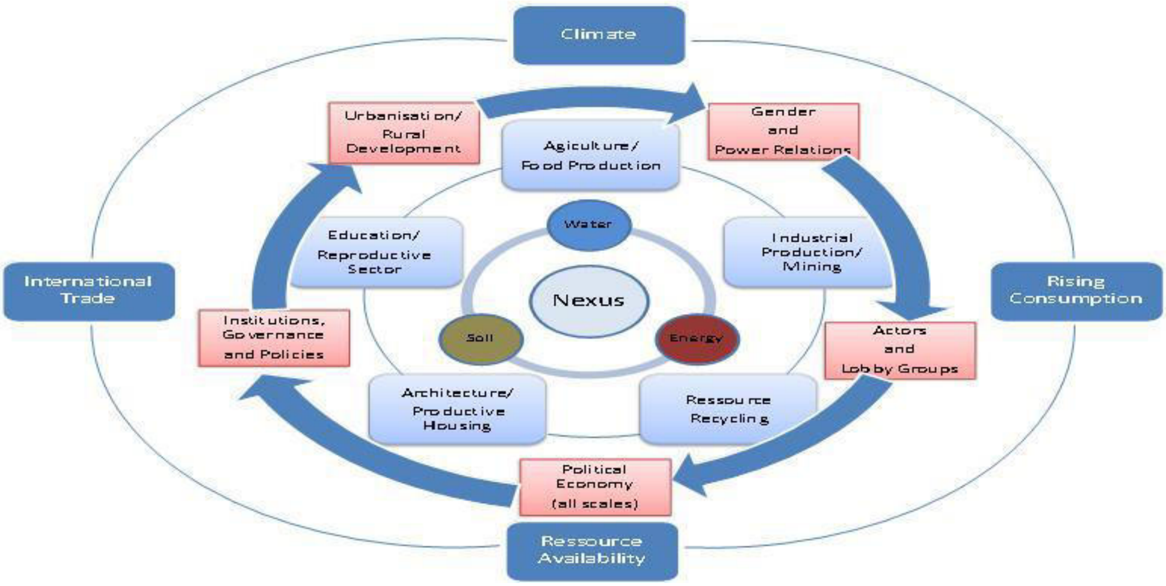

Figure 13: Energy-Soil-Water Nexus in a Global Context by R Schaldach.

This framework can be used to concentrate on specific discourses without losing the bigger picture of other dimensions and is the developed methodology for looking at local impacts and interlinked nexus cases. Inner, outer spheres and dimensions in one orbit impact each other and create synergies, it depends on the specific context which dimensions have a stronger or minor impact, however, play always a role.

Competing Cycles – Competing Interests in Australia

Australia is one of the driest continents and a major energy carrier exporter with an expanding market in unconventional gas. The second, but much smaller economical pillar is the agricultural sector and they are competing already for water along with the metropoles drinking water requirements.

As the table (STAT-WTO 2015) with Australia's exports in US dollars at the current price shows, there is a general increase in exporting fuels and mining products with a steep development during the last ten years.

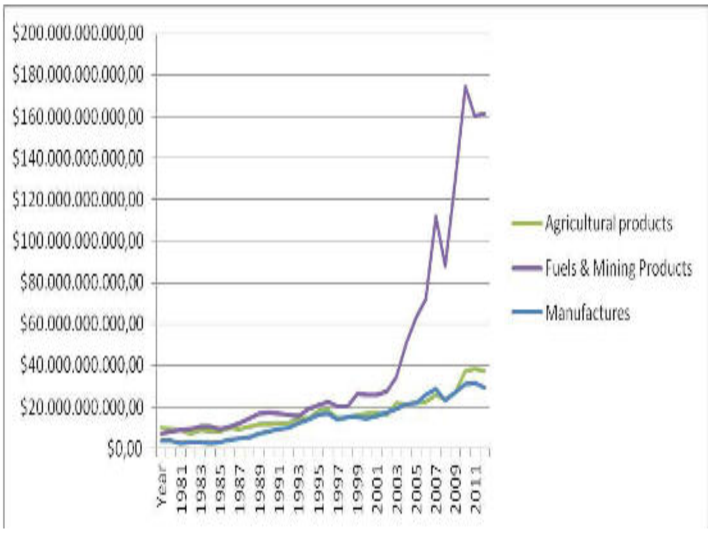

Figure 2: Australian Exports in US Dollar at the current price, data derived from WTO Stats 2015.

