Effects of Biogas Residues on Yield Formation and Soil Organic Carbon Stocks

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zur Erlangung des akademischen Grades

Doktor der Ingenieurwissenschaften
- Dr. Ing. –

genehmigte Dissertation

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Tag der wissenschaftlichen Aussprache: 03. Mai 2018

Berlin 2018

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Abbreviations

AOM	biological active soil organic matter
BFA	biogas fingerprint
BGP	biogas plant
BGR	biogas residue
BPU	biogas production unit
C	carbon
C_{org}	organic carbon
C_{rep}	carbon reproduction flux from FOM to SOM
CANDY	Carbon and Nitrogen Dynamics
CCB	Candy Carbon Balance
CH_4	methane
CO_2	carbon dioxide
COP 21	21th Conference of the Parties in Paris in 2015
DM	dry matter
DNDC	DeNitrification DeComposition
FOM	fresh organic matter
LTS	long-term stabilized soil organic matter with no turnover during simulation time
N	nitrogen
NH_4	ammonium
NO_2	nitrate
OM	organic matter
RothC	Rothamsted Carbon Model
SOC	soil organic carbon
SOM	soil organic matter
SSM	SSM: stabilized soil organic matter
TGW	thousand grain weight
TKN	total Kjeldahl nitrogen
w/w	weight by weight

Summary

The United Nations Framework Convention on Climate Change (UNFCCC) reached an agreement to combat climate change by keeping a global temperature rise this century less than 2 °C above pre-industrial levels. One of the measures to achieve this goal is to increase the share of renewable energy, e.g. the share of biogas which is produced by anaerobic degradation of biomass. During this process microorganisms transform biomass into methane (CH_4) and carbon dioxide (CO_2). The remaining organic and inorganic matter is a secondary product which is called biogas residue (BGR), which is usually used in the agriculture as organic fertilizer. As during the anaerobic digestion process about 60% of carbon is transformed into CH_4 and CO_2, it cannot be returned to the soil to rebuild soil organic matter (SOM). This supports the presumption that compared to undigested fertilizer this carbon extraction from the agricultural system can lead to soil carbon decrease and soil degradation. Soil organic carbon (SOC) is a part of SOM which is an essential component of soil that improves the soils functions thereby supporting several ecosystem services. Although BGRs are known as a predominantly positive crop fertilizer and soil conditioner, the effects on SOC are rarely studied, because the application of these substances within agriculture is a relatively new concept.

As biogas production expands globally, it is necessary to understand its environmental consequences. In this context, the overarching goal of this thesis was to contribute to an improved understanding of the BGR effects in agricultural systems. The objectives of this thesis were:

a) to identify the effects of separated BGRs from industrial and municipal wastes on yield formation and grain yields,

b) to determine the impact of the implementation of biogas production on the SOC on the farm scale,

c) to identify the areal demand for biogas production and its impact on carbon fluxes on the landscape scale.

To reach these objectives in the first step a field experiment was conducted to derive the information about the effect of different BGRs (treated / untreated, agricultural / industrial and municipal waste) on soil and crops (grain yields, yield formation). In the next step, a farm scale study was performed to determine the impact of the implementation of biogas production

implementation on the SOC. Here the focus was on agricultural and untreated BGRs due to their relevance and wide distribution. Therefore a system with a farm and an agricultural biogas plant (BGP) was regarded. A process model (CANDY: Carbon and Nitrogen Dynamics) was applied to make predictions for the SOC development over time. Incubation experiments were used for estimation of SOM turnover parameter for BGRs. In the next step the estimated parameters were applied on a landscape scale (Saxony). Here a quantitative and qualitative analysis of the SOC fluxes which were changed through BGP was conducted. Additionally a methodology of a BGP areal demand calculation, so called "biogas fingerprint area" was worked out.

The following results were obtained:

a) the initial short-term fertilization effect of soil treated with BGRs from industrial and municipal wastes is similar to agricultural BGRs and mineral fertilizer. Liquid fractions of BGRs caused less plants per m² than solid or complete BGRs, what was attributed to phytotoxic potential of the liquid fractions on the germination. Despite that, liquid fractions caused higher total grain yields than solid fractions. Here, barley compensated the disadvantages at the beginning during the vegetation period with higher number of ears per plant and grains per ear.

b) the values for the SOM turnover parameters for the process model CANDY were determined and a linear relationship between those parameters and chemical properties of BGRs (pH and C/N_{org}) was found. The findings at the farm scale suggest that the replacement of undigested organic fertilizers with BGR did not lead to a decrease in SOC within ten years of BGR application. Furthermore, the model indicated that, despite carbon removal during anaerobic digestion, the SOC did not decrease under the tested cropping conditions (until 2050).

c) in Saxony, BGPs can be operated sustainably with regard to SOC recycling. The "biogas fingerprint area" which is required to supply the BGPs and dispose of their BGRs, is on average only approximately the fifth part of the agricultural land. Overall, the total C flux into the soil increased in the observed time and the contribution of different C sources changed. Areas affected by biogas production showed higher SOC reproduction rates than the surrounding agricultural land due to high contributions from BGR and crop residues.

In conclusion, the present thesis shows that BGRs can contribute to productivity as well as to the maintenance of SOC in agricultural cropping systems.

Zusammenfassung

Während der Klimarahmenkonvention der Vereinten Nationen einigte man sich auf die Bekämpfung des Klimawandels und die Einhaltung des globalen Temperaturanstiegs in diesem Jahrhundert auf unter 2°C des vorindustriellen Niveaus. Eine der Maßnahmen, um dieses Ziel zu erreichen, ist die Erhöhung des Anteils der erneuerbaren Energien, z.B. den Anteil des Biogases, das durch anaerobe Vergärung der Biomasse erzeugt wird. Während der anaeroben Vergärung setzen die Mikroorganismen die Biomasse zu Methan (CH_4) und Kohlenstoffdioxid (CO_2) um. Die verbleibende organische und anorganische Substanz wird Gärrest genannt. Da während der anaeroben Vergärung ca. 60% des Kohlenstoffs zu CH_4 und CO_2 umgewandelt werden, kann er nicht zum Aufbau der organischen Bodensubstanz zurückgeführt werden, die als eine essentielle Komponente im Boden die Bodenfunktionen verbessert und damit diverse Ökosystemleistungen unterstützt. Die Entnahme des Kohlenstoffs aus dem landwirtschaftlichen System durch die Vergärung bekräftigt die Annahme, dass im Vergleich zu unvergorenem Dünger dies zu einer Abnahme des organischen Bodenkohlenstoffs (Teil der organischen Bodensubstanz) im Boden und damit zu einer Bodendegradierung führen kann. Gärreste sind zwar als Pflanzendünger und Bodenverbesserer mit überwiegend positiven Eigenschaften bekannt. Nichtdestotrotz, ist ihre Anwendung in der Landwirtschaft ein relativ neues Konzept.

Da die Biogasproduktion global expandiert, ist es notwendig Ihre Auswirkungen auf die Umwelt zu verstehen. In diesem Zusammenhang war das übergreifende Ziel dieser Doktorarbeit zu einem verbesserten Verständnis des Gärresteinflusses in landwirtschaftlichen Systemen beizutragen. Ziele dieser Arbeit waren:

a) Bestimmung der Wirkung der separierten Gärreste aus industriellen und kommunalen Abfällen auf Kornertrag und Ertragsstruktur,

b) Bestimmung des Einflusses der Biogasproduktion auf organischen Bodenkohlenstoff auf der Betriebsskala,

c) Bestimmung des Flächenbedarfs der Biogasproduktion und ihres Einflusses auf die Kohlenstoffflüsse auf der Landschaftsskala.

Um diese Ziele zu erreichen, wurde im ersten Schritt ein Feldexperiment angelegt, um Information über den Effekt verschiedener Gärreste (separiert / unsepariert, landwirtschaftliche / industrielle und kommunale Abfälle) auf den Boden und Pflanzen (Kornertrag, Ertragsstruktur) zu bekommen. Im nächsten Schritt, wurde eine Studie auf der Betriebsskala durchgeführt, um den Einfluss der Biogasproduktion auf den organischen Kohlenstoff im Boden zu untersuchen. Hier wurde der Fokus auf landwirtschaftliche, unseparierte Gärreste gelegt aufgrund ihrer Relevanz und deren Verbreitung. Hierfür wurde ein System mit einem landwirtschaftlichen Betrieb und einer Biogasanlage betrachtet. Ein Prozessmodell (CANDY: Carbon and Nitrogen Dynamics) wurde angewandt, um Vorhersagen zu treffen. Inkubationsexperimente wurden verwendet, um für Gärreste die Parameter des Umsatzes der organischen Bodensubstanz zu bestimmen. Im nächsten Schritt wurden die bestimmten Parameter auf der Landschaftsskala angewandt. Hier wurde eine quantitative und qualitative Analyse der Kohlenstoff-Flüsse, die durch die Biogasanlage verändert wurden, durchgeführt. Zusätzlich wurde eine Methodik zur Berechnung des Flächenbedarfs einer Biogasanlage, der so genannte „Biogas Fingerabdruck“ ausgearbeitet.

Folgende Ergebnisse wurden erzielt:

a) kurzfristige Düngewirkung der separierten Gärreste aus industriellen und kommunalen Abfällen ist vergleichbar mit der Düngewirkung der landwirtschaftlichen Gärreste sowie des Mineraldüngers. Flüssigphase der Gärreste führte zu weniger Pflanzen pro m² verglichen mit der Festphase oder dem gesamtem Gärrest, was dem phytotoxischen Potential der Flüssigphase auf die Keimung zugeschrieben werden kann. Nichtdestotrotz, führte Flüssigphase zu höherem Kornertrag als die Festphase. Die Gerste kompensierte die ursprünglichen Nachteile im Laufe der Vegetationsperiode durch die höhere Anzahl von Ähren pro Pflanze sowie Körnern pro Ähre.

b) Parameter des Umsatzes der organischen Substanz für das Prozessmodell CANDY wurden bestimmt und ein linearer Zusammenhang zwischen ihnen und den chemischen Gärresteigenschaften (pH und C/N_{org}) wurde gefunden. Die Ergebnisse auf der Betriebsskala zeigen, dass der Ersatz der unvergorenen organischen Dünger mit Gärresten während eines 10-jährigen Gärrest-Einsatzes zu keiner Abnahme des Kohlenstoffs im Boden geführt hat. Des Weiteren, wurde gezeigt, dass der organische Kohlenstoff im unter getesteten Anbauverhältnissen nicht abnimmt.

c) in Sachsen können die Biogasanlagen im Hinblick auf den organischen Bodenkohlenstoff nachhaltig betrieben werden. Der Flächenbedarf, welcher für die Versorgung der Biogasanlage und Gärrestausbringung notwendig ist, nimmt durchschnittlich nur ca. ein Fünftel der landwirtschaftlichen Fläche ein. Der gesamte Kohlenstofffluss in den Boden ist in der Untersuchungszeit angestiegen und der Beitrag verschiedener Kohlenstoffquellen hat sich geändert. Flächen, die durch die Biogasproduktion betroffen sind, zeigten aufgrund des hohen Beitrags der Gärreste und der Koppelprodukte höhere Kohlenstoff-Reproduktionsraten verglichen mit der umliegenden landwirtschaftlichen Fläche.

Zusammenfassend zeigt die vorliegende Doktorarbeit, dass Gärreste zu der Produktivität sowie der Erhaltung des organischen Kohlenstoffs im Boden in landwirtschaftlichen Anbausystemen beitragen können.

1 General introduction

1.1 Background and rationale

The member countries of the United Nations Framework Convention on Climate Change (UNFCCC) agreed in 1992 to protect the climate system for the benefit of present and future generations of humankind (United Nations, 1992). At the 21th Conference of the Parties (COP 21) in Paris in 2015, they reached an agreement to combat climate change (United Nations, 2015). The central aim of the Paris Agreement is to strengthen the global response to the threat of climate change by keeping a global temperature rise this century well below 2 °C above pre-industrial levels, which has been widely accepted as a threshold to 'dangerous' climate change (United Nations, 2015). One of the ways to achieve this goal is to reduce greenhouse gas emissions which contribute to global warming (United Nations, 2015). To reach this aim a number of measures is recommended such as the enhancement of energy efficiency, protection and enhancement of sinks and reservoirs of greenhouse gases, promotion of sustainable forest management and agricultural practices, development and increased use of new and renewable forms of energy. In this context at the COP 21 the international initiative "4 per 1000" was launched with an aspiration to increase global soil organic matter (SOM) stocks by 4 per 1000 (or 0.4%) per year as a compensation for the global emissions of greenhouse gases by anthropogenic sources (Minasny et al., 2017).

Thus, the EU Member States have committed themselves to increase the share of renewable energy in the EU`s energy mix to 20% and reduce GHG emissions by 20% by 2020 (European Commission, 2010). The climate protection goal of the German government also refers to the goal of the EU to reduce emissions of GHG by 80–95% in Germany by 2050 (in comparison to 1990) (BMUB, 2016). Therefore among other measures Germany's current policy strives to increase the contribution of renewable energy resources to substitute fossil energy resources in order to decrease CO_2 emissions but also to become less dependent on imports of fossil fuel.

Bioenergy is of all renewable resources the most CO_2 neutral and plays a central role in the accomplishment of aforementioned goals (European Environment Agency, 2013). In Germany biomass is the most important renewable energy source with a share of around 56.5% of the total renewable energy resources (FNR, 2015). Biogas which is produced out of biomass is an important component for energy production from bioenergy. It is a versatile renewable energy source, which can be used for replacement of fossil fuels in power (15.4% in 2016) and heat production (10.3% in 2016), and it can be used also as gaseous vehicle fuel (Federal Ministry for Economic Affairs and Energy, 2016). Additionally methane-rich biogas (biomethane) can replace natural gas as a feedstock for producing chemicals and materials (Weiland, 2010).

Energy production from biogas is well-known for a long time, but considerable use of biogas has grown since the 1990s (FNR, 2013; Nkoa, 2014). Massive increase of the biogas production in Germany started with the Renewable Energy Sources Act 2000 where renewable energies have become a central pillar of Germany's energy supply. In Germany, biogas is predominantly generated in agricultural biogas plants within a farming system. These biogas plants provide renewable energy and combined heat and power (Oehmichen and Thrän, 2017). Germany is the largest biogas producer in the European Union with almost 8,700 biogas plants installed in 2016 (Daniel-Gromke et al., 2017a, 2017b). The main types of feedstock for anaerobic digestion include maize silage, grass silage, and animal slurries, as well as domestic organic waste and waste from food industries, (Möller & Müller, 2012; Scheftelowitz et al., 2016). During this process, the microorganisms transform biomass into methane and carbon dioxide.

The remaining organic and inorganic matter that was not transformed into biogas is a secondary product which is called biogas residue (BGR). This BGR is mineralized and the availability of the nutrients is increasing during anaerobic digestion. That is why the BGRs are valuable organic fertilizers and are used for closing nutrient cycles on agricultural soils (Möller, 2015; Tambone et al., 2010).

While biogas production has the potential to improve nutrient cycling, the carbon which is burned as methane (CH_4) and carbon dioxide (CO_2) is lost from the agricultural system and cannot be returned to the soil to rebuild soil organic matter (SOM). During the anaerobic digestion about 60% of carbon is transformed into CH_4 and CO_2. This supports the presumption that compared to undigested fertilizer this carbon extraction from the agricultural system will lead to soil carbon decrease and soil degradation. Moreover, a shifting demand in the agriculture products may lead to changes inter alia of crop rotation with a higher rate of energy crops and reduced recycling of by-products (straw and beet leaves) into soils. Under such conditions the maintenance and improvement of soil quality in cropping systems may become critical to sustain agricultural productivity and environmental quality for future generations (Franko et al., 2015).

As biogas production expands globally, it is necessary to understand its environmental consequences. A mandatory precondition for an intensive usage of BGRs in agriculture is their environmental harmlessness and suitability for sustainable soil management. Therefore the

BGRs should be investigated related to their effect on yield formation and soil organic carbon (SOC) stocks and their reaction in soils should be known. Unlike manures, composts and sewage sludge which have been extensively studied in the past, research on BGRs has not yet provided all required knowledge (Nkoa, 2014). To close this knowledge gap, the following problems will be addressed within this thesis.

1.2 Treated BGRs from industrial and municipal waste as fertilizer on a field scale

In recent years the mainly investigated BGRs were from agricultural biogas plants which use animal excrements and energy crops as substrate mix for biogas production. In Germany only 8% of all biogas plants use organic wastes like source-separated household waste, food waste, kitchen waste, sewage sludge and green waste for the fermentation process (Thrän et al., 2015b). However with the goal to substitute fossil energy sources and reduce GHG emissions wastes like industrial and municipal wastes will become more attractive.

With the very few exceptions, the literature on the short-term effects of BGR on soil properties has consistently noted the improvement of the quality of soils amended with anaerobic BGRs (Nkoa, 2014). Biogas residues contain important plant nutrients like nitrogen (N), phosphorus (P), potassium (K) and magnesium (Mg) (Arthurson, 2009). The total nutrient content of substrates can change upon anaerobic fermentation due to losses (N, S, Cd, Zn, Mg) or accumulations (Fe, Mn) (Zirkler et al., 2014). However, losses of N are negligible compared to C losses, thus, the fermentation process leads to decreasing C/N ratios (Möller et al., 2008). Furthermore, anaerobic digestion increases the availability of N (NH_4-N) due to the breakdown of organically bound N during the process (Möller et al., 2008; Weiland, 2010). Overall, BGRs are known to increase microbial activity in soil after being applied (Petersen et al., 2003). Furthermore, BGRs can improve soil physical properties like soil structure and water holding capacity (Nkoa, 2014). Additionally, several studies evaluated the fertilization effect of BGRs (Abubaker et al., 2012; Odlare et al., 2011; Tambone et al., 2010). The authors found, that unseparated BGRs had good fertilizing properties due to the high content of plant available nutrients (NPK).

Usually such BGRs are used in agriculture without any further treatment like separation of solid and liquid phase. However, before being used in agriculture, there is a possibility to pre-treat BGRs. Solid–liquid separation is an established physical BGR treatment performed in 7% of the biogas plants in Germany (Möller and Müller, 2012a). Such pre-treatment permits a better handling of the fertilizer with regards to storage, transport and spreading. Furthermore solid–liquid separation of fermentation residues leads to an enrichment of P in the solid fraction while most of the N is located in the liquid fraction. Finally, the emission of odor may also be reduced by this pretreatment (De la Fuente et al., 2013).

Nevertheless, the information about the effects of utilizing different types of separated and unseparated BGRs from industrial and municipal wastes on crop yields is rare. Published literature about BGRs from industrial and municipal wastes refer to soil chemical, physical, microbial properties and yields, but not to yield formation which can be important for understanding of the weather effects on nutrient availability and crops. The knowledge about the fertilizing effect of separated BGRs from industrial and municipal wastes in combination with a nitrification inhibitor is also rare. Furthermore, studies on fertilizer value and environmental impacts of BGRs were primarily conducted in the laboratory and on agricultural BGRs. Field studies investigating the effect of the application of BGRs on crop yields are scarce. This applies especially to separated BGRs from industrial and municipal waste.

Therefore, the first objective of this thesis was to contribute to the knowledge about the yield formation affected by fertilization with separated BGRs from industrial and municipal wastes. Differently treated and untreated BGRs from four full-scale biogas plants in Germany were used in a field experiment.

The hypotheses were:

1) separated BGRs sourced from biodegradable household and industrial wastes result in similar yields to those obtained by mineral fertilizers;

2) the liquid fraction of BGRs results in higher yields than the solid fraction due to its high nitrogen (N) availability and

3) usage of a nitrification inhibitor leads to higher yields compared to the same BGR without inhibitor.

The results of this experiment are presented in chapter 2.

1.3 Impact of untreated agricultural BGR on SOC on a farm scale

The intensive biogas production and consequently increased usage of BGRs in Germany started after the ratification of Renewable Energy Sources Act in 2000. Long-term experiments for the assessment of SOC change under different soil management conditions were not possible due to the short period of BGRs usage. However, predictions of the impact and consequences of BGRs usage or crop rotation change on SOM are necessary.

The increase of bioenergy share for energy production is connected with increase of biomass demand from agriculture, biogas production and consequently increased supply of BGRs in agriculture as fertilizer. A such big demand in the agriculture may lead to changes inter alia of crop rotation with a higher rate of energy crops and reduced recycling of by-products (straw and beet leaves) into soil. Under such conditions the maintenance and improvement of soil quality in terms of sufficient SOM in cropping systems is essential to sustaining agricultural productivity and environmental quality for future generations.

One potential impact of BGRs deployment is the storage or release of SOC which is a major component of SOM. Soil organic matter is fundamental to soil function and its ecosystem service in particular the sequestration of carbon (Podmanicky et al., 2011; Campbell & Paustian, 2015; Yigini & Panagos, 2016). Soil organic carbon content is one of the most important indicators of soil fertility and agronomic sustainability (Reeves, 1997). It improves the soils functions thereby supporting some ecosystem services like increasing productivity and crop quality, improving water and nutrient retention, decreasing runoff of both sediment and pollutants, and increasing soil biodiversity (Janzen, 2004; Lal, 2004a; Reeves, 1997). Also it is the most often reported attribute from long-term studies which have consistently shown the benefit of manures, adequate fertilization, and crop rotation on maintaining agronomic productivity by increasing carbon inputs into the soil (Monteleone et al., 2015; Reeves, 1997).

Soil organic carbon storage is sensitive to climate, land use and soil management. Indeed, inappropriate soil management or cropping practices can cause a decline in SOM and potentially lead to emissions of C into the atmosphere (Reeves, 1997; Lal, 2004; Yigini & Panagos, 2016). Approaches are required to support the sustainable maintenance and improvement of SOC storage as a means to buffer the emissions of greenhouse gases (GHG) which contribute to global warming (Smith, 2004; Vidal Legaz et al., 2017). Thus, an understanding of changes in SOC under usage of BGRs is essential for thorough environmental impact analyses of the biogas production.

Effects of the BGR application on crop yield, soil chemical, physical and microbial properties were already studied in small scale experiments (Fouda et al., 2013; Sänger et al., 2014). Nevertheless there is still little information available on field-scale about long-term effects of BGRs on SOC level (Möller, 2015). Two studies observed the impact on SOC. After 8 years the SOC in BGR treated plots was similar to those treated with mineral fertilizer (Odlare et al., 2011). Wentzel et al. (2015) showed that long-term application (up to 25 years) of the BGR had no specific effects on SOC.

Hence, the question is still open for discussion whether it is possible to use biogas as a bioenergy source without depleting soil carbon stocks. BGRs application adds carbon and nutrients to the soil, thus affect SOM and long-term soil fertility directly. On the another hand biogas production could have an indirect effect induced by changes in the entire cropping system, e.g. changes in crop rotation or implementation of new energy crops. By now it is not entirely clarified whether the most influence on SOC comes directly from BGRs or indirectly from bioenergy induced management changes (Möller, 2015).

A general problem is that our knowledge of soil properties often originates from small soil samples in laboratory experiments. The role of smaller units compared to larger units is not always well understood and we have to find out how we can combine information, which represents different scales. It is important to be able to transfer our knowledge gained in laboratory in the field. That is why one chapter of this thesis (chapter 4) focuses on the farm scale and uses a predictive modeling as a tool. Several process-based agroecosystem models, such as DNDC (DeNitrification DeComposition) and RothC (Rothamsted Carbon Model), have been widely utilized for quantifying SOC sequestration and are capable for determining the effects of organic fertilizers on SOC dynamics (Li et al., 2016).

The carbon turnover sub-model which is integrated in the mathematical agroecosystem model CANDY (Carbon And Nitrogen DYnamics) and the SOM model CCB (Candy Carbon Balance) was used (Franko et al., 2011; Franko et al., 1995). CANDY processes site-specific information on soils, crops, weather, and land management to compute stocks and fluxes of carbon in the topsoil of agricultural fields. Here, the SOM pools have conceptual character (Franko et al., 1995). Soil organic matter is subdivided into three pools:

- biological active soil organic matter (AOM) where mineralization occurs,
- stabilized soil organic matter (SSM) that represents the passive but decomposable part of the SOM and
- long term stabilized soil organic matter (LTS) that is considered inert.

Fresh organic matter (FOM) is not a defined homogeneous pool but consists of n pools. Matter exchange between the active SOM (AOM) and stabilized SOM is assumed by the model. Decomposed FOM carbon is used to generate new SOM. All C turnover processes are formulated as first-order reactions.

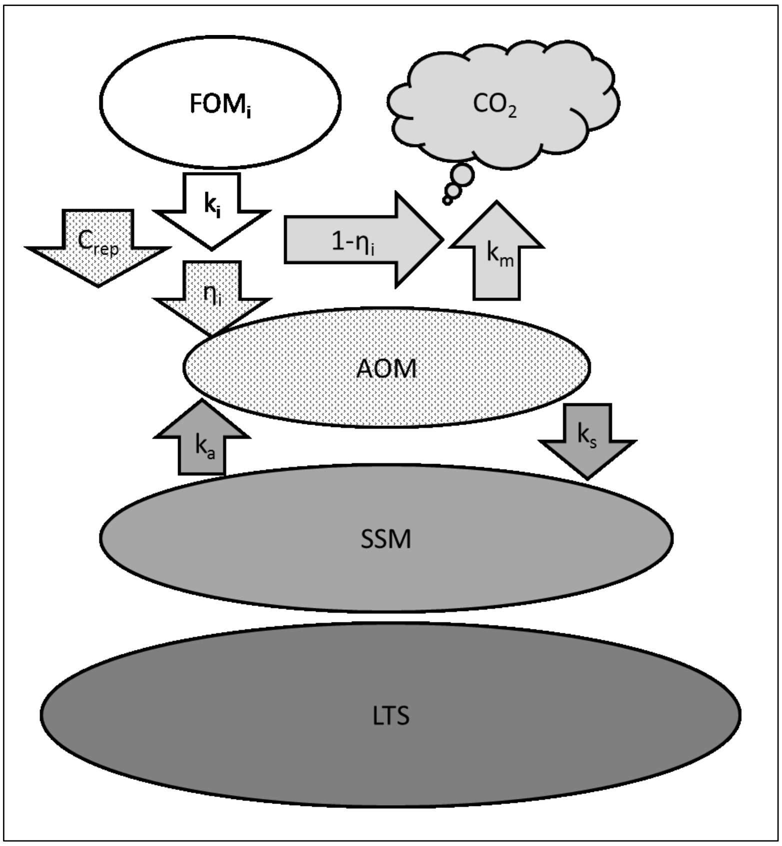

Figure 1-1 Block scheme of the soil organic matter (SOM) turnover model with pools (ovals) and fluxes (arrows). C_{rep}: carbon reproduction flux from fresh organic matter (FOM_i) to SOM. CO_2: release of carbon dioxide. AOM: biological active soil organic matter. SSM: stabilized soil organic matter. LTS: long-term stabilized soil organic matter with no turnover during simulation time. k_i (k in text), k_a, k_s, k_m: turnover coefficients. η_i (η in text): synthesis coefficient (adapted from *Franko et al.* (2011)).

The decomposition of FOM ($\frac{dC_{fom}}{dt}$is determined by its turnover coefficient *k (Franko et al., 2011)*. Thus, higher k values indicate a higher velocity of FOM decay.

$$\frac{dC_{fom}}{dt} = \acute{C}_{fom} = -kC_{fom} \quad (1)$$

The relation of the SOM production to the FOM decay is given by the synthesis coefficient η. Higher η values indicate higher contributions to SOM.

The amount of matter flowing from FOM into AOM is called $C_{rep.}$

$$\acute{C}_{rep} = \acute{C}_{fom}\eta = -k\eta C_{fom} \quad (2)$$

The turnover of the AOM (C_a) consists of the reproduction flux from FOM, the mineralization to CO_2 and matter exchange with SSM pool (C_s),

$$\dot{C}_a = \dot{C}_{rep} - (k_m + k_s)C_a + k_a C_s \quad (3)$$

$$\dot{C}_s = k_s C_a - k_a C_s \quad (4)$$

with set turnover coefficients k_a = 0.00032 d^{-1}, k_m = 0.00556 d^{-1} and k_s = 0.0009d^{-1} (Franko et al., 2011).

For process modeling it is crucial to define the carbon turnover parameters of BGR degradability and their efficiency for creating new SOM which are used in most SOC models (Post et al., 2008; Smith et al., 1997).

Therefore, incubation experiments over a period of several weeks are usually required to derive the carbon turnover parameters (Post et al., 2008). They are time consuming, and in practical applications, it is not possible to perform incubation experiments regularly. Furthermore, the chemical composition of BGRs is very heterogeneous in terms of the composition of plant nutrients and organic matter (Weiland, 2010; Zirkler et al., 2014). Therefore, a specific parameter set may be required for each BGR.

The next objective of this thesis was to determine on farm scale the impact of biogas production implementation on SOC stock. However, data preparation and proper parameterization of BGRs are one of challenges for realistic results. As no parameters for BGRs were available for the process model which I wanted to use, it was an additional objective to derive BGR carbon turnover parameters for process modeling.

I assumed that it is possible to relate the parameters identified from incubation experiments to some easily measurable BGRs properties. Therefore the carbon mineralization rates from an incubation experiment were used and applied for the organic matter turnover model in inverse mode to determine the SOM turnover parameters for several BGRs (Chapter 3).

These resulting parameter values were applied using the model CANDY on a farm scale. The hypotheses were:

1) the replacement of undigested organic fertilizers with BGRs will lead to SOC decrease due to carbon offtake during the anaerobic digestion process;

2) additionally the implementation of the biogas production will lead to a change of the cropping system, with impact on carbon sequestration.

Therefore data from a representative farm in Central Germany were collected. The soil management practices before and after the installation of the biogas plant were evaluated according to carbon fluxes using CANDY.

The results of this investigation are described in chapter 4.

1.4 Impact of untreated agricultural BGRs on SOC on a regional scale

The analysis at the farm scale was restricted to some boundary conditions: the studied region is only one example and it is characterized by loess soils, which have very beneficial agricultural properties, such as good air and water regimes. These conditions could mask the effects of BGR fertilization and the cropping system, which may be more apparent on sandy soils. Furthermore, the analysis was restricted by the low number of SOC observations per field used for the validation. More validation points would improve the results.

Thus, to improve the understanding of the impact of biogas production on SOC the aforementioned approach in areas with different soils and at larger scales (landscape) was applied. Another field scale study for the region of Central Germany already identified a number of hot spots where the usage of carbon may raise a conflict between sustaining SOC and producing bioenergy (Franko et al., 2015). Furthermore, the expansion of bioenergy production has resulted in an adaption of the agricultural management (e.g. cultivated crops, BGR application instead of slurry), which in turn has changed the SOM reproduction within agricultural landscapes.

If the plant material originates from crops produced specifically for that purpose, a biogas plant has a 'fingerprint' that is defined by the area of arable land needed for the production of these energy crops and the area for distributing the BGRs (see also chapter 5.5). The BGR can be used to fertilize these lands (reducing the need for carbon and nitrogen fertilizers), and the crop land can be managed to serve as a carbon sink, capturing atmospheric CO_2.

A study on the scale of the Federal state of Saxony was conducted, to describe the carbon fluxes which are influenced through a biogas plant. This was important to be able to assess C_{rep} flux changes in the related soil region due to the BGR.

The focus was on the ecological impact of biogas production in Saxony, with a specific interest in the long-term effect of BGR-fertilization on carbon storage within the biogas fingerprint of a BGP. Therefore nutrient fluxes were studied using the CCB (CANDY Carbon Balance) model, which processes site-specific information on soils, crops, weather, and land management to compute stocks within the fingerprints of BGPs of different sizes. Additionally the effect of the substrate mix for the BGP on the carbon dynamics of the soil was studied.

The hypotheses were:

1) In the case study region of Saxony the specific agricultural area required for each biogas plant can be determined and characterized with respect to its spatial extent and the related soil carbon fluxes.

2) This area demand can be used as an indicator to assess the influence of biogas production on SOC reproduction.

The results of the regional analysis are described in chapter 5.

1.5 Outline of the thesis

The overarching goals of this thesis were to contribute to an improved understanding of the impact of the biogas production on soil, especially on SOC and yield formation. The three specific objectives of the thesis were to

a) to identify the effect of separated BGRs from industrial and municipal wastes on yield formation and grain yields (objective 1, Figure 1-2),

b) to determine the impact of the implementation of biogas production on the SOC on the farm scale (objective 2, Figure 1-2),

c) to identify the areal demand of the biogas production and its impact on carbon fluxes on the landscape scale (objective 3, Figure 1-2).

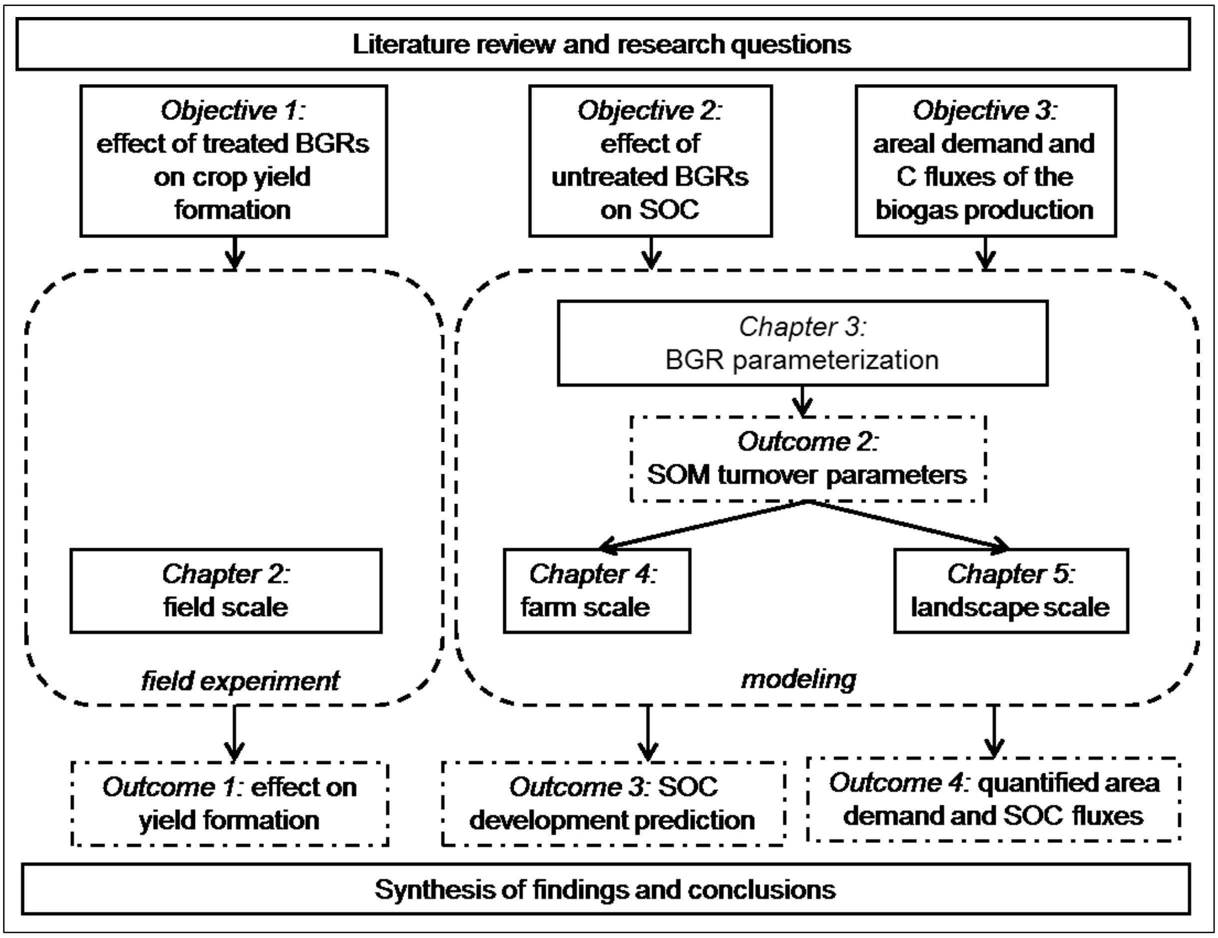

Figure 1-2 concept of the thesis

To reach these objectives this thesis was based on two different methods: field experiment and process modeling (Figure 1-2, dashed line).

Biogas residues are in comparison to other organic fertilizers a relatively new substrate, thus the knowledge about its impact on soil is relatively restricted. Thus, in the first step a field experiment was conducted to derive the information about the effect of different BGRs (treated / untreated, agricultural / industrial and municipal waste) on the soil and on the crops. Among other parameters the environmental harmlessness was evaluated by means of the measurable reaction of the crops on the BGRs: yield formation and crop grain yield.

The investigations on farm scale focused only on the agricultural and untreated BGRs due to their relevance and wide distribution compared to industrial and municipal waste and separated (treated) BGRs. Larger scale as well as a longer time period were chosen to determine the impact of the biogas production implementation on the soil SOC. Therefore I regarded a system with a farm and an agricultural BGP. It was assumed, that a part of the fields and matter fluxes are influenced by a BGP, another part is unaffected.

Soil organic carbon data from a farm as well as own measurements were used to reach the objectives. A model application was necessary to validate the measurements and to make predictions for the future. Hence, there was a need for BGRs parameterization because the BGRs were until now not described for predictive process modeling.

Incubation experiments were used to derive parameters for the modeling of SOM turnover where C mineralization of different BGRs was evaluated. A parameter set for six specific BGRs was calculated. Additionally a methodology was worked out which helps to derive SOM turnover parameters without conducting incubation experiments.

The farm scale study was only one example with very homogeneous soil conditions. Thus, in the next step the estimated parameters were applied on a landscape scale. Here a quantitative and qualitative analysis of the SOC fluxes which were changed through BGP was conducted. Additionally a methodology of a BGP areal demand calculation, so called “biogas fingerprint area” was worked out.

Chapter 1 introduces the background of the thesis and defines the social and scientific questions of the topic. The experimental working package which deals with a field experiment is described in chapter 2. It provides information about the impact of BGRs from industrial and municipal wastes on the crop yield and yield formation in the field experiment.

The modeling working package comprises chapter 3 - 5. In Chapter 3 the methodology is described in detail how to derive SOM turnover parameters for BGRs. Chapter 4 complies all results on farm level that were gained for changes on SOC induced by the establishment of a biogas plant. In chapter 5 the methodology from the chapter 3 and the approach of the chapter 4 are used. Chapter 5 includes study results to SOC fluxes on the landscape level.

Finally, chapter 6 reconsiders the working hypotheses that have been stated in the beginning. In the first place, it provides a final statement on general conclusions, limitations of the thesis and elucidates the future research need in this topic.

2 Initial effects of differently treated biogas residues from municipal and industrial wastes on spring barley yield formation[1]

2.1 Abstract

Soil application of biogas residues (BGRs) is important for closing nutrient cycles. This study examined the efficiency and impact on yields and yield formation of solid-liquid separated residues from biodegradable municipal and bio-waste wastes (bio-waste) in comparison to complete BGRs, nitrification inhibitor, agricultural BGRs, mineral fertilizer and unfertilized plots as control. The experiment was set up as a randomized block design on silt loam Cambisol. Biogas residues from four biogas plants were evaluated. Plants per m², ears per plant, grains per ear and thousand grain weight (TGW) were measured at harvest. Fertilization with BGRs resulted in similar biomass yields compared with mineral fertilizer. Mineral fertilizer (71 dt/ha) and plots fertilized with liquid fraction (59-62 dt/ha) indicated a trend to higher yields than solid fraction or complete BGR due to its high ammonia content. Liquid fractions and fraction with nitrification inhibitor induced fewer plants per m² than corresponding solid and complete variants due to a potential phytotoxicity of high NH_4-N concentration during germination. However, barley on plots fertilized with liquid fraction compensated the disadvantages at the beginning during the vegetation period and induced higher grain yields than solid fraction. This was attributable to a higher number of ears per plant and grains per ear. In conclusion, BGRs from biodegradable municipal and bio-waste wastes can be used for soil fertilization and replace considerable amounts of mineral fertilizer. Our study showed that direct application of the liquid fraction of BGR is the most suitable strategy to achieve highest grain yields. Nevertheless potential phytotoxicity of the high NH_4-N concentration in the liquid fraction should be considered.

2.2 Introduction

The government of Germany enacted legislation to increase the percentage of energy from renewable sources like solar, wind, and bioenergy to 60% of the total energy consumption by 2050 (Federal Ministry for Economic Affairs and Energy, 2014). At present, bioenergy, with a share of 61.8%, is the most important renewable energy source (FNR, 2014). Bioenergy is an

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[1] Prays, N., and Kaupenjohann, M. (2016): Initial Effects of Differently Treated Biogas Residues from Municipal and industrial Wastes on Spring Barley Yield Formation. PLoS ONE 11, e0154232.

essential component due to its broad range of applications and its storage capacity. One of the bioenergy sector's key technologies is the conversion of organic sources to biogas via anaerobic digestion (Thrän et al., 2015a). The advantage of biogas is that it can be produced from nearly all kinds of biological feedstocks, e.g. plant biomass, animal manure, industrial organic waste and organic household waste (Alburquerque et al., 2012a; Herrmann, 2013; Weiland, 2003). The production of energy plants, especially maize, however has caused competition between food, fodder and energy production on arable land. One possible way to address the competition problem is by using other biogenic substrates like organic wastes.

Most of the approximately 8,000 installed biogas plants in Germany are fed with energy plants and animal excrements, while 8% use organic wastes like source-separated household waste, food waste, kitchen waste, sewage sludge and green waste (further bio-waste) (Thrän et al., 2015b).

The average production of biogas from 1 t FM bio-waste is 120 Nm³ (FNR, 2014).This value lies between that for animal excrements and crop silages (maize, grass, rye), indicating that such wastes are suitable substrates for biogas production (FNR, 2014). In terms of the usage of organic residues and wastes it must be taken into consideration that in Germany nearly all waste materials are currently used in well-established utilization processes and only few residues and wastes are not completely utilized (German National Academy of Sciences Leopoldina, 2012). Residues from agriculture and agroindustry are typically used for animal feeding, compost and biogas production, whereas wastes from municipal collection and wastewater treatment are mostly used in incineration and composting plants (German National Academy of Sciences Leopoldina, 2012). As a consequence, the utilization of wastes for energy production is in the majority of cases in strong competition with existing utilization routes. However, there is potential for increasing the efficiency of the usage of wastes. For example, bio-wastes could first be used for biogas production with the non-fermented residues then being composted (Weiland, 2010). Further BGRs can be used as soil amendments due to their high plant available nutrient (N, P, K) contents and considerable amount of residual organic carbon (Möller, 2015; Odlare et al., 2011; Tambone et al., 2010). An additional benefit of such a usage chain is the reduction of the amount of organic waste landfilled (Abdullahi et al., 2008; Odlare et al., 2011; Svensson et al., 2004). The sustainability of the usage chain requires that the BGRs are reused without any negative environmental impacts (Insam et al., 2015).