Coal makes Australia, especially, to one of the global energy exporters highly linked to Chinas increasing demand as one of their major trading partners. Due to resource abundance and low population numbers, even with high per capita consumption, energy production is not driven by seeking energy independence. The debate is compared to the U.S. more focused on shifting energy flows towards more sustainable and less emitting solutions as the environmental consequences of mining are on Australian soil.

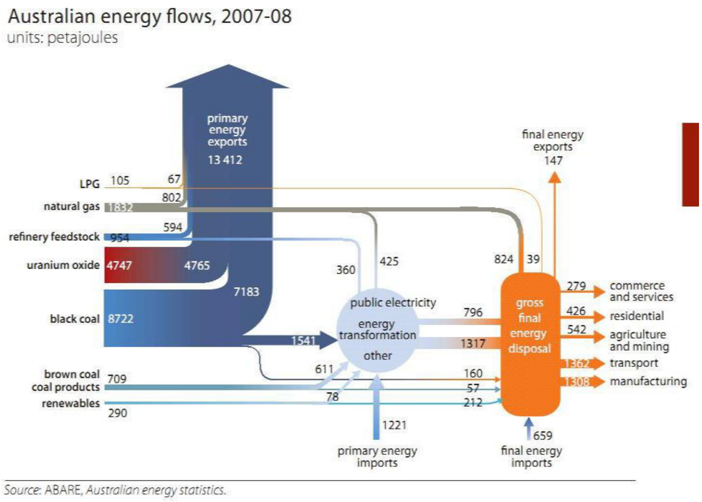

Figure 3: Australian energy flows, 2007-08.

The energy flow graph shows that renewables play only a minor role in Australia's energy mix and the majority of energy carriers is getting exported.

Even when the unconventional energy carrier sector seems small compared to coal exports developments mainly of coal seam methane increased after 2004 and is planned to expand (CIA 2011). The Greens Party critcised this expansion plan of 40.000 more drills in Queensland until 2030 (Australian Greens PolicyInitiative 2011). Similar to the U.S. situation environmental concerns grew and were mainly fired by the anyway difficult water situation of farmers in Australia after severe droughts. Therefore is water anyway a sensible topic and triggered conflict between the major users in rural areas: mining companies and farmers. In respond to public debates since end of 2010 regulations are debated and set into place or bans in place like in Tasmania (Davidson 2015). However, until now these regulations are mainly concerned with land rights, compensations, flow back waste water treatment and similar to the U.S.: publication of fracking fluids.

Interviews with the Anti Fracking movement and other stakeholders showed, that this movement grew partly out of the Anti Uranium Mining Movement, which is connected to the Peace Movement due to the proliferation component of Uranium. There are also strong ties to farmers with sometimes even surprising partnerships between environmentalist activists and the Australian high scale meat farming industry, which is historically mostly in the political conservative corner, however, water concerns are there the binding interest.

The anti-uranium mining movement grew also very close together with the movement for indigenous rights as often land rights of indigenous people and prosperous mining areas are in conflict. These indigenous lands often also harbor sacret water spaces and connect these struggles to the spiritual component of water in several religions. Religions are included in the framework as interests groups, but were considered as a dimension on its own as there are strong links between water usage and religious rules. At the same time are these struggles a case for conflicting discourses resulting in a growing one on sustainable resource usage as a precondition for a fast developing organic farming industry during the last decades as one way of using synergies between resource cycles.

An example for another global spanning component in this discourse is, that the energy crisis of one country can push the water crisis of another. The new and highly discussed trade agreement between India and Australia will fuel Indias nuclear energy expansion plans (signed in Sept 2014). This will secure markets for the planned further expansions of uranium mining activities in Australia, but is also highly debated as india is one of the nuclear weapon powers, never signed the non-proliferation treaty (NPT) and is seen, therefore, as a breach of the third pillar of the treaty by Australia by delivering uranium to a non signatory country.