Studies on fertilizer value and environmental impacts of BGRs were primarily conducted in the laboratory. Field studies investigating the effect of the application of BGRs on crop yields are scarce. This applies especially to separated BGRs from bio-waste.

Tambone et al. (2010) and Odlare et al. (2011) evaluated the fertilizing effect of a large number of unseparated organic waste products inclusive household wastes. The authors found, that unseparated BGRs had a good fertilizing properties due to the high content of plant available nutrients (NPK). Furthermore, unseparated BGRs can improve soil structure and water holding capacity and provide other advantages such as greater microbial stability and hygiene compared with untreated waste (Holm-Nielsen et al., 2009; Möller et al., 2008; Odlare et al., 2011). With the very few exceptions of cases involving feedstock with very high C/N ratio, the literature on the short-term effects of BGR on soil properties has consistently noted the improvement of the quality of soils amended with anaerobic BGRs (Nkoa, 2014).

Haraldsen et al. (2011) studied the fertilizing effect of separated BGRs from source-separated household wastes especially on barley and recommended liquid BGR as a fertilizer for cereal production. De la Fuente et al. (2013) evaluated separated co-digested cattle slurry and suggested a solid-liquid separation followed by composting of the solid fraction and soil application of the liquid fraction as the most suitable strategy for agricultural purposes.

Nevertheless, so far no study has compared the effects of utilizing different types of separated and unseparated BGRs from bio-wastes on yields and yield formation. There is also no data available on the fertilizing effect of separated BGRs from bio-wastes in combination with a nitrification inhibitor.

We therefore compared the fertilizing performance of separated and unseparated BGRs from different sources in a field experiment with spring barley. The hypotheses are: 1) separated BGRs sourced from biodegradable household and bio-waste wastes result in similar yields to those obtained by mineral fertilizers; 2) the liquid fraction of BGRs results in higher yields than the solid fraction due to its high nitrogen (N) availability and 3) usage of nitrification inhibitor leads to higher yields compared to the same biogas residue without inhibitor.

2.3 Material and methods

2.3.1 Biogas residues

The biogas residues (Table 1) were collected from four large-scale biogas plants. Biogas plant 1 utilizes the organic fraction of source separated household waste, green waste and catering waste. The substrate is wet digested for 20 d at 55 °C. Biogas plant 2 uses dry digestion as a processing method for 25 d at >45 °C. The source of the feedstock is the organic fraction of separated household waste and catering waste. In biogas plant 3 sewage sludge, catering waste, expired food and animal by-products were sanitized at 70 °C and wet digested at 37-40 °C for 25 d. All organic waste processing biogas plants had a thermophilic treatment over 55 °C to provide epidemic hygienically harmless products. The fourth biogas plant is an agricultural plant where 90% cattle manure and 10% maize and grass silage are used. The biogas residues from waste material processing biogas plants are separated into a solid and a liquid fraction. Liquid fractions from biogas plants 1 and 2 are usually treated and disposed of in a sewage system. Solid fractions are usually composted. The liquid fraction from biogas plant 3 was used in combination with wheat straw and a nitrification inhibitor Piadin®. Biogas residue from the agricultural biogas plant was utilized without separation. Mineral fertilizer calcium ammonium nitrate was used to compare the fertilizing performance of BGR.

2.3.2 Site characteristics and experimental design

The experiment was carried out at the ca. 5,000 m² experimental field of BioChem agrar GmbH in Motterwitz, Saxonia, Germany (51°11'46.63"N, 12°52'55.15"E). The long-term mean annual temperature is 8.8 °C and the long-term mean annual precipitation is 641 mm.

The growing period was from 04 April to 27 July 2011. The mean temperature during the experiment was 14.5 °C (Figure 2-2). Total precipitation was 253.5 mm. The soil at the experimental site derived from 2 m thick loess layer and is classified as Gleyic Cambisol according to the world reference base classification system (IUSS Working Group WRB, 2014). The soil (6.6% sand, 76.4% silt, and 17% clay) had a plant available water capacity of between 21.4% and 26.2%.

Biogas residues (Table 2-1), mineral fertilizer and unfertilized control plots (CN) were distributed in a randomized block design with three replications and an amended plot size of 4 m² (2*2 m) (Figure 2-1). Distances between the amended plots were 8 m and 15 m (Figure 2-1).

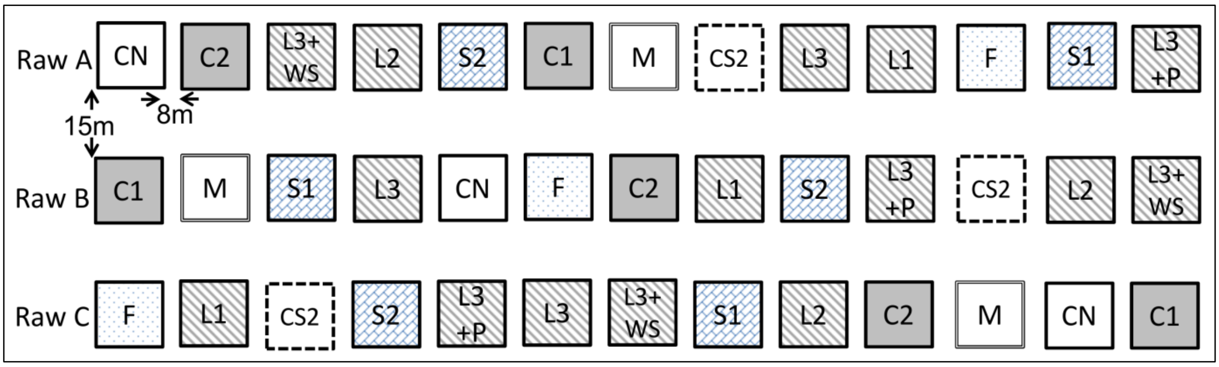

Figure 2-1 Experimental design. Abbreviations are explained in Table 2-1.

All organic and inorganic fertilizers were applied on 4 April 2011 at a rate corresponding to 65 kg N per ha. The amount of fertilizer applied was based on total Kjeldahl nitrogen (TKN) for BGR and on mineral N for mineral fertilizer. Biogas residues were spread over the soil surface manually. After application the organic materials were immediately incorporated into the soil (10 cm depth) with a rotary cultivator. Mineral fertilizer was not incorporated into the soil. Spring barley (*Hordeum vulgare L.)* cultivar "Laverda" was sown on 13 April using 450 seeds per m^2 over the entire 5,000 m^2 field.

2.3.3 Soil sampling

Soil samples (0-20 cm) were collected before BGR application on 04.04.2011 and after application on the 12.04.2011, 29.04.2011, 18.05.2011, 10.06.2011, 04.07.2011 and 26.07.2011. One composite soil sample per plot consisting of 15 individual samples was packed in a portable cooling box in the field. For ammonium and nitrate analyses, the field moist samples were frozen moist (-20 °C) upon arrival at the laboratory. For particle distribution, pH, total carbon (TC) and nitrogen (TN) aliquots were air dried and sieved at 2 mm.

2.3.4 Yield formation

Plants were hand harvested on 27 July 2011 on a subplot of 0.25 m^2 between yellow ripeness and full ripeness. Each plant was tied separately, stored in material bags and dried in the cold air compartment at 25 °C for 1 week. Plants per m^2 as well as ears per m^2 were calculated from the 0.25 m^2 subplots. The number of grains per ear was obtained from the mean grain number of 20 randomly selected ears from each sample. The thousand grain weight (TGW) was calculated by counting and weighing 100 grains three times. Grain yield was calculated as product of the number of grains per ear, ears per m² and TGW.

2.3.5 Biogas residue, soil and plant analysis

Samples of BGR were analyzed within 24 h of collection. The BGR pH was measured potentiometrically directly in the residue. Solid fractions were diluted with distilled water (1:10) prior to measurement. The dry matter of BGR was measured gravimetrically after drying at 105°C (Deutsches Institut für Normung, 2001b) and organic dry matter was estimated after heating at 550°C (Deutsches Institut für Normung, 2001a). Ammonium N was measured photometrically (Deutsches Institut für Normung, 1983). Total Kjeldahl nitrogen was measured after titration (Deutsches Institut für Normung, 1993).

The soil pH (1:2.5; $CaCl_2$) was measured potentiometrically with a glass electrode (METTLER TOLEDO, SevenEasy). Particle size distribution was analyzed according to DIN ISO 11277 (Deutsches Institut für Normung, 2002). The soil water content was determined gravimetrically after the samples were dried at 105 °C for 24 h.

Dried and ground soil, straw and grain samples were used for measurement of TC and TN concentrations using the elementary analyzer (elementar, Vario EL).

The proof of inorganic carbon with 10% HCL showed an absence of lime in the soil. Thus TC is equivalent to the total soil organic carbon. Total N uptake was calculated as a sum of straw N and grain N. Inorganic N (N_{min}) was calculated as the sum of NH_4-N and NO_3-N. Ammonium N and $N0_3$-N concentrations in soil samples were measured photometrically (MERCK SQ 118).

2.3.6 Data treatment

Means and standard deviations were calculated. The effects of different treatments on spring barley yield formation and soil chemical parameters were analyzed by a one-way ANOVA. Normal distribution was tested with a Shapiro test. Least significant difference t-Test (LSD) was used to compare mean values and to assess the significance of the differences between mean values. Effects were considered significant for $p < 0.05$. All statistical analyses were performed using R version 3.0.1 (The R Foundation for Statistical Computing, 2013).

2.4 Results

2.4.1 Composition of biogas residues

The pH in all BGRs was mostly similar (Table 2-1). Dry matter differed due to the wet or dry digestion process as well as due to the liquid or solid separation. After separation the N concentration was higher in the liquid than in the solid fraction.

Table 2-1 Chemical properties of used BGRs and mineral fertilizer. TKN = total Kjeldahl nitrogen; NH_4-N = ammonia nitrogen, FM = fresh matter, DM = dry matter, oDM = organic dry matter, BGP = biogas plant.

BGP	Biogas residue	pH	DM (%FM)	oDM (%DM)	TKN (%DM)	NH_4-N (%DM)	NH_4-N from TKN (%)
1	complete residue (C1)	8.2	1.1	52.8	12.8	1.6	12.4
	liquid fraction (L1)	8.4	0.9	52.7	15.0	8.8	58.5
	solid fraction (S1)	8.4	35.8	69.6	1.6	0.04	2.5
2	complete residue (C2)	8.1	19.5	57.6	3.5	1.7	49.9
	liquid fraction (L2)	8.2	14.4	51.8	4.8	1.5	30.5
	solid fraction (S2)	8.6	38.8	66.8	1.8	0.03	1.7
	composted solid fraction (CS2)	8.6	52.9	54.0	1.4	0.02	1.8
3	liquid fraction (L3)	8.6	6.5	56.9	12.0	1.7	14.3
	liquid fraction + Piadin (L3+P)	8.6	6.5	56.9	12.0	1.7	14.3
	liquid fraction + wheat straw (L3+WS)	8.6	6.5	56.9	12.0	1.7	14.3
4	dairy farm biogas residue (F)	8.6	6.5	56.7	12.0	0.7	6.1
	mineral fertilizer (M)				27.0	13.5	50.0

2.4.2 Effect on soil properties

Chemical parameters of the soil (Table 2-2) were homogeneous throughout all experimental plots at the beginning of the experiment. Total C and TC/TN ratio did not change during the experimental period. The pH tended to decrease, but was not statistically significant. After harvesting, the concentration of N_{min} in soil decreased by about 50% due to nitrogen plant uptake. Soil N_{min} in plots fertilized with liquid fraction increased after BGR application and decreased over the vegetation period (Figure 2-3). Soil N_{min} in plots fertilized with the solid fraction of BGR remained low throughout the observation period (Figure 2-3). We used only BGRs from plant 1 and plant 2 to illustrate the differences between the solid and the liquid phase of BGRs; the same comparisons could not be drawn for plants 3 and 4 because both liquid and solid fractions of BGRs from these plants would be required for this and these were not available. For this reason we did not present data from plants 3 and 4. Soil water content fluctuated according to precipitation (Figure 2-2), without any difference between the experimental plots. The water supply among the plots was equal and without significant differences.

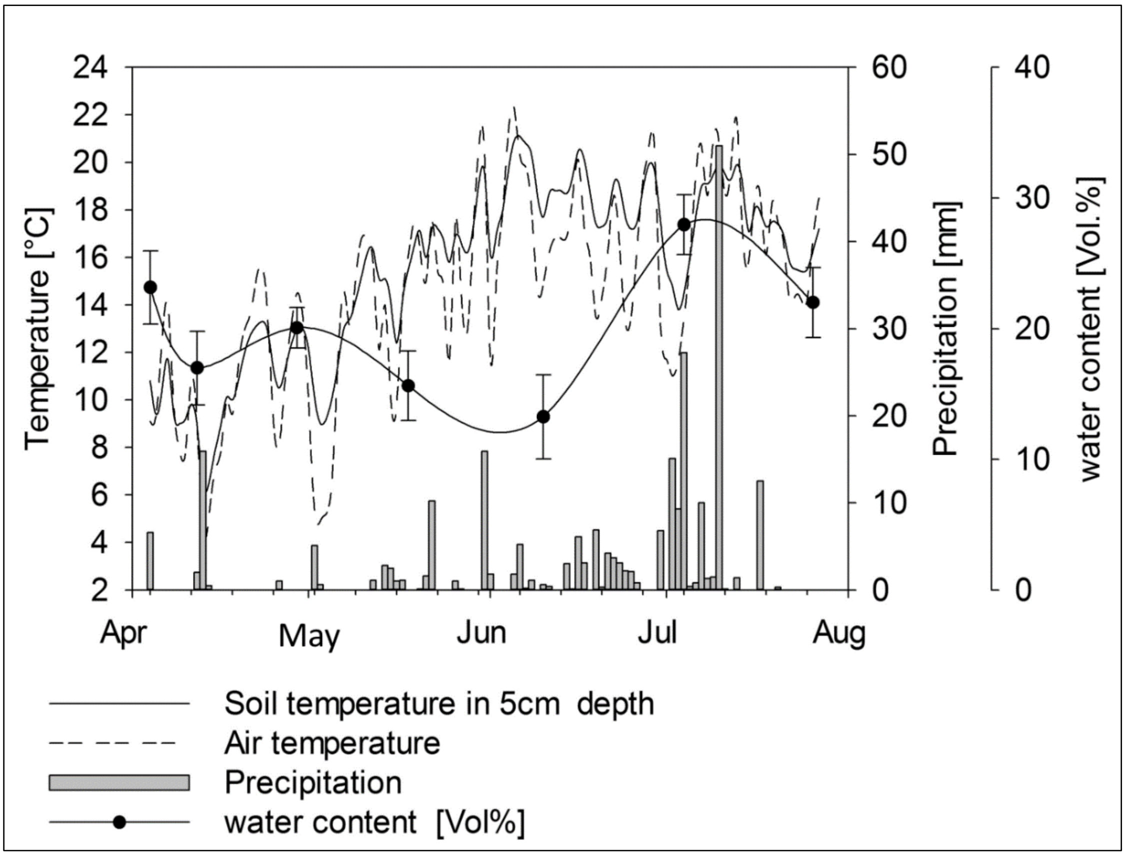

Figure 2-2 Weather conditions and soil water content during the experiment.

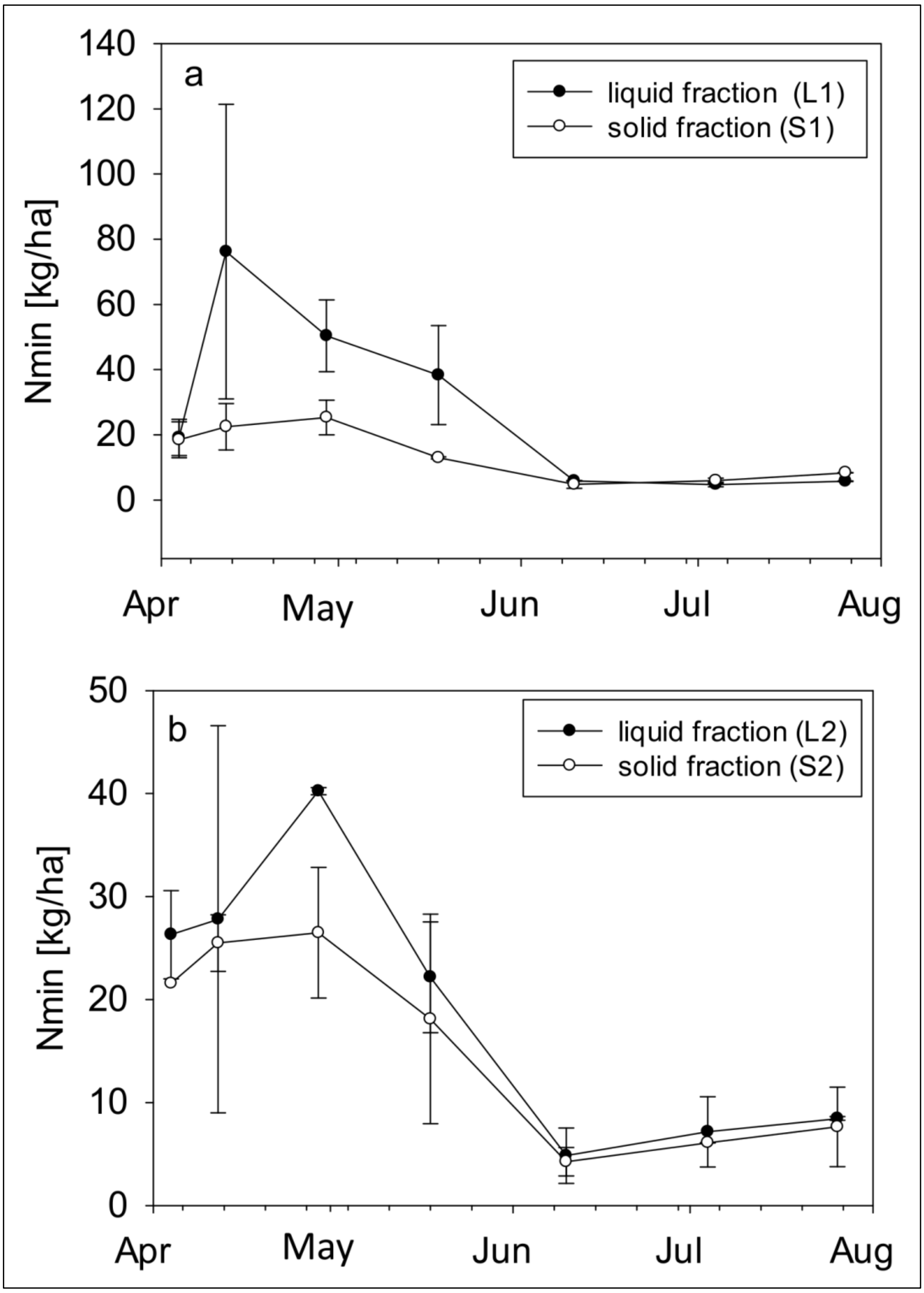

Figure 2-3 Temporal sequence of mineral nitrogen (Nmin) in soil (0-20cm depth) during the vegetation period in plots fertilized with a) L1 = liquid fraction of BGR from biogas plant 1 and S1 = solid fraction of BGR from biogas plant 1 and b) with L2 = liquid fraction of BGR from biogas plant 2 and S2 = solid fraction of BGR from biogas plant 2.

Table 2-2 Soil properties before fertilization and after harvesting. mean ± sd = mean value ± standard deviation, n.a. = sd is not possible, TC = total carbon, TN = total nitrogen, Nmin = mineral nitrogen;

	before fertilization				after harvest			
	pH	TC (%)	C/N	Nmin (kg/ha)	pH	TC (%)	C/N	Nmin (kg/ha)
	mean ± sd	mean ± sd	mean ± sd	mean ± sd	mean ± sd	mean ± sd	mean ± sd	mean ± sd
C1	5.9 ± 0.2	1.3 ± 0.1	9.2 ± 0.1	29.5 ± 10.8	5.6 ± 0.2	1.3 ± 0	9.1 ± 0.4	6.7 ± 0
L1	5.9 ± 0.3	1.1 ± 0.1	9.2 ± 0	19.2 ± 5.5	5.6 ± 0.4	1.1 ± 0.1	9.4 ± 0.1	5.8 ± 0.1
S1	5.8 ± 0.3	1.1 ± 0.1	9 ± 0.5	18.5 ± 5.5	5.6 ± 0.3	1.2 ± 0.1	9.3 ± 0.2	8.3 ± 0
C2	5.8 ± 0.3	1.2 ± 0.1	9.3 ± 0.1	28.4 ± 0	5.5 ± 0.3	1.2 ± 0.1	9.4 ± 0.3	8.3 ± 1.3
L2	5.5 ± 0	1.3 ± 0.1	9.2 ± 0.1	26.3 ± 4.3	5.8 ± 0	1.3 ± 0.2	9.2 ± 0	8.5 ± 0.2
S2	5.7 ± 0.2	1.1 ± 0.2	9.3 ± 0.2	21.6 ± n.a.	5.5 ± 0.2	1.1 ± 0.2	9.1 ± 0.3	7.6 ± 3.9
CS2	5.9 ± 0.4	1.2 ± 0.3	9.2 ± 0.3	25.5 ± 6.6	5.9 ± 0	1.2 ± 0.3	9.2 ± 0.4	9.3 ± 1.6
L3	6.2 ± 0	1.1 ± 0.2	9.3 ± 0.3	22.4 ± 2.6	5.6 ± 0.3	1.1 ± 0.2	9.2 ± 0.5	7.8 ± 3.8
L3+P	6.1 ± 0	1.1 ± 0.2	9.4 ± 0.3	28 ± 1.1	5.6 ± 0.3	1.1 ± 0.2	8.6 ± 0.5	7.8 ± 0
L3+WS	5.8 ± 0.4	1.2 ± 0.2	8.8 ± 0.4	26.6 ± 2.1	5.6 ± 0.3	1.2 ± 0.2	9.3 ± 0	9.8 ± 0.6
F	5.9 ± 0.4	1.3 ± 0.3	9.3 ± 0.3	24.6 ± 0.6	5.6 ± 0.4	1.1 ± 0.1	9.3 ± 0.3	6.9 ± 2.1
M	5.9 ± 0.2	1.3 ± 0	9.2 ± 0.1	30.6 ± 6.1	5.6 ± 0.1	1.3 ± 0	9.3 ± 0.2	8.8 ± 0.1
CN	6.2 ± 0.1	1.2 ± 0.2	9.4 ± 0.1	29.4 ± 5.5	5.9 ± 0.2	1.2 ± 0.2	9 ± 0.3	7.9 ± 0.1

2.4.3 Effects on yields and yield formation

All applied fertilizers induced significantly higher yields of barley grain and straw than the unfertilized control (Table 2-3). Nonetheless, plots which were fertilized with mineral fertilizer showed a trend towards 15% higher yields than BGR fertilized plots. Fertilizer treatments with high grain yields did not necessarily lead to a high straw yield (Table 2-3). Plots which were amended with liquid fraction of BGR (L1, L2, L3) tend to produce a higher grain yield than plots amended with solid fractions (S1, S2), composted (SC2) or complete BGR (C1, C2, F). Composted and solid fractions (CS2, S2) gave the same yield. Grain N content was higher in plots fertilized with S2. Addition of the nitrification inhibitor did not influence biomass and grain yield.

Plots with highest yields did not necessarily have the best performance in each category of the yield formation. Liquid fractions led to fewer plants per m² at the beginning of the vegetation period in comparison to solid and complete BGR. Later these variants compensated this disadvantage with more ears per plant and grains per ear compared to the other treatments. Differences in TGW among all treatments are not significant. Nevertheless, the TGW of the liquid variants tended to be higher compared with other treatments. Within biogas plant 3, L3 and L3+WS led to 50% more plants per m² than L3+P. During the vegetation period L3+P plants compensated the low number of plants with a high number of ears and a high number of grains per ear. In general, plots fertilized with M had the best performance in every yield formation category.

Table 2-3 Yield formation, grain and straw yields, N content in grain and aboveground biomass N uptake. Biogas residue abbreviations are explained in Table 1. mean ± sd = mean value ± standard deviation, LSD=Least Significant Difference t-Test. Means with the same letter are not significantly different.

BGR	plants/m²		ears/m²		ears/plant		grains/ear		TGW (g)		yield (dt/ha)		straw (g/m²)		N grain (%)		N uptake (kg/ha)	
	mean ± sd	LSD	mean ± sd	LSD	mean ± sd	LSD	mean ± sd	LSD	mean ± sd	LSD	mean ± sd	LSD	mean ± sd	LSD	mean ± sd	LSD	mean ± sd	LSD
C1	394 ± 70.7	ab	914 ± 144.2	a	2.3 ± 0.1	b	14.9 ± 0.3	ab	42.1 ± 1	abc	57.1 ± 6.5	ab	861.1 ± 82.1	ab	1.8 ± 0	ab	147.9 ± 22.5	ab
L1	298 ± 2.8	ab	812 ± 67.9	ab	2.7 ± 0.3	ab	16.3 ± 0.2	ab	46.1 ± 5.7	abc	58.5 ± 3.8	a	655.1 ± 59.4	ab	1.8 ± 0.3	ab	152.3 ± 40.2	ab
S1	350.7 ± 97.8	ab	897.3 ± 130.1	a	2.6 ± 0.5	ab	14.9 ± 1.6	ab	41.6 ± 2.9	bc	55.1 ± 4	ab	800.2 ± 41.7	ab	1.8 ± 0.2	ab	136.7 ± 46.3	ab
C2	384 ± 41.8	ab	969.3 ± 40.5	a	2.5 ± 0.2	ab	15.8 ± 0.6	ab	39.3 ± 1.4	bc	60.3 ± 1.9	a	877.5 ± 102.4	ab	1.6 ± 0.2	ab	120.6 ± 56.2	ab
L2	354 ± 42.4	ab	868 ± 67.9	ab	2.5 ± 0.1	b	15.7 ± 1.9	ab	45.2 ± 2.6	ab	62.3 ± 1.3	a	795.1 ± 89.2	ab	1.7 ± 0.2	b	128 ± 66.4	ab
S2	350.7 ± 70.5	ab	952 ± 188.6	a	2.8 ± 0.5	ab	14.6 ± 2.1	ab	41 ± 6.6	bc	55.9 ± 6.5	ab	855.7 ± 200.2	ab	1.9 ± 0.1	a	158.6 ± 11.9	ab
SC2	362.7 ± 22	ab	856 ± 8	ab	2.4 ± 0.2	b	15.4 ± 0.7	ab	42.2 ± 3.3	abc	55.5 ± 5	ab	782.3 ± 67.2	b	1.6 ± 0.2	b	122.7 ± 19.7	ab
L3	374.7 ± 10.1	ab	885.3 ± 181.5	a	2.4 ± 0.5	b	14.7 ± 0.8	b	46.6 ± 1.6	ab	60.1 ± 10.4	a	871.5 ± 245.1	b	1.6 ± 0.3	ab	140.9 ± 23.1	ab
L3+P	258.7 ± 96.1	b	876 ± 43.3	ab	3.8 ± 1.6	a	17.1 ± 0.2	a	41.4 ± 3.3	abc	61.7 ± 2	a	711.9 ± 170.7	ab	1.6 ± 0.1	ab	134.1 ± 6.8	ab
L3+WS	336 ± 170.3	ab	785.3 ± 119.5	ab	2.9 ± 1.6	ab	15.6 ± 2	ab	40.2 ± 6.6	abc	48.8 ± 9.9	ab	646.6 ± 260.9	ab	1.7 ± 0.2	ab	100.5 ± 11.7	ab
F	420 ± 32	a	885.3 ± 224	ab	2.1 ± 0.7	b	15.2 ± 0.4	ab	41.1 ± 2.9	Abc	56 ± 19	ab	723.9 ± 208.2	a	1.7 ± 0.4	ab	160.7 ± 80	ab
M	480 ± 118.8	a	950 ± 291.3	a	2 ± 0.1	b	16 ± 1.2	ab	46.6 ± 2	Ab	71 ± 20.7	a	957.4 ± 213.2	ab	1.8 ± 0.1	ab	173.2 ± 77.6	ab
CN	336 ± 45.1	ab	644 ± 74.9	bc	1.9 ± 0.2	b	14.2 ± 1.7	b	40.8 ± 3.2	Bc	37 ± 2.6	b	483 ± 38.3	b	1.5 ± 0.1	b	77.6 ± 4.2	b

2.5 Discussion

2.5.1 Biogas residue properties

The BGRs tested in this study were produced in different biogas plants operating with different digestion techniques and substrates. The composition of the different BGRs can vary greatly depending on the feedstock used for co-digestion and the process characteristics (Alburquerque et al., 2012a; Zirkler et al., 2014). The result of different input or different treatment and separation technique can also be different N concentrations among the BGRs (Jørgensen and Jensen, 2009; Moller et al., 2007). The fermentation process increases the availability of N (NH_4-N) due to the breakdown of organically bound N during the anaerobic process (Jørgensen and Jensen, 2009; Möller et al., 2008; Weiland, 2010). Measured NH_4-N proportion of total N in liquid fraction was between ca. 14% and 59% and in solid fraction between 2% and 3%. This is because after the separation process NH_4-N almost entirely migrates into the liquid fraction (Jørgensen and Jensen, 2009; Møller et al., 2002; Nkoa, 2014). Like compost, the solid fraction is a poor source of N due to the low content of mineral N and low mineralization, which is in agreement with our results (Bath and Ramert, 1999; Svensson et al., 2004). Thus the liquid fraction of BGRs from bio-wastes can replace substantial amounts of mineral N and can substitute mineral fertilizer. However, increased NH_4-N concentration in digested slurries compared to undigested slurries does not necessarily guarantee improved uptake efficiency of slurry N or increased savings in fertilizer N (Möller and Müller, 2012). Nitrification of the ammonium N in the BGR can cause significantly increased nitrate leaching (Haraldsen et al., 2011). Furthermore, alkaline pH and high NH_4-N

concentration of BGRs may result in N losses due to NH_3 volatilization (Loria et al., 2007; Möller and Stinner, 2009; Sänger et al., 2010). However, a higher ammonia volatilization potential compared with undigested slurry does not necessarily result in higher emissions since the lower solid content and reduced viscosity lead to a better infiltration characteristics (Herrmann, 2013). Results in the literature on the effect of anaerobic digestion on ammonia volatilization after field application therefore are inconsistent (Herrmann et al., 2013).

2.5.2 Effects on soil properties

Biogas residues have an alkaline pH. Al-Juhaimi et al. (2014) and De la Fuente et al. (2013) reported that the pH of the soil decreased after application of the liquid fraction from BGRs and increased after solid fraction application. In our study, biogas residues did not affect soil pH despite values above pH 8 in BGRs. This could be attributed to the high buffer capacity of the clay minerals in the silty-loam soil.

Contradicting information is given in literature about C and N change in soil after BGR application. Some authors showed that land application of BGRs had short-term benefits in terms of improving SOM stock due to the addition of the organic matter with this material (Chen et al., 2012; De la Fuente et al., 2013; Nicoletto et al., 2014). In contrast, but similar to our study, Bachmann et al. (2014) reported that, even after 3 years, there was no change of organic C content in soil. The effect on soil N and C turnover and in contributions to soil C storage of digested materials is small compared to the amount of N and C already residing in the soil and therefore difficult to quantify over shorter time spans (Möller, 2015). The mean TC of our soil was 1.2%. With BGRs only 0.02-0.18% organic matter was additionally applied in the upper 10 cm.

2.5.3 Effects on yields and yield formation

Contrasting results for the effect of BGRs on yield are reported (Möller and Müller, 2012). Those research results can be grouped into three categories of performance: (a) performances similar to unfertilized controls (Svensson et al., 2004), (b) performances similar or higher than undigested feedstock (Loria et al., 2007; Möller et al., 2008) and (c) performances equal or better than mineral fertilizers (Abubaker et al., 2012). None of the BGRs used in the present study showed a negative effect on barley yield. This is in line with results reported by others (Abubaker et al., 2012; Odlare et al., 2014). The yields measured in our experiment correspond to average yields (59 dt/ha) for the "Landkreis Leipzig" study region in 2011 (StLa, 2012a). Fertilization with BGRs was based on TKN which is not completely directly available to plants. Nevertheless, no significant differences in the grain yields of BGR- and M-fertilized plots were found, hence BGRs have a high fertilizing potential.

One important plant nutrient is mineral N, which is generally the limiting factor for crop growth (Odlare et al., 2008). In the literature, high fertilizing potential is associated with NH_4-N content which is immediately plant available after application (Alburquerque et al., 2012b). The mineral fertilizer N is directly plant available too. A large proportion of N in liquid variants was directly plant available as well. In accordance with that, soil N_{min} in L2 variants was high at the beginning of the vegetation period and decreased until the end of the vegetation period due to plant uptake. Nitrogen in solid variants has to be mineralized before plants can take it up. Soil N_{min} in plots fertilized with S2 was low throughout the observation period. The plant availability of N in these plots started only during the grain filling phase. As a result, grain N of solid variants was equal or significantly higher compared to liquid variants.

In some cases factors like soil water content and temperature may have a greater effect on the uptake of nutrients than the amount of nutrients applied as fertilizer (Petr et al., 1988). Under dry weather conditions barley yield did not increase, even if soil N resources were high (Ma et al., 1992). Since water supply did not differ significantly among the plots, fertilization is the single influencing factor on yield formation in this case.

The grain yield of cereals is a product of three basic compounds: ears per m², grains per ear and TGW (Petr et al., 1988). Ears per m² is a product of plants per m² and ears per plant. The number of plants per m² depends on sowing and initial growth conditions. The number of ears per plant depends on growth conditions during tillering. Spring barley was sown at a constant number of 450 seeds per m² over the whole field and produced a number of plants per m² which is in line with the literature (Koutna et al., 2003). Nevertheless, the liquid fractions L1 and L2 led to fewer plants per m² than corresponding solid and complete variants. The reason for this could be a higher NH_4-N concentration, which can reflect a potential phytotoxicity of organic products during germination (Alburquerque et al., 2012b; Schittenhelm and Menge-Hartmann, 2006; Teglia et al., 2011). L3+P gave rise to fewer plants per m² than L3 and L3+WS. Nitrification was inhibited, NH_4-N concentration remained high, and induced higher damage through ammonia toxicity compared to variants without Piadin®. Nevertheless plots fertilized with liquid fraction compensated the initial disadvantages over the vegetation period and induced higher grain yields.

One result of our study was that fertilization with liquid variants in comparison to other variants tended to result in a higher number of grains per ear, which is the most common limiting yield component for wheat and barley, and is dependent on N supply during tillering (Baethgen et al., 1995). Average value for grains per ear in Germany is 17.9 grains (Diepenbrock et al., 1999). Other authors have stated a higher number of grains per ear of between 18.1 and 28 grains (Koutna et al., 2003; Ma et al., 1992; Schittenhelm and Menge-Hartmann, 2006). In our study the higher grain yield was attributable to a higher number of ears per plant and grains per ear, which is in line with Schittenhelm and Menge-Hartmann (2006).

The thousand grain weight on plots fertilized with liquid fractions was higher than on corresponding variants as well. Grains achieved a TGW which is average for spring barley in Germany (45 g) (Diepenbrock et al., 1999). Since TGW depends on temperature, moisture and mineral nutrient availability at grain filling, the N supply of liquid fractions was adequate (Petr et al., 1988). In addition liquid variants achieved a high TGW without a late application of N which affects the weight of grains (Petr et al., 1988).

2.5.4 Risks associated with land application

The relatively high mineral N content, up to 80% of which was present in the form of NH_4-N, indicates a high potential for N losses during BGR handling and application (Möller and Müller, 2012). High pH and NH_4-N concentrations are conditions that favour NH_3 emissions (Nkoa, 2014). Inappropriate storage or application of BGRs represent a risk of air and water pollution due to possible gaseous nitrogen emissions and/or nutrient leaching and runoff into surface and ground waters (Nkoa, 2014). Land application of BGRs as fertilizer is not risk-free, since it may result in soil contamination including physical contaminants such as plastics, glasses and stones as well as chemical contaminants such as phytotoxic compounds, pathogens and heavy metals (Nkoa, 2014).

2.6 Conclusions

The initial short-term fertilization effect of soil treated with BGRs from bio-wastes is similar to agricultural BGRs and mineral fertilizer. Thus BGRs can replace a considerable amount of mineral fertilizer. The nitrification inhibitor did not impact the grain yield. Biogas residues affected the yield formation. Liquid fraction alone or in combination with nitrification inhibitor induced fewer plants per m² than corresponding solid and complete variants or variants without the inhibitor. This might be explained by the high potential phytotoxicity of NH_3. Nevertheless, barley compensated the initial disadvantages over the vegetation period.

To date liquid fractions from biogas plant 1 and 2 were disposed of through the sewage system after treatment. Our study showed that direct application of the liquid fraction is the most suitable strategy to achieve highest yields. It provides agricultural benefits and helps to close nutrient cycles. However, it can cause damage due to NH_3 phytotoxicity. Solid fractions could be composted before being used as a soil conditioner. This would reduce transport costs due to the resulting lower volume and facilitate its addition to soil. Furthermore, composting reduces water content and odor emissions due to the reduction of volatile compounds and potential phytotoxicity, and also contributes to the elimination of pathogens. Nevertheless during a land application potential physical, chemical and biological soil contaminations as well as atmospheric and water pollutions should be considered.

2.7 Acknowledgements

We would like to thank Christian Krebs from DBFZ for enabling the participation in the field experiment; Dr. Jürgen Döring from MLU Halle for providing the weather data; Petra Peterson, Sylvia Schmögner, Martina Stange and Aleksey Prays for their support in the field and in the laboratory.

3 Biogas residue parameterization for soil organic matter modeling[2]

3.1 Abstract

A variety of biogas residues (BGRs) has been used as organic fertilizer in agriculture. The use of these residues affects the storage of soil organic matter (SOM). In most cases, SOM changes can only be determined in long-term observations. Therefore, predictive modeling can be an efficient alternative, provided that the parameters required by the model are known for the considered BGRs. This study was conducted as a first approach to estimating the organic matter (OM) turnover parameters of BGRs for process modeling. We used carbon mineralization data from six BGRs from an incubation experiment, representing a range of substrate inputs, to calculate a turnover coefficient k controlling the velocity of fresh organic matter (FOM) decay and a synthesis coefficient η describing the SOM creation from FOM. An SOM turnover model was applied in inverse mode to identify both parameters. In a second step, we related the parameters k and η to chemical properties of the corresponding BGRs using a linear regression model and applied them to a long-term scenario simulation. According to the results of the incubation experiment, the k values ranged between 0.28 and 0.58 d^{-1} depending on the chemical composition of the FOM. The estimated η values ranged between 0.8 and 0.89. The best linear relationship of k was found to occur with pH (R^2 = 0.863). Parameter η is related to the C_t/N_{org} ratio (R^2 = 0.696). Long-term scenario simulations emphasized the necessity of specific k and η values related to the chemical properties for each BGR. However, further research is needed to validate and improve these preliminary results.

3.2 Introduction

In recent decades, there has been increasing interest in Europe about the implementation of anaerobic digestion plants in farming contexts because of EU policies and directives aimed at reducing greenhouse gases (GHG) and the promotion of renewable energy production (United Nations, 2015). The expansion of biogas production in Germany began with the adoption of the Renewable Energy Sources Act in 2000 and its amendment in 2004. (DBFZ, 2012). Germany's current policy strives to increase the contribution of renewable energy resources as a substitute for fossil energy resources in order to decrease CO_2 emissions (BMUB, 2016). Thus, biogas production will be a key technology in the future (Thrän et al., 2015a).

[2] Prays, N., Dominik, P., Sänger, A., Franko, U.: Biogas residue parameterization for soil organic matter modeling. PlosOne. (submitted)

By 2015, approximately 8,000 biogas plants (DBFZ, 2015) were producing biogas from different biomass sources. During the production of biogas, an organic byproduct, known as biogas residue, is produced in large quantities. Biogas residues contain high levels of plant-available nutrients (N, P, K) and a considerable amount of organic carbon, which explains their widespread usage in agriculture for closing nutrient cycles (Möller, 2015; Tambone et al., 2010). Several studies have demonstrated the benefits of the agricultural use of BGRs for soil (Nkoa, 2014; Prays and Kaupenjohann, 2016). Anaerobic digestion transforms approximately 20–95% of the carbon in the feedstock into gaseous carbon compounds. Consequently, the amount of organic carbon introduced into the soil is reduced in comparison with direct soil incorporation of undigested feedstocks (Reinhold et al., 1991). On a regional scale, biogas production can also lead to an increase in organic matter inputs into soil, in the case of the introduction of new crops, as well as the complete redesign of crop rotations, crop acreage, and cropping systems (Möller *et al*. 2011). Furthermore, anaerobic digestion increases the availability of N (NH_4-N) due to the breakdown of organically bound N during the anaerobic process (Möller et al., 2008; Weiland, 2010). Odlare et al. (2008) found that, relative to other treatments (pig manure, cattle manure, compost, inorganic fertilizer), soils treated with BGRs from household wastes displayed the highest microbial biomass, nitrogen mineralization rate and potential ammonia oxidation. Thus, BGRs may have an effect on the storage of soil organic matter (SOM) and the nitrogen balance in the soil, both of which are important for sustainable soil use and the maintenance of soil functions. Since the impact of BGRs on SOM changes can only be determined in long-term applications, predictive modeling can be a helpful tool to estimate changes in SOM. Several process-based agroecosystem models, such as DNDC and RothC, have been widely utilized for quantifying soil carbon sequestration and are capable of determining the effects of organic fertilizers on soil carbon dynamics (Li et al., 2016); however, until now, no model considers BGRs. It is crucial to define the carbon turnover parameters of BGR degradability and their efficiency for creating new SOM when modeling SOM dynamics (Post et al., 2008; Smith et al., 1997).

Some of the required parameters, such as dry matter and carbon concentration, are fast and easy to obtain for modeling. However, to derive the carbon turnover parameters, incubation experiments over a period of several weeks are usually required. (Post et al., 2008). Incubation experiments are time consuming, and in practical applications, it is not possible to perform incubation experiments regularly. Furthermore, since the substrate mix applied for biogas production is very heterogeneous in terms of the composition of plant nutrients and organic matter, the chemical composition of biogas residues is also very diverse (Weiland, 2010; Zirkler et al., 2014). Therefore, a specific parameter set may be required for each BGR.