This overview on some discourses in the Australian energy nexus context shows the connectedness of struggles and at the same time geographically disconnectedness between the materialized outcomes of these struggles for the water basins and soil quality.

3. Conclusion

Water regimes resulting in over withdrawal and in the worst case collapsing basins can not be understood without understanding hegemonic struggles of interests on ample scales, which can be also geographically, sometimes topic wise highly disconnected from the basin. Further research needs to be done on building up a strong hegemonic bloc supporting discourse shifts resulting in water preservation and sustainable ressource management.

References

[1] Abares Australian Energy Statistics (2012) Energy in Australia available at: http://www.industry.gov.au/Office-of-the-Chief-Economist/Publications/Documents/energy-in-aust/energy-in-australia-2012.pdf (Accessed 15.08.2015)

[2] Australian Bureau of Statistics (2012) *Australian Organic Market Report 2012*. available at: http://austorganic.com/wp-content/uploads/2013/09/Organic-market-report-2012-web.pdf (Accessed 15.08.2015)

[3] Davidson, Helen (2015) 'Tasmania bans fracking for another five years to protect agriculture' *The Guardian 26.02.2015* Environmental Protection Agency (2011) Hydraulic Fracturing Study available at: http://www2.epa.gov/hfstudy (Accessed 15.08.2015)

[4] Gerbens-Leenes, Winnie; Hoekstra, Arjen Y. (2008) *The water footprint of bioenergy*. available at: http://waterfootprint.org/media/downloads/Gerbens-Hoekstra-VanderMeer-2009-WaterFootprint-Bioenergy_1.pdf (Accessed 30.08.2015)

[5] Hoekstra, Arjen Y (2010) *The relation between international trade and freshwater scarcity. available at:* https://www.wto.org/english/res_e/reser_e/ersd201005_e.pdf (Accessed 15.08.2015)

[6] Hoekstra, Arjen Y (2013): *The Water Footprint of Modern Consumer Society*, Routledge

[7] Jessop, Bob (2003): Globalization: It's about Time too! Institute for Advanced Studies, Vienna Foucault, Michel (1969) *Archäologie des Wissens.* Suhrkamp [8]Verlag, Frankfurt a. M. 1969/1997

[9] Weber, Max (1922) *Wirtschaft und Gesellschaft. Grundriß der verstehenden Soziologie.* Tübingen 1922/1980

[10] WTO Statistics Data available at http://stat.wto.org/ (Accessed 15.08.2015)

[11] Wullweber, Joscha (2009) *Globalisierung, Macht und Hegemonie. Dampfboot Verlag*

TE-P_12 Volunteers to translate waste prevention policy to citizens

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1. Objectives

Flanders (northern part of Belgium) is one of the best students in the field of separate collection of waste. This also counts for the organic fraction. A large amount of organic residues are processed by the people, at home or in their gardens. The Flemish government wants to further increase the number of home processors and the quality of home processing of these organic materials. To put this into practice, Flanders follows a very specific and unique approach.

2. Methodology

In Flanders, Vlaco npo supports and implements the policy of biowaste. Vlaco is a membership organisation with representation of both the Flemish government (OVAM and intermunicipal waste associations) and the private sector (private waste treatment companies). All the Vlaco activities support the sustainable material cycle of biowaste.

The 'Biocycling at home' unit of Vlaco focuses on raising an environmental awareness concerning bio-organic materials. All individuals are to be sensitized and convinced in a different way. On the longer term 'Biocycling' (or 'Kringlopen', a verb in dutch) must evolve in a 'way of life'. A 2-fold awareness approach has proven its success. The first approach - the most specific one - focusses on volunteering, enthusing and engaging. The second focusses on the content of the message and educating.

2.1 First approach

Around the year 2000, home composting was hot; it entirely fitted within the former Waste Decree. The main issues we distinguish in that 'Home Composting' scheme were composting itself, and the use of that self-made compost.

As the Waste Decree was interpreted more broadly, through the years, the 'Home Composting' scheme evolved to a 'Closed Loop Gardening' scheme (± 2003). From that moment on, besides composting, also other methods were recommended to process organic residues at home: f.i. lawn maintenance, processing of branches, perennials and how you can reduce organic waste by using these perennials instead of grass, chicken keeping ….