The objective of this study was to derive BGR carbon turnover parameters for process modeling. Therefore, we first used the carbon mineralization rates from an incubation experiment, which is described in detail in Sänger et al. (2014), to calculate the parameters for BGRs with different composition. In a second step, we hypothesized that it is possible to estimate these parameters using some easily measurable BGR properties and to develop a simpler approach as an alternative to the existing time- and labor-consuming incubation experiments.

3.3 Material and methods

3.3.1 Biogas residues

We chose only BGRs that had been sampled from an additional BGR store and not directly from the fermenter to ensure a comparable (high) degree of degradation of all BGRs for our parameterization. The BGRs used in this study represent a wide range of substrate inputs (Table 3-1) and were taken from two-stage biogas plants. These indices refer to the percentage of maize in the substrate input. Other compounds include grass silage, rye silage, cereals, pig slurry, cattle slurry and farmyard manure. All of the BGRs were derived from wet digestion under mesophilic conditions. The methods used for BGR analysis are described in detail in Sänger et al. (2014). The chemical properties of used BGRs are shown in Table 3-2.

Table 3-1 Composition of biogas residues (in% mass) from Sänger et al. (2014)

BGR	maize silage	grass silage	rye silage (whole plant)	shredded grain	pig slurry	cattle slurry	farmyard manure
	%	%	%	%	%	%	%
D_{17}	17	-	-	-	19	64	-
D_{24}	24	31	8	-	-	37	-
D_{33}	33	-	25	-	20	-	22
D_{52}	52	8	-	2	35	-	3
D_{61}	61	-	-	5	34	-	-
D_{100}	100	-	-	-	-	-	-

Table 3-2 Chemical properties of BGRs from Sänger et al. (2014). DM = dry matter, DM_{org}= organic dry matter, C_t = total carbon, N_t = total Kjeldahl nitrogen, N_{org} = organic nitrogen calculated as N_{org} = N_t-NH_4-N

BGR %	DM %	DM_{org} %	pH	C_t %DM	NH_4-N %DM	N_{org} %DM	N_t %DM	C_t/N_t	C_t/N_{org}
D17	5.5	28.1	8	38.4	2.9	3.4	6.3	6.1	11.3
D24	9.2	27.9	7.8	39.2	3.5	3.0	6.5	6.0	13.1
D33	9.6	33.0	8	41.3	2	3.0	5.0	8.3	13.8
D52	7.2	29.2	7.7	40.7	4.3	3.4	7.7	5.3	12.0
D61	8.1	33.48	7.9	42	5.5	2.7	8.2	5.1	15.6
D100	6.8	34.08	7.7	43.2	2.9	4.3	7.2	6.0	10.0

3.3.2 Model and carbon turnover parameter description

We used the carbon turnover sub-model that is integrated in the agroecosystem model CANDY (Carbon And Nitrogen DYnamics) and the SOM model CCB (Candy Carbon Balance) (Franko et al., 2011; Franko et al., 1995). Here, SOM is divided into an active pool where mineralization occurs, a stabilized pool that represents the passive but decomposable part of the SOM, and a long-term stabilized pool that is considered inert. Matter exchange between the active SOM and stabilized SOM is assumed by the model. A more detailed description of the interaction between these model pools was given by Franko et al. (2011). In our model, the carbon reproduction (C_{rep}) flux, representing that part of FOM that is incorporated within the active SOM pool, is the driver of SOM accumulation. The FOM turnover is described using first-order kinetics with the following parameters: 1) a turnover coefficient k describing the resistivity against microbial breakdown of the material and 2) a synthesis coefficient η specifying the carbon transfer from decomposed FOM to new active SOM. Thus, higher k values indicate a higher velocity of FOM decay. Higher η values indicate higher contributions to SOM.

Furthermore, the biogas residues need to be characterized based on the following properties: organic carbon, organic and inorganic nitrogen, total dry matter, and all values as concentration of % weight by weight (% w/w), for quantitative modeling of matter fluxes.

3.3.3 Incubation

The incubation experiment is described in detail in Sänger et al. (2014) and is briefly summarized here. Two different soils were used for the incubation experiment: a silty soil (5% sand, 75% silt, 20% clay, pH value = 6.5) and a sandy soil (46% sand, 39% silt, 15% clay, pH value = 7.5). The dried and sieved soils were amended with each of the BGRs at a rate of 0.5 g N (kg soil)$^{-1}$, adjusted to 60% of water holding capacity, and incubated at 25 °C for 6 weeks. Emissions of CO_2 were measured daily on days 1-20, afterwards on days 22, 24, 27, 30, 34, 36, 41. The mineralization rates were calculated as part of the added carbon. All of the BGRs and control treatments were incubated in four replications. The observed mineralization rates were cumulated in the following modeling step. The control treatments were measured with another frequency; thus, the missing values were interpolated to a polynomial. Differences between each sample and each control treatment were calculated for each observation time step. These differences were used to calculate a mean cumulative mineralization rate value and the total variance of each BGR and time step. For our analysis, we selected only the first 20 days of the experiment for daily measurements.

3.3.4 Inverse modeling

We used inverse modeling to fit the parameters k and η to the observed CO_2 mineralization for each BGR–soil combination, minimizing the root mean square error (RMSE) between the observed and modeled values. The fitting procedure is based on a numerical simplex method for minimization of a non-linear function (Nelder and Mead, 1965). The parameter estimation was completed using uncertainty calculations based on the Fisher Information Matrix approach (DeGroot, 1986). This approach is based on the sensitivity of the model output and the variance of the measured CO_2 emissions at each observation point. For further analysis, we took the mean values of k and η for each BGR because, according to the model approach, k and η depend only on FOM type. Site conditions, such as soil type, are considered in the model using the Biological Active Time (BAT), which was calculated for each soil separately (Franko and Oelschlägel, 1995).

3.3.5 Prediction of model parameters based on chemical properties

In the next step, the estimated carbon turnover parameters from inverse modeling were related to the chemical properties of BGRs. This was done to predict the model parameters without using time-consuming incubation experiments. Therefore, a linear regression ($f(x) = mx+n$) was performed and the coefficient of determination R^2 was calculated; the slope m and intercept n were calculated using R version 3.3.1 (The R Foundation for Statistical Computing, 2016). For the x variable, the chemical properties DM, DM_{org}, C_t, NH_4-N, N_{org}, N_t, C_t/N_t, C_t/N_{org} and pH were used. The approach with the best R^2 was selected. In the following text, we use the symbols k and η to represent the parameters that were estimated with inverse modeling and k* and η* to represent when their predictions were based on the chemical properties of the BGRs.

3.3.6 Scenario modeling

A simple bioenergy production scenario of continuous maize (yield = 500 dt ha^{-1}) and annual BGR application as organic fertilizer (170 kg N ha^{-1}) was modeled with the CCB model that was already validated (Franko et al., 2011). The influence on carbon storage for each BGR was evaluated using the mean, minimal and maximal k and η as well as k* and η*. The influence on carbon storage was evaluated using k and η as well as k* and η* for each BGR. The parameter errors were included in the simulation; thus, for each parameter the mean, minimal and maximal values were calculated. For the scenario simulation, we assumed typical site conditions of the Chernozem region in Central Germany, with a mean annual temperature of 8.5 °C, mean annual precipitation of 480 mm and Haplic Chernozem soil (21% clay, 68% silt, 11% sand). The initial value for soil organic carbon was set to 2%. Soil organic carbon change was calculated for 100 years of unchanged soil management. The mean C_{org} concentrations as well as the standard deviations were calculated. The effects of different treatments were analyzed with a one-way ANOVA. A least significant difference t-test (LSD) was used to compare the mean values and to assess the significance of the differences between the mean values. The effects were considered significant for $p < 0.05$. All statistical analyses were performed using R version 3.3.1 (The R Foundation for Statistical Computing, 2016).

3.4 Results

3.4.1 Estimation of parameters from incubation results

Using inverse modeling to identify the parameters from the observed CO_2 emissions provided good results (Figure 3-1). The adaptation error was for all treatments lower than 1% of emitted carbon (Table 3-3). Our estimated k values were between 0.28 and 0.58 d^{-1}, and the η values of BGRs were between 0.8 and 0.89 (Table 3-3).

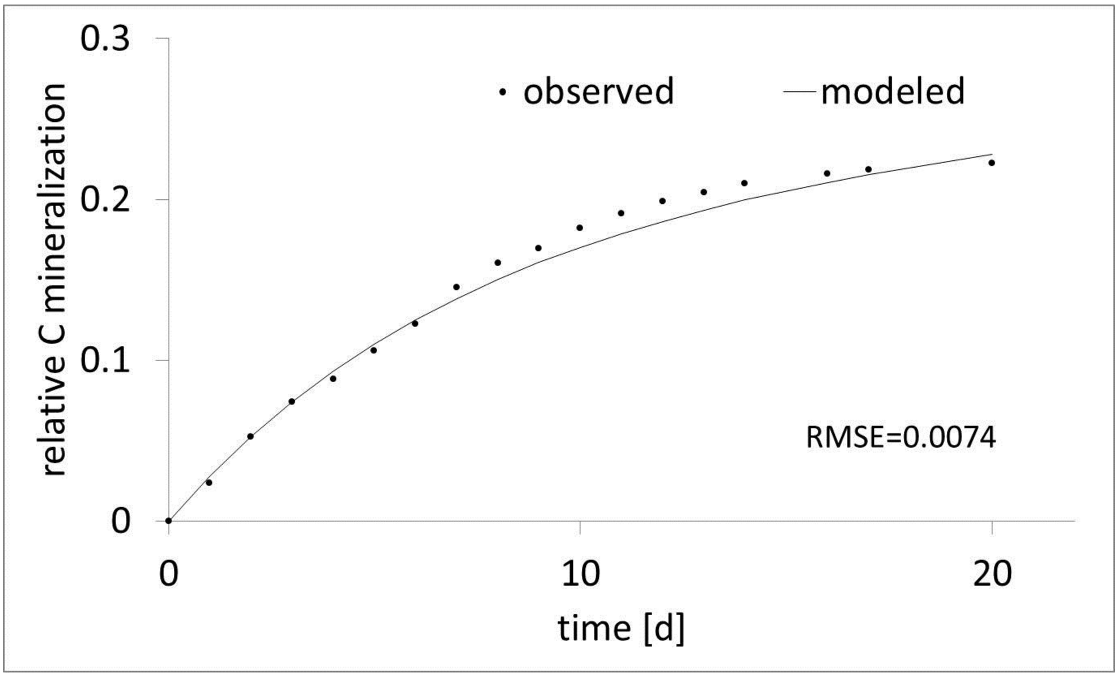

Figure 3-1 Measured and modeled CO_2 mineralization of BGR with a biggest RMSE (D_{100})

Table 3-3 Fitted parameterized values of six different BGRs. RMSE = root mean square error between the modeled and observed values of C mineralization, sd standard deviation of the fitted parameters

BGR	k		η		RMSE
	[d^{-1}]	Sd	[-]	sd	[part of emitted C]
D_{17}	0.362	0.060	0.844	0.009	0.007
D_{24}	0.420	0.075	0.871	0.018	0.008
D_{33}	0.279	0.036	0.828	0.018	0.006
D_{52}	0.506	0.081	0.851	0.016	0.008
D_{61}	0.391	0.034	0.802	0.015	0.009
D_{100}	0.575	0.076	0.890	0.009	0.009

3.4.2 Prediction of model parameters based on chemical BGR properties

The results of the regression analysis are shown in Table 3-4. Parameter k in d^{-1} was strongly related to the pH values: k = = 5.996-0.710*pH (R^2 = 0.863) (Figure 3-4a). Synthesis coefficient η was best described using C_t/N_{org} ratio: η = 1.016-0.013* C_t/N_{org} (R^2 = 0.696) (Figure 3-4b). Multiple regressions were not considered because the correlations between C/N_{org} and the pH value (r = 0.43), and N_{org} and the pH value (r = 0.55) were too high. The correlation between the η and the C/N_{org} ratio, k and pH value were found to be significant ($p<0.05$).

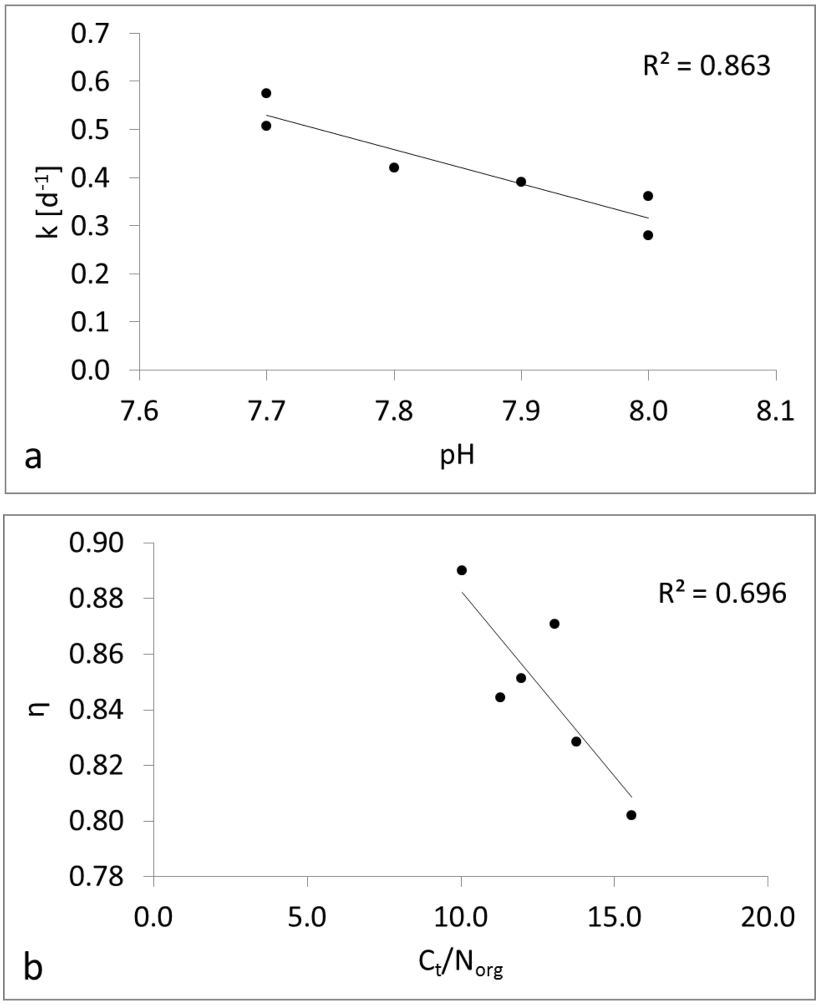

Figure 3-2 a) relationship and R^2 of k and pH ($p<0.05$), b) relationship and R^2 of η and C_t/N_{org} ($p<0.05$)

Table 3-4 R^2 of the linear relationship between the model parameters k or η and selected the BGR property

BGR property	R^2 (k)	R^2 (η)
DM	0.245	0.046
DM_{org}	0.035	0.139
C_t	0.180	0.001
NH_4-N	0.064	0.189
N_{org}	0.547	0.624
N_t	0.402	0.007
C_t/N_t	0.495	0.028
C_t/N_{org}	0.371	0.696
pH	0.863	0.411

3.4.3 Modeled vs. predicted parameters – a comparison of results

The simulation of a 100-year scenario with parameters k and η resulted in a range of 0.18% w/w between the lowest and highest C_{org} concentration (Figure 3-3). The range between the parameters predicted with chemical BGR properties was the same.

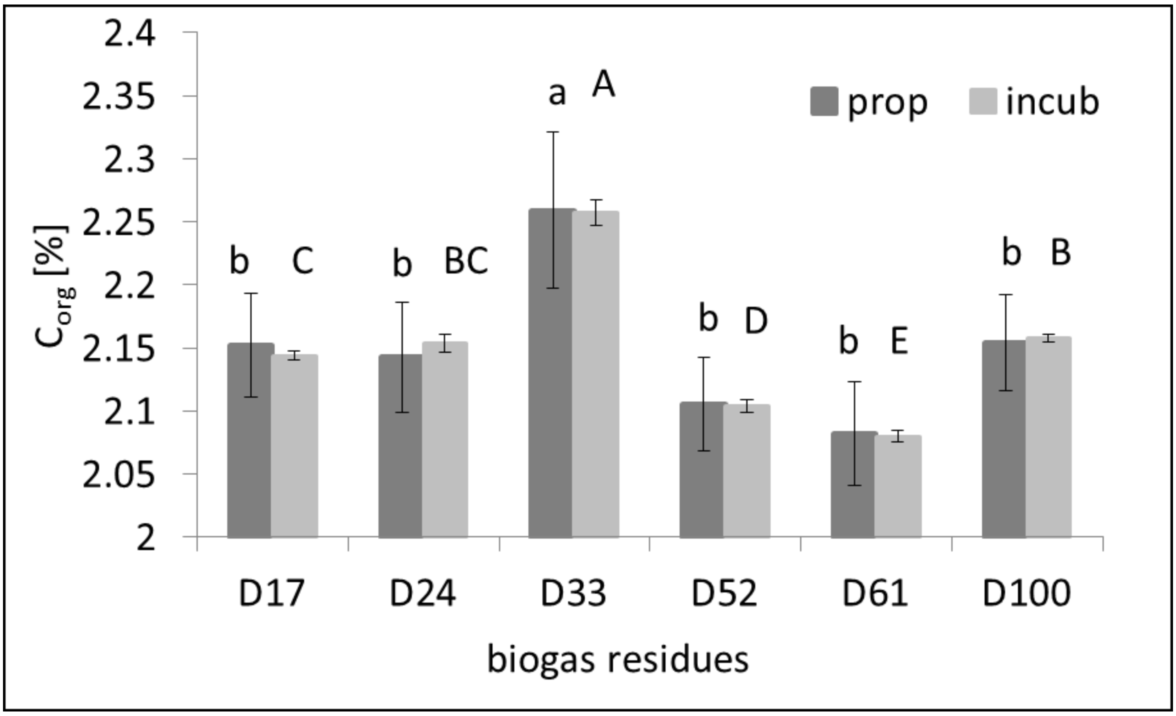

Figure 3-3 C_{org} concentrations after a 100-year scenario simulation with continuous maize (yield 500 dt ha $^{-1}$) and BGR application (170 kg N ha $^{-1}$). incub = parameters estimated with the results of the incubation experiment (k, η), which were used for the modeling; prop = parameters predicted with chemical properties of the BGRs (k*, η*). D_{17}-D_{100} are different biogas residues. Letters (small = prop, capitals = incub) indicate the results of the Least Significant Difference t-Test. Means with the same letter are not significantly different

3.5 Discussion

3.5.1 Quality of parameters based on incubation results

The decomposition of FOM was determined by its turnover coefficient k. The higher the k value, the higher the velocity of the FOM turnover process. Our estimated k values were slightly higher than the k values used for other organic materials in the CANDY process model. Regrettably, there are no references in the literature to compare with our results. An explanation could be that during anaerobic digestion, the complex organic materials, such as carbohydrates, lipids and fats, cracked into monomers and then into fatty acids followed by degradation into biogas (Weiland, 2010), thus providing more easily decomposable compounds. However, according to the model, the synthesis coefficient η has a stronger influence on the carbon reproduction flux than on the turnover coefficient k (Franko et al., 2011). Thus, we focus on η in our further discussion.

The FOM decomposition results in the creation of SOM. This part of the carbon flux was mainly determined by the synthesis coefficient η. The bigger the η value, the more FOM carbon was integrated into SOM. In Sänger et al. (2014), the amount of emitted CO_2 relative to the supplied carbon (up to 22.2%) was slightly lower compared to other authors (Sänger et al., 2010; Sänger et al., 2011; Terhoeven-Urselmans et al., 2009). Thus, the contribution to SOM storage was high, with correspondingly high η values. In the CANDY model, the following parameter values are integrated: $\eta = 0.6$ for cattle manure and $\eta = 0.65$ for cattle slurry (Franko, 1989). Nielsen et al. (2015) classified the biodegradability of BGRs between cattle slurry and cattle manure. In contrast, Chen et al. (2012) showed in a 21 days incubation experiment that in BGR-treated soils, only 6.4% of the initial carbon input was mineralized compared to 30% of the initial mineralized carbon in maize straw-treated soils. This means that BGRs had a higher η than maize straw (η of maize straw = 0.62 in CANDY), which corresponds to our findings with an η range between 0.8 and 0.89. Our findings are also confirmed by the observations of De la Fuente et al. (2013), who reported that approximately 60% more carbon was mineralized in cattle slurry-treated soil than in BGR-treated soil after 56 days of incubation.

In the experiment used for this study, only the total carbon of the BGRs was measured, so no information about inorganic carbon concentration was available. Carbonates in BGR can represent up to 7.6% of dry matter depending on the substrate mix (Fouda et al., 2013). The incorporation of BGRs with a pH value of 7.5 - 7.8 into a relatively acid soil with a pH value of 5.5 - 6.5 can very likely be accompanied by the reaction of carbonate with protons to form CO_2 and H_2O (Chen et al., 2011). This CO_2 can be mistakenly interpreted as organic carbon degradation, and consequently, carbon mineralization rates could be over-estimated. This could be an error source in this study, since higher mineralization rates would mean smaller η values.

3.5.2 Quality of parameter estimation based on chemical BGR properties

Soil organic matter results from FOM turnover and is a result of microbial productivity. It is composed of plant residues, microbial compounds and molecules resulting from biological degradation (Carr et al., 2013; De la Rosa et al., 2012; González-Pérez et al., 2012). When applied to soil, microorganisms start to utilize the BGRs, which are rich in compounds of microbial biomass (Coban et al., 2015). The efficiency of the microbial organic matter turnover depends on the quality of the FOM, the microbial community composition and environmental conditions (Lettau and Kuzyakov, 1999).

The pH value can be an important factor of environmental conditions for the microorganisms. Biogas production is performed using complex microbial communities that need different pH values for optimal performance during the different organic matter degradation stages (Weiland, 2010). In soil, the pH value influences a number of factors affecting microbial physiological status, microbial activity, like solubility, and the ionization of inorganic and organic solution constituents; these, in turn, affect soil enzyme activity (Paul, 2007; Royer-Tardif et al., 2010). The pH influence on several processes during anaerobic digestion, as well as on the soil, are not a complete explanation, but it is still reasonable to relate the k parameter to this generally available property.

It is well known that the C/N ratio is important for microbial decomposition. Bacterial biomass generally has a much lower C/N ratio (3.5:1 to 7:1) than fungi (10:1 to 15:1), plant residues or soil (Józefowska et al., 2017; Paul, 2007). The stronger the processing and decomposition of the fermenter feedstock in the biogas plant, the lower the C/N ratio, the higher the NH_4^+ concentration and pH value, and consequently, the lower the concentration of microbial biomass in the remaining BGR (Wentzel and Joergensen, 2016). The BGRs used in this study have C/N ratios between 10 and 15.6. In the model, the C/N ratio of the decomposable SOM is fixed at 8.5, which is similar to the C/N of microorganisms (Franko et al., 2011). The average fungal carbon to bacterial carbon ratio for BGRs is 0.29 (Wentzel and Joergensen, 2016). This means that microbial carbon consists of 23% fungal carbon and 77% bacterial carbon, neglecting the possible presence of Archaea (Wentzel and Joergensen, 2016). Thus, when the C/N ratio is high, fungal development is favored over bacterial development (Henriksen and Breland, 1999; Hodge et al., 2000). Therefore, increasing C/N ratios means consequently decreasing η values (Figure 3-2b).

This study represents the first attempt to characterize BGR organic turnover parameters for process modeling. In doing so, we tried to cover the whole spectrum of animal excrement and plant-based BGRs. We compared our dataset with BGRs published in the literature (Figure 4). An examination of 85 BGRs from the literature indicate that the range and average pH values in our study are similar to those published (Chen et al., 2012; De la Fuente et al., 2013; Ernst et al., 2008; FNR, 2010; Fouda et al., 2013; Haraldsen et al., 2011; Reinhold, 2013; Rivard et al., 1995; Tambone et al., 2010). In terms of $C/N_{org,}$, we found a sample of 23 BGRs with a range between 5.9 and 26.4, which exceeds the range of BGRs used in this study (De la

Fuente et al., 2013; Ernst et al., 2008; Fouda et al., 2013; Reinhold, 2013; Rivard et al., 1995; Tambone et al., 2010; Wragge, 2013). The BGRs chosen from Sänger et al. (2014) cover only 27% of the C/N_{org} range found in the literature. If we take into account only BGRs that are produced from the same substrates, as in this study, and neglect BGRs derived, for example, from municipal waste, our predicted results cover 35% of the C/N_{org} values.

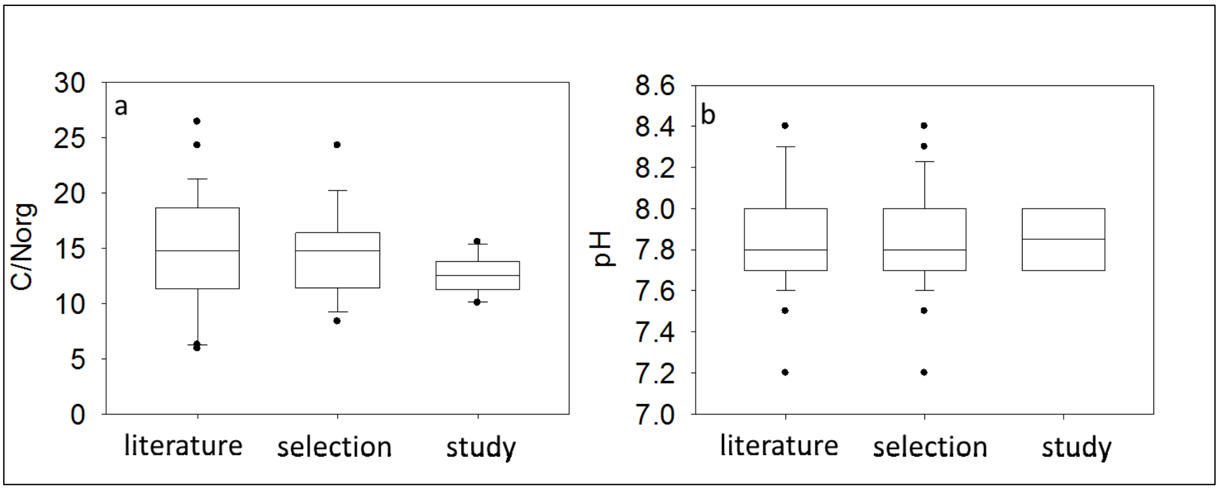

Figure 3-4 (a) C/N_{org} and (b) pH distribution of all BGRs found in the literature (literature), BGRs that are produced from the same substrates as in this study (selection) and BGRs used in our study (study).

3.5.3 Scenario simulation

The scenario simulation showed that the BGRs used as organic fertilizer will lead to C_{org} changes in the range of 0.18% w/w after 100 years under the assumed conditions. We found significant C_{org} differences between the BGRs (Figure 3-3). The error bars for variants where estimations were based on chemical BGR properties were bigger than for variants where the parameters were calculated from inverse modeling. Nonetheless, these uncertainties are smaller than the model error and smaller than errors from measurement of C_{org} in the soil samples (Körschens, 2010).

However, we covered only 27% of all BGRs for parameter η. In general, looking at the high diversity of BGRs, the resulting range of C_{org} values at the end of the scenario can be expected to be even greater. This means that the usage of one general parameter set for all BGRs may lead to considerable errors in SOM change modeling.

3.6 Conclusions

We determined the values for the turnover coefficient k and the synthesis coefficient η and found a linear relationship between those parameters and the chemical properties of BGRs.

For the carbon turnover parameter estimation, we recommend conducting incubation experiments for every BGR or using proposed functions instead of using one general value to represent all BGRs. The preferred method should depend on the purpose. The parameter estimation from chemical composition is easily available and can be beneficial for practical applications. Both of the suggested properties can be determined by simple analytical means. Incubation experiments are time consuming, but the parameters can be defined more precisely. The results of this study are preliminary, but to the best of our knowledge, there is currently no BGR parameterization solution available. However, since our results partially cover the variability of BGRs, more BGRs should be included in the calculations.

3.7 Acknowledgements

The authors would like to thank the anonymous reviewers for their careful reading and very helpful and constructive comments. The present study was supported by the Helmholtz Association of German Research Centres within the Initiative and Networking Fund for the Department of Bioenergy (UFZ).

4 Long-term impact of biogas production on soil organic carbon storage3

4.1 Abstract

Biogas is a widely used and important renewable energy source. It is produced in biogas plants (BGP) that convert plant material and animal excrement into CH_4 and CO_2, leaving biogas residue (BGR). In agriculture, BGR is used as organic fertilizer. We hypothesized that replacing undigested organic fertilizers with BGR leads to a decrease in soil organic carbon (SOC) due to (1) carbon offtake during the anaerobic digestion process and (2) the change in cropping system after biogas production is implemented.

A farm in Central Germany with nine fields that were amended with BGR was selected to study carbon fluxes using the CANDY (CArbon and Nitrogen Dynamics) model. Two scenarios were analyzed. First, a simulation from 1973 to 2050 with a repetition of the cropping system was performed. Here, crop rotation data from 2005 to 2016 was repeated from 2017 onwards. It was used to evaluate the impact of BGR on SOC. Second, we evaluated a scenario where BGR application was replaced by undigested cattle slurry using the same amount of N (kg N ha^{-1}). Additionally, the cropping system from 1973 to 2016 was analyzed to highlight the most important drivers of C_{org} accumulation.

The results demonstrated that BGP installation and BGR application over a ten-year period did not have a negative effect on the SOC. The simulation showed that in 2050, BGR would have the same effect on SOC as cattle slurry if both applications contained equal amounts of nitrogen. The analysis of the cropping system showed that changes in byproduct management, tillage and the use of improved cultivars had a greater impact on C_{org} than fertilization.

4.2 Introduction

Biogas is an important renewable energy resource that decreases CO_2 emissions and can substitute fossil fuels. In 2016, most of the approximately 8,000 operating German biogas plants were part of agricultural farms. These farms use cattle slurry and energy crops to convert biomass into CH_4 and CO_2 (DBFZ, 2015). During anaerobic digestion, approximately 60% of the carbon is transformed into CH_4 and CO_2. This observation supports the hypothesis that the application of biogas residue (BGR) decreases soil carbon and induces soil degradation when compared to the application of undigested organic material. Moreover, a shifting demand in

[3] To be submitted, Prays, N., S., Franko, U.

agricultural products associated with biogas production may lead to changes, inter alia, in crop rotations that have a higher proportion of energy crops and management practices that reduce the recycling of byproducts (i.e., straw and beet leaves) into the soil. Under such conditions, the improvement and maintenance of soil quality in cropping systems may become critical to sustain agricultural productivity and environmental quality for future generations (Franko et al., 2015).

The byproduct of biogas production is BGR. Biogas residues are usually applied as fertilizers to return nitrogen, carbon, and other nutrients to the soil. The effects of BGR application on crop yield, soil chemical, physical and microbial properties have been studied in small-scale experiments (Fouda et al., 2013; Sänger et al., 2014) and short-term field experiments (Prays and Kaupenjohann, 2016; Terhoeven-Urselmans et al., 2009). Nevertheless, there is still little information available on the long-term effects of BGR on soil organic carbon (SOC) (Möller, 2015). Odlare et al. (2011) observed an increase in SOC after eight years of BGR application compared to a control treatment. In contrast, Wentzel et al. (2015) showed that the application of biogas slurry over a 15- to 25-year period had no negative effect on SOC.

Therefore, it is still an open question whether biogas can be used as a bioenergy source without depleting soil carbon stocks. BGR application adds carbon and nutrients to the soil, thus directly affecting soil organic matter (SOM) and long-term soil fertility. In contrast, biogas production could have an indirect effect via changes in the entire cropping system, e.g., in the crop rotation or the implementation of new energy crops. It is not entirely clear whether the largest impact on C_{org} results from direct application of BGR or indirect bioenergy-induced management changes (Möller, 2015).

We hypothesized that the replacement of undigested organic fertilizers with BGRs leads to a decrease in SOC due to carbon offtake during the anaerobic digestion process together with the change in the cropping system after the implementation of biogas production. Therefore, we selected nine fields amended with BGR from a farm in Central Germany to study carbon fluxes using the CANDY (CArbon and Nitrogen Dynamics) model. This model processes site-specific information on soils, crops, weather, and land management to compute carbon stocks and fluxes in the topsoil of agricultural fields.

To determine the sustainability of biogas production, the farmer’s data records were used as input for the CANDY model. We evaluated the cropping system and focused on the changes in SOC stock during the following scenarios: a) the period before BGP installation, b) ten years after BGP installation and c) until 2050 with unchanged conditions.

4.3 Material and methods

4.3.1 Study area and farm environment

The farm is located south of Saxony-Anhalt, Germany. The long-term mean annual temperature is 9.6°C, and the long-term mean annual precipitation is 536 mm. The soils are derived from loess and sand loess. The farm produces market and fodder crops as well as milk. A portion of the crops as well as the cattle slurry from dairy cows and cattle manure are used as substrates for the on-farm BGP.

Five fields with typical cropping systems were selected. Three fields (62, 65, and 85) were less than 2 km from the biogas plant. Two fields (25 and 44) were the most distant (approximately 5 km) from the BGP. Between 1973 and 2003, four of the selected fields were split into subplots (e.g., field = 62, subplot = 620 and 621) if the chosen field was divided into two fields more often than 15% of the studied time-period. We evaluated nine fields, including 250, 251, 440, 441, 620, 621, 650, 651 and 850.

4.3.2 Biogas plant and BGR

The biogas plant was established in 2005 and only uses substrates that are produced on the farm. The feeding mixtures consist of crops (27.6% maize silage, 2.1% lucerne, 1.1% grass silage, and 2.9% cereals) and animal excrement (52.9% cattle slurry and 13.4% cattle manure). The substrate is wet digested in two fermenters for 94 d at 40°C. The BGR is characterized by 5.4 (± 0.5)% of dry matter (DM), 7.9 (± 0.2)% total nitrogen, 4.5 (± 0.9)% NH_4-N, a C/N_{org} ratio of 12.2, a pH of 7.9, k of 0.41 d^{-1} and η of 0.86 (Table 4-3). Each chemical parameter of the BGR used in this study is a mean value from samplings in four different years. The decomposition rate coefficient k describes the rate of organic matter decay and the synthesis coefficient η describes SOM creation from BGR. These values were calculated from the pH and C/N ratio according to Prays et al. (2017).

4.3.3 Model calculations and input data

We used the simulation model CANDY (CArbon and Nitrogen Dynamics, http://www.ufz.de/index.php?en=39725) as described in detail by (Franko et al., 1995). The model requires a site-specific description of the soil profile (texture, wilting point, water capacity, saturated conductivity, bulk and particle density), meteorological data (air temperature, precipitation, and global radiation or sunshine duration), and management information (tillage, fertilizer, organic amendments, and harvest). One important application of the CANDY model is the calculation of the long-term dynamics of organic matter turnover in arable soils and the short-term dynamics of nitrogen transformation (Franko et al., 1995). We used the CANDY model to calculate the SOC concentration (C_{org}) in the upper 30 cm of soil as well as the yearly N uptake by the crops. Model initialization was performed by adjusting the initial value of C_{org} manually during the spin-up run to fit the SOC values to measured values.

4.3.4 Soil organic carbon measurements

Soil organic carbon measurements were required for model validation. Therefore, the C_{org} data from the farmer as well as our measurements were used. Data from the farmer included measurements from fields 250 and 251 from 2000 through 2002 and fields 620, 621, 650, 651, and 850 from 2000. Our soil sample measurements were taken between 27.08.2013 and 05.11.2013. On each field, five to nine mixed samples were taken from 0-30 cm, each with approximately 1 kg of soil. For the organic carbon analysis, samples were milled and analyzed with a CN auto-analyzer (LECO Instruments, St. Joseph, USA). Mean values and standard deviations were calculated. RMSE was calculated between the modeled and measured C_{org} values and was used for validation.

Further input data is described in the following section.

4.3.5 Climatological time series

Daily means of air temperature (°C), daily sums of precipitation (mm), and daily sums of sunshine duration (hours) were collected from the meteorological station in Bad Kösen prior to 2007. This meteorological station is approximately five kilometers away from the farm (beeline) and therefore reflects the same climatic conditions of the farm. After 2007, data from the meteorological station in Bad Kösen were no longer available, and we used data from the station in Naumburg/Saale-Kreipitzsch (Deutscher Wetterdienst, 2015). The farm and the meteorological station are approximately six kilometers apart. Nonetheless, the sunshine duration data from 1992-1993 and 2009-2014 as well as the air temperature between 1985 and 1993 were missing. For this period data, gaps were filled from the meteorological station in Osterfeld, which is approximately 30 km away from the farm. For the predictions between 2016 and 2050, the weather data from a 30-year period (1987 to 2016) were repeated.

4.3.6 Soil data

Texture and bulk density [g cm-³] were extracted from the soil map VBK50 (scale 1:50,000) (LAGB, 2012) and were used to derive the following parameters: field capacity [Vol.-%], wilting point [Vol.-%] and saturated conductivity [mm d^{-1}] according to the German mapping guideline KA5 (Ad-hoc-AG Boden, 2005). Particle density was set to 2.65 [g cm-³].

The soil map provided information about silt dominated soils, which are typical in the region (Table 4-1). As soil heterogeneity in this region is low, we selected the most representative soils for modeling: soil 1 for fields 250, 251, 620, 621, 650, and 651; soil 2 for fields 440 and 441; and soil 3 for field 850.

Table 4-1 Soil properties of the investigated fields that were used for carbon flux modeling. BD = bulk density, PV = pore volume, FC = water content at field capacity, WP = water content at wilting point, Ks = saturated conductivity

Soil	Horizon	depth [dm]	BD [g/cm³]	PV [Vol-%]	FC [Vol-%]	WP [Vol-%]	Ks [mm/d]	Clay [M-%]	Silt [M-%]
1	1	3	1.23	44	37	16	130	18	74
	2	5	1.42	43	37	12	120	16	79
	3	7	1.48	44	37	16	130	24	75
	4	20	1.52	43	37	12	120	14	79
2	1	3	1.23	44	37	16	130	18	74
	2	4	1.52	43	37	12	120	14	73
	3	8	1.53	43	37	20	90	25	70
	4	20	1.52	43	37	12	120	14	79
3	1	3	1.23	44	37	16	130	18	74
	2	5	1.34	44	37	16	130	21	72
	3	7	1.51	44	37	16	130	17	78
	4	20	1.52	43	37	12	120	14	79

4.3.7 Cropping system

To model the cropping system, information on sowing and harvest (date and yield), fertilization (date and amount) as well as date and depth of tillage were required. From 1973 to 1991, crop yields, application rates of mineral and organic fertilizers and information on tillage were available. Dates of sowing and organic fertilization missed completely. Seventy percent of the dates were available between 1973 and 1991 for mineral fertilization, tillage and harvest. All gaps were filled according to typical farm management from other years or according to good agricultural practices (Doleschel and Frahm, 2014). From 2003 to 2016, all required data were available. From 1992 to 2002, only crop rotation data were available. After consulting with the farmer, this data gap was filled by repeating the soil management from 2003 to 2016 with respect to the cultivated crop. During this period, cattle slurry was used instead of BGR, with an equivalent concentration of nitrogen. For data and trend analyses, we used the data from 1973-1991 and 2003-2016.

We calculated the yields in dt ha^{-1}, N application (mineral, organic and total) and N uptake by crops in kg ha^{-1} as well as the yearly means, standard deviations and linear trends. For every year, one mean value for all fields was calculated. The periods from 1973-1991 and 2003-2016 were compared. Trends were analyzed by a one-way ANOVA. Tukey's 'Honest Significant Difference' method (HSD) was used to compare mean values and to assess the significance of the differences between mean values. Trends were considered significant when $p < 0.05$. For silage maize and sugar beet yields, the trends were considered significant when $p < 0.1$. All statistical analyses were performed using R version 3.3.1 (The R Foundation for Statistical Computing, 2016).

4.3.8 Crop parameters

The model parameters of crops that were cultivated between 1973 and 2016 are listed in Table 4-2.

Table 4-2 List of crops and their parameters for modeling. N = nitrogen concentration in above ground biomass (yield + by-product); HI = harvest index, relation of by-product to main product; CEWR = N amount in harvest residues independent from yield (constant); FEWR = factor between N in harvest residues, roots and yield, RP = raw protein content, sp. = spring, DM = reference dry matter during harvest

English name	Latin name	DM [%]	N [%]	HI	CEWR [kg ha^{-1}]	FEWR [kg kg $^{-1}$]
alfalfa (perennial)	*Medicago sativa*	20	0.6	-	100	0.1111
Carrot	*Daucus carota*	15	0.22	-	23	1
clover-grass (perennial)	*Trifolium pratense*	18	0.52	-	107	0
clover-grass (permanent)	*Trifolium pratense*	18	0.52	-	105	0
durum wheat	*Triticum durum*	86	2.5	0	5.2	0.052
field bean	*Vicia faba*	86	5.6	-	37	0
Mustard	*Brassica juncea*	17	3.4	0	20	0
oats 10%RP	*Avena sativa*	88	1.92	1.1	6.2	0.0812
oil radish	*Raphanus sativus*	10	0.41	-	28	0
Potato	*Solanum tuberosum*	22	0.35	-	3.2	0.264
summer rape	*Brassica napus*	88	4.47	1.6	6.6	0.0984
silage maize	*Zea mays*	33	0.38	-	23.1	0.0292
sp. barley brewing 11%RP	*Hordeum vulgare*	86	1.86	0.7	5.2	0.0699
sp. barley fodder 13%RP	*Hordeum vulgare*	86	2.19	0.8	5.2	0.05935
spring rye 11%RP	*Secale cereal*	86	1.96	0.9	8	0.0816
spring wheat 13%RP	*Triticum aestivum*	86	2.36	0.8	5.2	0.0551
sugar beet and fodder beet	*Beta vulgaris*	23	0.36	0.7	8	0.1111
winter barley 13%RP	*Hordeum vulgare*	86	2.19	0.8	8	0.073
winter rape-seed 23%RP	*Brassica napus*	88	4.47	1.6	6.6	0.0984
winter rye 11%RP	*Secale cereal*	86	1.96	0.9	8	0.0816
winter rye 14%RP	*Secale cereal*	86	2.38	0.9	8	0.0672
winter wheat 13%RP	*Triticum aestivum*	86	2.36	0.8	8	0.0678

4.3.9 Fertilizer parameters

The fraction (%) of NH_4-N from total N is desicive for mineral fertilizers. For ammonium phosphate, solution of urea and ammonium nitrate (UAN), urea and sulfur acid ammonia, this fraction was assumed to be 100%. For ammonia nitrate and calcium ammonia nitrate, this fraction was assumed to be 50%.

In addition to BGR, cattle manure, cattle slurry, liquid manure and, in rare cases, pig slurry were applied as organic fertilizers (Table 4-3). Different parameters were taken from the CANDY database.