As the Waste Decree was replaced by the Materials decree (2012) solving the waste problem was no longer the central theme, but the using of waste as a basic material within a closed loop (or biocycle) became the main message from that moment on. Also Vlaco as initiator, facilitator and supervisor of the policy on organic biological waste evolves in this direction. In 2012 the 'BioCycle at Home' scheme was initiated. A new theme that was added to the responsibilities of Vlaco was the communication about food losses and how to prevent them. From that moment on also a lot of secondary, more specific themes came into the picture.

Currently, the Vlaco-unit 'Biocycling at Home' has trained several thousands of volunteers. These volunteers (so called Master Composters, or Biocycle Volunteers as we call them now) assist the Municipality by promoting the 'Biocycle at Home'. They communicate about seven different techniques to achieve the biocycle of:

- Food waste
- Lawn
- Prunings
- Home composting
- Compost use
- Chicken keeping
- Perennials

Vlaco trains the Master Composters. About 40 teachers are available to regularly train these volunteers and to update them. In total 4000 Master Composters have been trained the last 20

years. For the moment 2700 of those Master Composters are still active (which is about 1 per 2000 inhabitants).
They are volunteers that assist municipalities and the so called Intermunicipal Waste Associations in promoting the BioCycle at Home, in their own municipality or city.
Master Composters (or biocycle volunteers as we call them now) are volunteers in a team supported by the environmental, sustainability or green officer at the municipality.
The reason why Master Composters are being 'used' in communication (and not just folders or online information), and the cause of their ascertainable success, is because Master Composters have a better credibility compared to the 'official message', they stand with both feet in practice (that's why people believe them), they use their own words and work in their own village, and they can demonstrate things.

2.2 Second approach

Vlaco also approaches the public directly. This includes:

- organizing courses (about the preventing and processing of organic remains),
- (co-)organizing campaigns and events (Closed Loop Weekend, Closed Loop Festival, Floralies 2016 …),
- distributing leaflets, brochures, posters (leaflets as some sort of teasers, and booklets for those who want to know more about a specific theme),
- communicating by several types of (social, internet or paper) media, and through inter-municipal waste associations and the local environmental services,
- using other educational materials (demonstration tools about processing organic remains f.i. compost boxes and –bins, wormeries, insect hotels, mulch mowers, wood chipper, school games, compost information box…)

3. Results and discussion

Research shows that this approach makes sense and is gradually evolving the way we wanted. A combination of instruments undoubtedly lies at the basis of the success (financial, policy and public awareness).
In the table below you see the percentage of citizens that are practising the closed loop technique named 'Home Composting'.

- In 1991 5% of the people in Flanders were composting at home.
- In 21 years time this amount evolves to 52%.

Note that this percentage includes all inhabitants of the Flemish region, not just those who have a garden. Currently, pproximately 106.000 tons of organic waste is processed at home, by composting. The exact amount is very difficult to estimate.

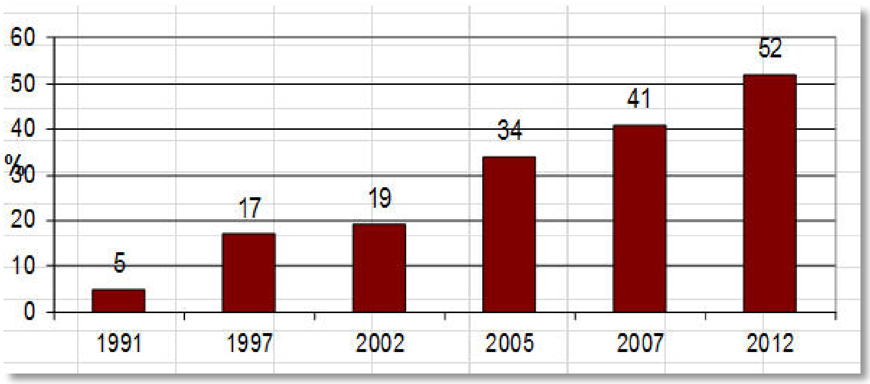

Figure 1: The percentage of inhabitants that are composting (all or part of) their organic waste at home (by using wormery, compost box, compost bin or compost heap), throughout the past 21 years.