Table 4-3 Organic fertilizers and their parameters used for modeling. DM = dry matter content, C in DM = carbon content in dry matter, C_{org} = organic carbon concentration, N_{org} = organic nitrogen concentration, N_{min} = mineral nitrogen concentration, k = decomposition coefficient, η = synthesis coefficient

Fertilizer	DM [% w/w]	C in DM [% w/w]	C_{org}/N_{org}	$C_{org}/(N_{org}+N_{min})$	N_{min}/N_{org}	k [d^{-1}]	η
biogas residue	5,4	0,41	12,2	5,2	0,93	0.41	0,86
cattle manure	0.25	0.31	18	14.07	0.28	0.1	0.6
liquid manure	0.02	0.24	16	2.18	6.33	0.05	0.65
cattle slurry	0.08	0.34	16	7.77	1.06	0.05	0.65
pig slurry	0.08	0.31	13	4.66	1.79	0.05	0.65

4.3.10 Scenarios

A simulation from 1973-2050 was performed to evaluate the long-term effects of BGR on soil. Therefore, the cropping system and crop rotation from 2005 to 2016 were repeated from 2017 onwards. To determine the effects of BGR on carbon storage in soil, a second scenario was simulated where the application of BGR was replaced with undigested cattle slurry with an equivalent concentration of N in kg N ha^{-1}.

4.4 Results

4.4.1 Organic carbon

Measured C_{org} values in own soil samples confirmed the results from the farmer from the farmer. Thus, we combined both measurements into one subset and used it for model validation. Standard deviations were calculated for our samples only and ranged between 0.11% and 0.17%.

The modeled trends of C_{org} were similar on all fields and soils (Figure 4-1). Prior to 1991, C_{org} decreased up to 0.1% w/w, whereas from 1991 to 2005, a slight increase of approximately 0.1% w/w was observed. After the installation of the biogas plant in 2005, C_{org} values in fields 250, 251, 620 and 621 were stable until 2016 and changed less than 0.05% w/w In the other fields, an increasing trend is assumed and an increase in C_{org} of over 0.1% w/w is predicted by the model. The difference between the measured and simulated value (RMSE) was smaller than the errors from the C_{org} measurements in the soil samples.

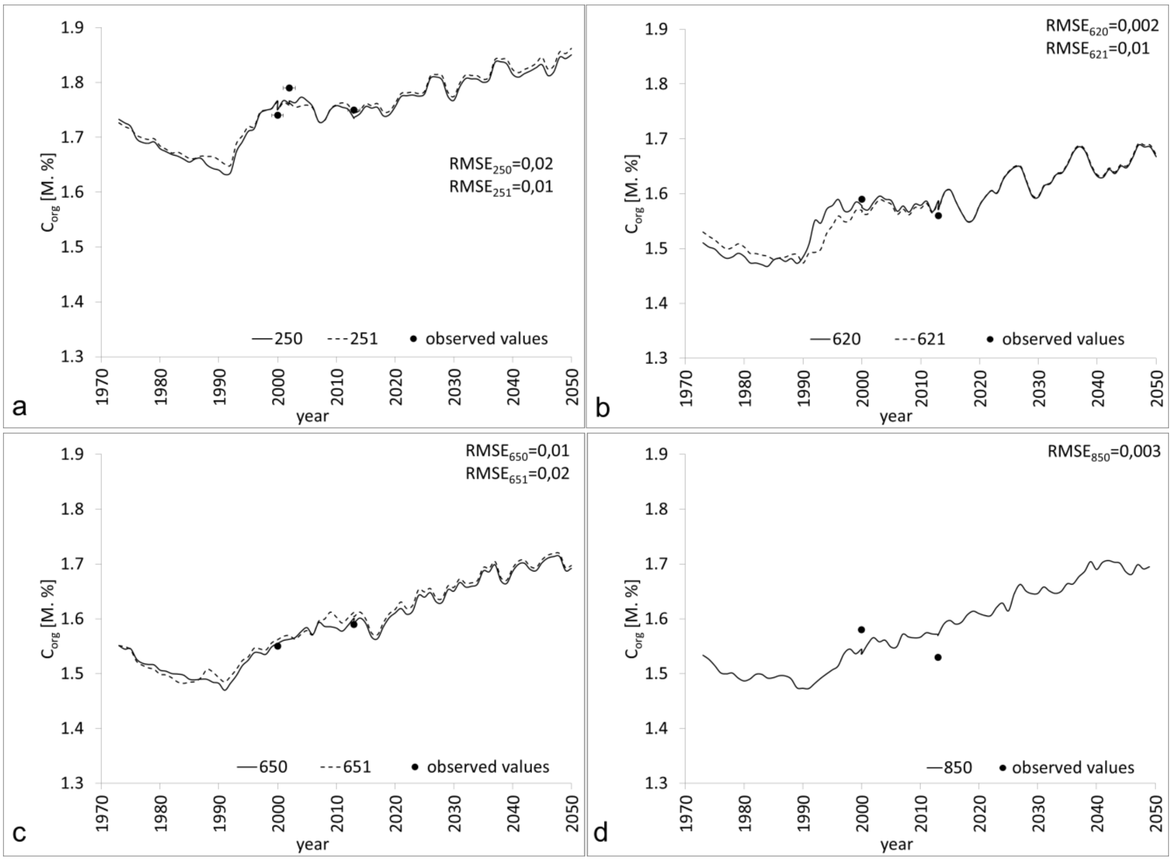

Figure 4-1 Soil C_{org} change over time in fields 25, 62, 65, 85 which were split into subplots. (e.g., field = 62, subplot = 620 and 621) if the chosen field was divided into two fields more often than 15% of the studied time-period.

The modeled scenario with cattle slurry instead of BGR provided similar results. The C_{org} differences between both scenarios ranged from 0.001% and 0.02% in 2050. Biogas residue and cattle slurry appear to have the same impact on carbon storage, even though approximately 33% less C_{org} is applied to the soil when BGR is used (with equivalent N concentrations).

4.4.2 Cropping system

The cropping system changed over the 40 years of farming. From 1973 to 1991, 17 different crops were cultivated within the crop rotation. After 1991, the crop rotation was oriented for market development and consisted of sugar beets, winter rape or silage maize, winter wheat planted twice, followed by summer barley. During the experimental period, the area share of root crops (sugar beet and potato) decreased in selected fields from approximately 30% in 1973-1991 to approximately 10% after the establishment of the biogas plant. The area share of silage maize increased from 7% to 28% during the same period.

Prior to 1991, byproducts such as straw and sugar beet leaves were removed after harvest. From 1991 onwards, byproducts were incorporated into the soil, and in 1995, the tillage system was changed to no-till.

From 1973 to 2016, crop yields improved continuously (Figure 4-2). The grain yield of winter wheat increased from approximately 50 dt ha^{-1} to approximately 90 dt ha^{-1}, spring barley increased from 40 dt ha^{-1} to 50 dt ha^{-1}, silage maize increased from 330 dt ha^{-1} to 450 dt ha^{-1} and sugar beets doubled from 1973 to 2016 from 330 dt ha^{-1} to over 660 dt ha^{-1}. The only change that was not statistically significant was the increase of winter barley from less than 50 dt ha^{-1} to approximately 60 dt ha^{-1}. Winter rape was not included in the trend analysis because it was only cultivated in 1991 and after 2004.

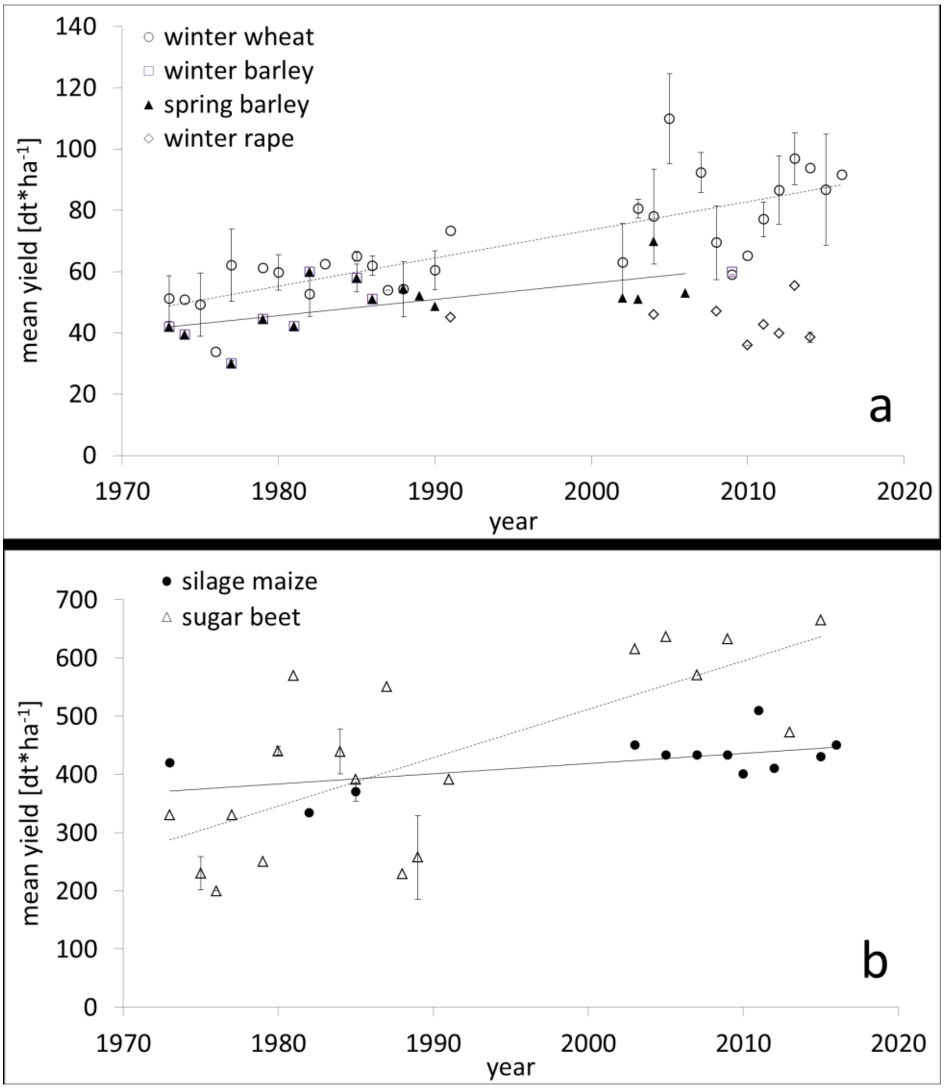

Figure 4-2 Mean annual yields and standard deviations of a) wheat, barley and rape, b) maize and sugar beets. Trend lines are shown for winter wheat and sugar beets (dashed line) and spring barley and silage maize (continuous line)

The cropping system as well as the fertilization regime changed over the 40 years of farming. Prior to 1991, during the vegetation period, cattle slurry was used as an organic fertilizer. Calcium ammonium nitrate and urea were applied as mineral fertilizers. After harvest in autumn, cattle manure was applied before plowing. After 2003, calcium ammonium nitrate, urea and ammonium nitrate (UAN) were primarily used as mineral fertilizers. Cattle slurry was replaced after 2005 with BGR, although cattle manure is still applied as organic fertilizer. The frequency of the BGR application on fields which are closer to the biogas plant and which are furthest away was in the same range of 4 to 6 times from 2005-2016.

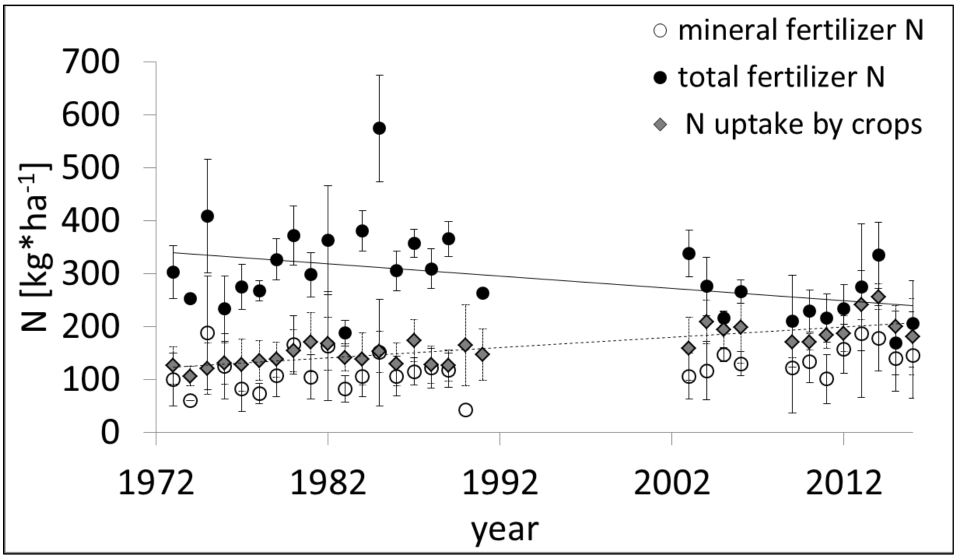

Figure 4-3 Mean yearly sum of mineral and total (mineral and organic) fertilizer N application and total yearly N uptake by crops. The trend line for total fertilizer N ($p<0.05$) is represented by a continuous line and the trend line for N uptake by crops ($p<0.05$) is represented by a dashed line

Mineral fertilizer usage increased from approximately 105 kg N ha^{-1} in the 1970s to approximately 150 kg N ha^{-1} after 2010. In contrast, the usage of cattle slurry and cattle manure as organic fertilizer decreased significantly ($p<0.05$) from 190 kg N ha^{-1} in the 1970s to approximately 90 kg N ha^{-1} after 2010. Prior to 1991, applied N was primarily derived from organic fertilizer (Figure 4-3). In 2015, the largest fraction of total applied N was derived from mineral fertilizers. Overall, the trend of total applied N decreased from the 1970s to 2015. At the same time, the N uptake by the crops increased from approximately 126 kg N ha^{-1} in the 1970s to 195 kg N ha^{-1} from 2005 to 2015.

4.5 Discussion

4.5.1 C_{org} development prior to implementation of the biogas plant

According to our model, C_{org} decreased in all fields from 1973 to 1991, despite high N application rates with organic fertilizer, likely as a result of the intensive use of root crops and the removal of byproducts (Figure 4-3). The amount of applied organic fertilizer in the field was more than two times the N uptake by crops. An increase in SOC is expected following the application of animal manure, as reported by many studies. For example, (Gami et al., 2009) observed an increase in SOC (0-30 cm) of approximately 19,100 kg C ha^{-1} after cattle manure was applied for 25 years, compared to plots with mineral fertilizer. After pig manure was applied for 22 years in China, the surface soil layer (0-15 cm) increased by 3,800 kg C ha^{-1} compared to when mineral fertilizer was applied (Huang et al., 2010). However, and in agreement with our results, some studies reported no significant change or a negative change in SOC stocks following manure application (Angers et al., 2010; Franzluebbers et al., 2001). Clearly, there is significant variability in the change in SOC after manure application.

Soil organic matter storage capacity in agroecosystems varies with soil type, climate and agricultural management practices (Angers et al., 1997). After 1991, C_{org} increased in all fields. This increase is a result of the change in byproduct management (increased carbon input) and the introduction of no-till (reduced carbon turnover) practices. In our study, the farmers incorporated byproducts such as straw and sugar beet leaves into the soil after 1991. Schlesinger (2000) showed that approximately twice as much carbon can be retained from crop residues than from manure, as indicated by our results. Liao et al. (2015) showed that incorporation of crop residue can significantly contribute to the maintenance of SOM in agricultural systems. Smith et al. (2005) reported that the input of crop residue could attain the highest rate of C sequestration in comparison with that of mineral N fertilizer.

In addition to residue management, it is likely that soil tillage has a substantial impact on SOM turnover. Long-term experiments have shown that plowing can lead to decreases in soil carbon (Franko and Spiegel, 2016). Frequent soil disturbances (i.e., tillage) expose protected organic matter and increase the rate of decomposition, resulting in lower steady-state SOC (Grandy and Robertson, 2007). In our study, no-till practices were instituted after 1995. A comparison of conventional cropping systems to those converted to no-till indicate higher median concentrations of carbon in the no-till cropping systems (Kopittke et al., 2017). Additionally, other studies have shown that conventional tillage is associated with decreased SOC compared to no-till (Alvarez, 2005). Abreu et al. (2011) stated that intensive tillage has greatly reduced the organic carbon content of cropland in Oklahoma. They observed a greater C_{org} in no-till fields compared to tilled fields across all locations and depths. In contrast, no significant differences were found between tillage treatments in the total organic carbon storage to a depth of 60 cm in a range of soils in eastern Canada under continuous corn and small grain cereal production (Angers et al., 1997).

4.5.2 C_{org} development after the implementation of the biogas plant

During the cycling of organic matter, we expect a carbon deficit because CH_4 and CO_2 are extracted during anaerobic digestion. Furthermore, organic farmers argue that fertilizing with BGR may impair the microbiota and soil fertility because it contains more mineral N and less organic carbon than undigested manure (Insam et al., 2015).

Nonetheless, C_{org} measurements together with the modeling results showed that the SOC did not decrease during the ten years of biogas production on the farm. Furthermore, the CANDY simulation predicted an increase in C_{org} until 2050, even though BGR supplies less carbon and biomass than cattle slurry for equivalent amounts of N. Thus, at the farm scale, and when applied according to the good agricultural practice, anaerobic digestion is not averse to other manure treatment options. This result is possibly due to the higher residence time of the carbon from BGR compared to undigested material. Presumably, SOC pools do not suffer from reduced carbon input (Insam et al., 2015). Thomsen et al. (2013) confirmed these findings and suggested that the retention of plant-derived carbon in soil is only slightly affected by anaerobic digestion over a greater time scale. Möller (2015) concluded that carbon losses during the anaerobic digestion process are compensated by lower C degradation after field application. Furthermore, De Neve et al. (2003) reported that the organic carbon in BGR is more stable compared to other organic wastes. In these reports, stability is used synonymously with residence time. In our model, the higher residence time is a consequence of the improved transfer of carbon from fresh organic matter to the SOM pools. The SOM synthesis coefficient η of BGR is higher than that of cattle slurry or manure, and thus its contribution to SOM is higher (Prays et al., 2017). Numerous studies have compared BGRs with other organic amendments (Abubaker et al., 2013a; Abubaker et al., 2013b; Möller and Stinner, 2009; Stumpe et al., 2012) and concluded that there is no long-term negative effect on the SOC if manures are anaerobically digested.

The analysis of the cropping system revealed that the fraction of silage maize increased substantially after the biogas plant was established. This result corresponds with other observations in Germany (FNR, 2010).

Important cropping system improvements allow for yield increases and a decrease in applied organic fertilizer over time. From 1973 to 2016, the yields of most crops increased significantly. After 1991, significantly less organic fertilizers and N were applied to soils compared to 1973-1991. These developments are assumed to be independent from the establishment of the biogas plant. Nevertheless, increased yields lead to improved carbon input due to additional roots and byproducts (Franko et al., 2011; Merbach and Schulz, 2013; Wiesmeier et al., 2014).

4.5.3 Limits and outlook

To the best of our knowledge, this study is the first analysis of the impact of biogas production at the farm scale. The farm in this study is a typical farm in Central Germany and the biogas plant is representative of other plants in the area with respect to the electrical capacity and substrate mix (FNR, 2010). In addition, the studied region is characterized by loess soils, which have beneficial agricultural properties, such as good air and water regimes. These conditions could mask the effects of BGR fertilization and the cropping system, which may be more apparent on sandy soils. Our analysis is restricted by the low number of C_{org} observations per field used for the validation. More validation points would improve the results.

The application of this approach in areas with different soils or at larger scales such as in landscapes and catchment areas requires future studies, which will improve our understanding of the impact of biogas production on C_{org}. However, we consider our results as a representative example for highly productive soil in Central Germany.

4.6 Conclusions

In summary, our findings suggest that at the farm scale, the replacement of undigested organic fertilizers with BGR does not lead to a decrease in SOC within ten years of biogas plant operation. Furthermore, our model indicated that, despite carbon removal during anaerobic digestion, the C_{org} did not decrease under the tested cropping conditions (until 2050). The incorporation of crop residues, no-till practices and successive yield growth were the most important drivers that influenced soil carbon. The direct effect of fertilization with BGR appears to be of minor relevance at the farm scale.

4.7 Acknowledgements

We cordially thank the farm in Saxony-Anhalt, which wishes to remain anonymous, for the opportunity to conduct our research.

This study was funded by the Helmholtz Association of German Research Centres within the Initiative and Networking Fund for the Department of Bioenergy (UFZ) and was supported by the HIGRADE graduate school.

5 Biogas production and soil related carbon fluxes - a regional analysis[4]

5.1 Abstract

The inclusion of biogas production into the agricultural system has modified crop management and as a result the soil organic carbon (SOC) cycle of the agricultural landscape. To evaluate the effects for the German federal state of Saxony this study determines: (1) the share of agricultural land required for biogas production, (2) the change in regional carbon input fluxes to soil during the time of the establishment of the biogas production considering also the quality of sources of different fresh organic matter (FOM) for the formation of SOC and (3) the differences in carbon input to SOC between the area influenced by biogas production (here ‘biogas fingerprint area’ (BFA)) and the surrounding arable land.