In the table mentioned below, one can see some other methods of processing home produced organic residues and the percentage of people that are practising these methods at home.

- About 17% of the people practices one or another residue reducing method to maintain their lawn.
- Over 22% of the people applies one of the pruning processing methods I mentioned earlier.
- Around 58% applies perennials in the garden to reduce the garden residues and garden work.
- And more than 1 on 4 families in Flanders keeps chickens to reduce the organic garden and kitchen residues.

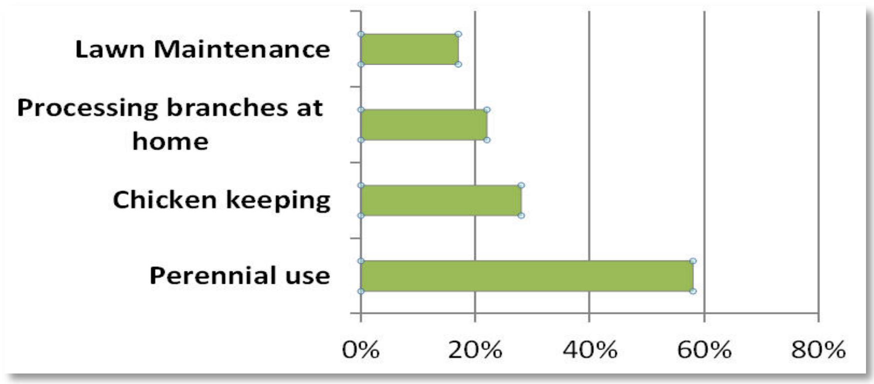

Figure 2: The percentage of inhabitants that are applying other closed loop techniques to reduce their organic waste.

The amount of people that practises (our) specific food loss reducing methods (= the most recent Biocycle at Home-theme) we do not know yet.

Almost 75% applies one or another biocycling technique to process organic waste at home, and more than 75% is planning to do so.

The unique co-operation model with authorities on different levels, a network of well trained teachers and trained volunteers (Master Composters / Biocycle Volunteers) who sensitize citizens was and still is successful.
The last years, we see that citizens still want to engage themselves and like to co-operate in organisations, but they want to do it in a more noncommittal and trend-sensitive way. Social media and direct action are playing an important role in this. The lifelong commitment that we saw in former days is slowly disappearing.

4. Conclusion and outlook

The 2-fold policy followed, clearly has a positive result, but integrating the experience and knowledge of our original volunteers in the new ways of communicating and community creating, is the challenge for the future, ànd at the same time it is the assurance that the biocycle message is alive ànd meaningful!
What will bring the future for the ‘BioCycle at Home’ in Flanders?
The next years we will focus on some potential, namely:

- Home Composting: research showed that 59% of the people are composting at home or are planning to do that on the short term. Getting to that 59% home composters is one target.
- Idem ditto what concerns Closed Loop Gardening. 75% is planning to implement one or another BioCycle theme in his own garden. So 75% or more is our goal.
- A gradual change of tasks and communication content requires a change of name. That’s why we will gradually evolve to the a new terminology (from Master Composters to BioCycle Volunteers).
- Trying to introduce the principles of the BioCycle into other (garden-, environment- and nature related) npo’s
- Taking into account and flexibly respond on a constantly changing legislation
- Further expanding our communication and sensitizing items

We will focus on the future, but remembering and taking into account some preconditions, namely:

- That all individuals must / want to be sensitized, convinced in a different way (leaflet, education, google-discovering …)
- That Home composting is a valuable first step towards an improved relationship with environment
- That BioCycling at Home is part of a sensitizing process in order to change consumption behaviour (f.i. food loss).
- The roadmap on circular economy http://ec.europa.eu/smart-regulation/roadmaps/index_en.htm.

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References

[1] http://www.ovam.be/achtergrond-van-het-materialendecreet
[2] OVAM, Vlaco, 2003. - De kwaliteit van ‘thuis’compost. 136p.
[3] OVAM, 2012. - Evaluatieonderzoek materialenkringloop gft- en groenafval. 123p.

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