Based on the location of biogas plants the region was subdivided into biomass providing units (BPUs) where a part of the arable land was considered as affected by biogas production (BFA). We hypothesized that each biogas plant uses a specific substrate mix according to its capacity. The carbon fluxes for each BPU were estimated for the years 2000 (without biogas plants) and 2011 (with biogas plants). For the year 2011, the analysis included the area demand for production of biogas feedstock and digestate recycling.

On average 17.6% of the BPU agricultural land was required to supply the biogas plants and dispose of their digestate. Per kilowatt installed electrical capacity this equates to 2.0 ha, including inter alia 0.4 ha for energy crops. Highest area requirements have been observed for biogas plants with less than 500 kW installed capacity. Between 2000 and 2011 the total carbon flux into soil increased by 2.1%. When considering the quality of different FOM sources the gain in carbon input was 2.8%. The BFAs showed higher carbon input to soil than the surrounding agricultural land due to high contributions from digestate and crop residues (esp. agricultural grass). This compensated the low carbon input from crop by-products (e.g. straw).

5.2 Introduction

Soil is one of the most important and most complex natural resources and is an essential contributor to the global ecosystem, providing a regulatory system that supports a multitude of ecosystem functions and services (Adhikari and Hartemink, 2016; Garrigues et al., 2012;

[4] Witing, F., Prays, N., O'Keeffe, S., Gründling, R., Gebel, M., Kurzer, H.-J., Daniel-Gromke, J., Franko, U., 2018. Biogas production and changes in soil carbon input - A regional analysis. Geoderma 320, 105-114.

Podmanicky et al., 2011). Soil organic matter (SOM) and its major component soil organic carbon (SOC) are fundamental to soil and its ecosystem functions in particular the sequestration of carbon (Campbell and Paustian, 2015; Podmanicky et al., 2011; Yigini and Panagos, 2016).

Biogas production within conventional agricultural systems has been promoted as an integrated approach to support nutrient cycling, while mitigating greenhouse gases emissions from conventional fossil energy production. Germany is the largest biogas producer in the European Union, with almost 8,700 biogas plants installed in 2016 (Daniel-Gromke et al., 2017a; Daniel-Gromke et al., 2017b). A previous study by Franko et al. (2015), for the region of Central Germany, identified a number of hot spots where the usage of carbon may raise a conflict between sustaining SOC and producing bioenergy. The expansion of the agricultural system to include bioenergy production has resulted in an adaption of the agricultural management (e.g. cultivated crops, digestate application instead of slurry), which in turn has changed the carbon input to soil within these agricultural landscapes. At the same time biogas production is heavily influenced by the regional availability and variability of feedstock.

To date, no general approach has been developed to understand the potential influence of bioenergy production on regional soil carbon cycling. It is a challenge to tackle the additional complexity which biogas production can introduce into agricultural systems (Arthurson, 2009; Barbosa et al., 2014; Möller and Müller, 2012). Therefore, the aim of this study was an ex-post evaluation of the biogas production within the agricultural landscape of a case study region. For each biogas plant within the federal state of Saxony we estimated the agricultural area required for the provision of biogas feedstock and recycling of digestate, proposing the combination of this as “biogas fingerprint area” (BFA) of a biogas plant. The carbon input to arable soil has been estimated for two separate years 2000 (without biogas production) and 2011 (with biogas production). Here also the quality of different sources of fresh organic matter (FOM) regarding the formation of new SOC was considered. Furthermore, for the year 2011 we compared the carbon input on the BFAs and the arable land not affected by biogas production.

5.3 Material & Methods

5.3.1 Spatial units of investigation

The federal state of Saxony, in East Germany was used as the study region. During the last decade a rapid development of the biogas industry has been observed in this area (Grunewald, 2012). For regional subdivision of Saxony and main spatial element of the study we used ‘biomass providing units’ (BPU) which separate catchment areas (i.e. for agricultural substrates) from competing biogas plants as defined by Franko et al. (2015). The location and capacity of the biogas plants within Saxony were determined by Das et al. (2012). Relevant cropping and livestock data was aggregated to the BPU level.

We assumed that every BPU had a closed matter cycle regarding agricultural substrates in the context of biogas production. The feedstock demand of a biogas plant was supplied by the agricultural area within the associated BPU, with the biogas digestate being returned to the same area. The agricultural land required for the production of biogas feedstock and disposal of digestate was defined as “biogas fingerprint area” (BFA) of a BPU (section 5.5). The soil related carbon flows within the BFAs are assumed to differ from the surrounding agricultural land (section 5.6). It was hypothesized that depending on the installed electrical capacity and the feedstock mix of the biogas plant, as well as the regional agricultural parameters (e.g. crop mix and yields, livestock mix, management of the arable land), every biogas plant will have its own unique BFA.

For each BPU the associated land use considerations are shown in Figure 5-1. The crop mix of the BFA corresponds to the direct and indirect demands for biogas feedstock. Depending on the fertilization intensity, the agricultural area needed for the application of digestate may be smaller or larger than the area for production of biogas feedstock. If the area needed is larger, an additional area for the application of biogas digestate was considered to be necessary. Prior to the implementation of biogas production, livestock excrement were applied to all arable land (year 2000). However, with the installation of biogas plants (year 2011), it was assumed that excrement not used for biogas production were applied only to the BPU area outside of the BFA.

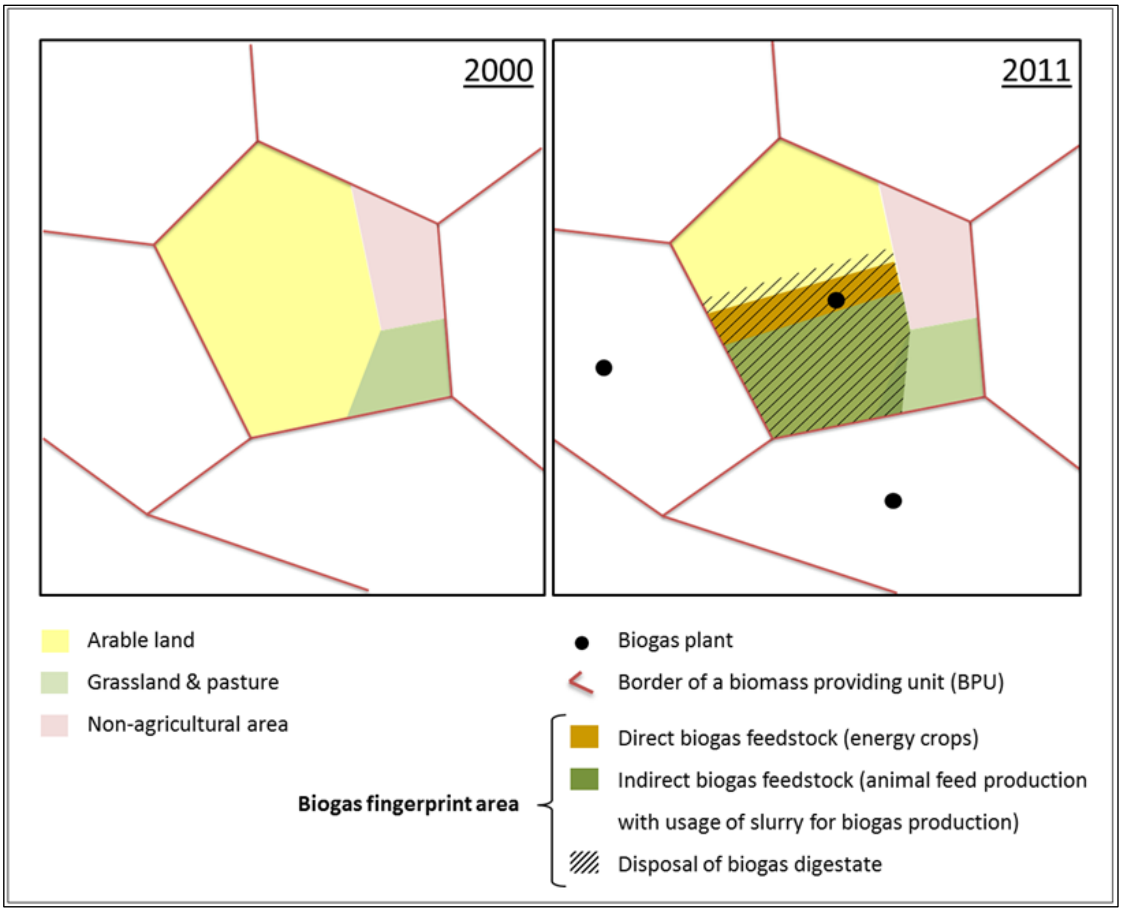

Figure 5-1 Graphical representation of a 'biomass providing unit' (BPU) and its associated land use categories for the base year of 2000 (without a biogas plant) and the year 2011 (with a biogas plant). For the 2011 time step a 'biogas fingerprint area' (BFA) is shown, to denote the area where the cycling of agricultural matter and the input of carbon to SOM is influenced by biogas production.

5.3.2 Regional agricultural parameters

5.3.2.1 Land use and agro-economic regions

The federal state of Saxony (approx. 18,400 km²) is dominated by arable land-use (Figure 5-2). Due to the very fertile loess soils, which cover a large part of the study area, 52% of the region is used for agricultural purposes. Saxony can be subdivided into three main "agro-economic regions", based on characteristics of soil, landscape characteristics and their associated agricultural activities (LfL, 1999). These include: (1) Saxon heath and pond landscape, (2) Saxon loess region, (3) Saxon low mountain range and foreland. For more information see supplementary material (Table A1).

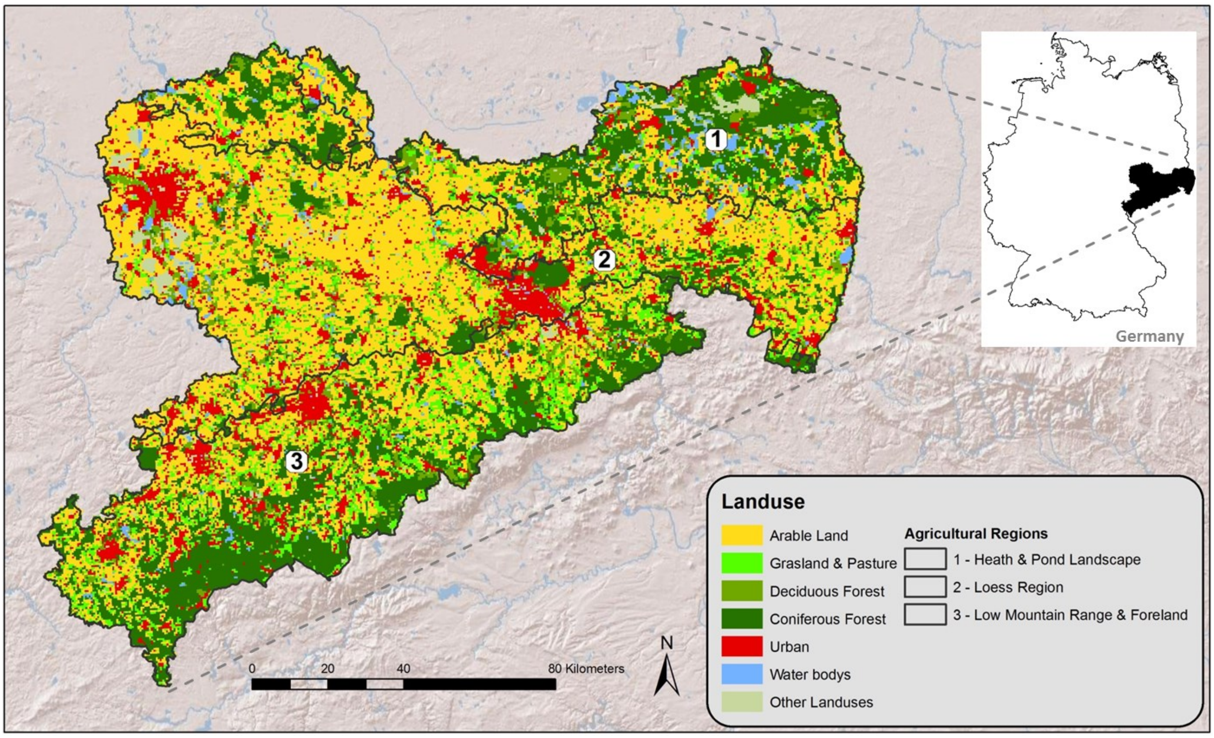

Figure 5-2 Location in Germany, land-use and the agro-economic regions of Saxony

5.3.2.2 Crop harvest areas and yield

Data on crop harvest areas and crop yield for 20 different crops as well as catch crops have been provided by the ‘State Agency for the Environment, Nature Conservation and Geology of Saxony’ (LfULG). Crop harvest areas are derived from statistics on municipality level (year 2000) and InVeKoS data (Integriertes Verwaltungs- und Kontrollsystem) for the year 2011. Crop yield data was based on analysis of the software BEFU, a fertilization advisory system used by Saxon farmers (Förster, 2013). Essential crops included in the analysis, as well as their average areal share and yield for the period 2000-2011 are shown in Table 5-1. For these years cereals were found to be the dominant crops (58%) in Saxony, followed by winter rape (15%) and maize for silage (9%).

Non-harvested biomass was characterized into two groups, crop residues and crop by-products, -based on the potential usage of the material (see also section 845.6). While residues like crop roots and stubble were assumed to be left on the field, the fate of by-products depends on farmers decision: by-products (i.e. straw) can be left on the field or carried away to be used as litter for the livestock stable or sold on the market. Based on expert knowledge, at the state agency LfULG, it was assumed that by-products of relevant crops were removed from approx. 20% of the arable area.

Table 5-1: Average crop shares and crop yields within the agro-economic regions of Saxony for the period 2000-2011

	Heath & Pond Landscape		Loess Region		Low Mountain Range & Foreland	
	Share [%]	Yield [t ha^{-1}]	Share [%]	Yield [t ha^{-1}]	Share [%]	Yield [t ha^{-1}]
Winter Wheat *(Triticum aestivum)*	13.7	6.4	31.9	7.2	15.5	6.5
Winter Barley *(Hordeum vulgare)*	13.0	5.9	14.5	6.9	11.8	60.0
Winter Rye & Triticale *(Secale cereal & Triticosecale)*	26.1	5.0	6.8	6.2	8.3	5.5
Spring Cereals *(Hordeum vulgare & Triticum aestivum)*	4.0	4.4	5.2	5.0	19.8	4.7
Winter Rape *(Brassica napus)*	12.2	3.4	16.9	3.9	14.3	3.8
Maize for Silage *(Zea mays)*	9.7	42.4	7.7	46.7	10.5	44.0
Field Grass *(Lolium multiflorum & Lolium perenne)*	2.5	30.9	1.6	38.5	5.4	39.0
Clover Grass *(Trifolium pretense & Lolium multiflorum)*	1.3	38.7	1.3	39.7	6.5	38.0
Other[1]	17.4		14.0		8.1	

[1]fallow, sugar beet *(Beta vulgaris)*, grain maize *(Zea mays)*, vegetables, legumes, sunflower *(Helianthus annuus)*, potatoes *(Solanum tuberosum)*

5.3.2.3 Excrement

We calculated the amount of excrement available for field application or biogas production ($excr_{av}$ in t a^{-1}) based on livestock statistics on district and municipality levels (StLa, 2016a; StLa, 2016b). Therefore the total amount of excrement produced from all livestock was corrected for the amount that is left on pasture during grazing (StLa, 2012b). For each animal group i the specific average annual amount of excrement (*excri* in t a^{-1};(LfULG, 2015), the share of grazing time within one year (*grzt* [-]) and the number of individuals within this group (n) was used to calculate the amount of excrement which we assumed to be slurry:

$$excr_{av} = \sum_i \left(n_i \cdot excr_i \cdot (1 - grzt_i)\right) \tag{1}$$

The data was aggregated from municipality level to BPU level using the areal share of municipalities in the BPUs. Within the BPUs the excrement not used for the production of biogas was assumed to be equally distributed on arable land outside of the BFA.

5.4 Profile of regional biogas plants

5.4.1 Deriving representative feedstock mixes

The substrate mix used for the production of biogas can vary widely between individual biogas plants making it difficult to parameterize in large scale assessments. Therefore, the demand for biomass substrate was estimated using the approaches outlined in O'Keeffe et al. (2016). in collaboration with the DBFZ (Deutsches Biomasseforschungszentrum) (Ponitka et al., 2015). Six biogas clusters with representative feedstock profiles for agricultural biogas plants were identified for the federal state of Saxony (Table 5-2). The biogas clusters were differentiated by installed capacity and for the capacity class 151-500 kW also by agro-economic region. For the other capacity classes, a regional differentiation was not possible due to data limitations. The representative feedstock profiles for each biogas cluster were used to generate the appropriate feedstock demand for each biogas plant based on their individual installed electrical capacities (kW_{el}). Manure and slurry have been merged to the feedstock class "animal excrement" using the differences in dry matter and carbon content of dry matter to be consistent with the calculation of available excrement (section 5.3.2.3).

Table 5-2: Profiles of representative feedstock demand (in tons of fresh matter) for 1 kW installed electrical capacity (tFM kW_{el}^{-1})

Power Category [kW_{el}]	<150	150–500			500–1000	>1000
Associated sub-region[1]		HPL	LR	LMRF		
Animal slurry	43.4	22.9	54.6	77.9	43.8	5.9
Animal manure	2.8	3.3	1.9	0.6	1.0	6.4
Maize silage	6.43	6.72	6.78	2.03	5.31	14.81
Cereals[2]	2.95	1.76	0.88	0.57	1.84	0.85
Grass silage	-	3.30	1.27	3.29	1.36	0.23

[1]HPL=Heath & Pond Landscape; LR=Loess region; LMRF= Low Mountain Range & Foreland

[2]Cereals is a grouping referring to the following crops: Rye, Barely, Triticale

5.4.2 Indirect feedstock requirements

Beside direct area requirements for the production of energy crops, the use of animal excrement for biogas production implicates an indirect land use, in relation to the fodder crops used for livestock production (i.e. the original carbon sources for the animal excrement). We determined the livestock associated with a biogas plant from the relation between the required amount of excrement of the biogas plant ($excr_{bg}$ in t a^{-1}) and the available excrement within a BPU, assuming that this relationship describes the proportion of animals associated with biogas production (N_i):

$$N_i = n_i \cdot \frac{excr_{bg}}{excr_{av}} \quad (2)$$

The total fodder amount of type k (tfd_k in t a^{-1}) necessary to feed the animals associated with a biogas plant was calculated, based on the typical daily fodder demand of type k ($dfd_{i,k}$ in t d^{-1}) and the total number of animals associated with biogas production:

$$tfd_k = \sum_i \left(N_i \cdot 365 \cdot dfd_{i,k}\right) \quad (3)$$

The diet for dairy cows and cattle was assumed to be a silage mix from grass and maize of 25% and 75% respectively, with a cereals diet assumed for pigs (Table 5-3) (Gruber et al., 2006; Gruber et al., 2004). Additionally, it was assumed that only a basic diet is produced on the farm and concentrates were imported. Therefore, these were not considered for the calculation of the BFA (see section 5.5).

Table 5-3: Daily fodder demand of cows, cattle (elder than one year), brood sows and other pigs used for the calculation of indirect feedstock requirements. Calves and piglets are not considered. DM = dry matter; FM = fresh matter

		Dairy cows	Cattle	Brood sows	Other pigs
Total forage intake	[kg DM d^{-1}]	18,4	10,7	6,5	2
Basic diet		70%	70%	80%	80%
Maize silage	[kg FM d^{-1}]	36	21	-	-
Cereals	[kg FM d^{-1}]	-	-	6	1,8
Grass silage	[kg FM d^{-1}]	14	8	-	-

5.4.3 Biogas digestate

The amount of biogas digestate (*BGD* in t a^{-1}) produced and available for field application was estimated using equation (4).

$$BGD = \sum_x (FM_x - BG_x - L_x) \quad (4)$$

Where *FM* is the quantity of required substrate (t a^{-1}), *BG* is the amount of produced biogas (t a^{-1}), *L* is the amount of losses during the fermentation process (t a^{-1}) and *x* are the substrates listed in Table 5-4. According to Vogt (2008), the carbon flows in the biogas were assumed to consist of the sum of CH_4 and CO_2. The amount of biogas (t DM a^{-1}) was calculated with:

$$BG_x = FM_x \cdot DM_x \cdot oDM_x \cdot \frac{\alpha_x}{1000} \cdot \rho_x \tag{5}$$

Where *DM* is the substrate specific matter content (% FM), *oDM* is organic dry matter content (% DM), α is a substrate specific conversion factor for biogas (l kg^{-1} oDM^{-1}) and ρ is the substrate specific biogas density. Additionally, the nitrogen (N) content of the biogas was assumed to be insignificant. The carbon content of the biogas was determined from the share of CH_4 and CO_2 according to the specific substrate mix of the cluster.

Table 5-4 Substrate parameters used for biogas production calculations. DM = dry matter, oDM = organic dry matter content, biogas yield = substrate specific conversion factor for biogas (α), biogas density = substrate specific biogas density (ρ), losses = ensiling losses for silages (L_s), CH_4 = methane share in produced biogas.

Substrate	DM[1] [%]	oDM[1] [%]	C cont. [%]	N cont.[1] [%]	losses[3] [%]	CH_4 [3] [%]	Biogas yield[3] [l kg^{-1} oDM^{-1}]	Biogas density [kg m^{-3}]
Animal slurry	10	80	35[1]	4.67	0	55	380	1.28
Maize silage	28	95	45[2]	0.38	12	52	650	1.32
Cereals	86	97	45[2]	1.96	0	52	730	1.32
Grass silage	20	90	45[2]	0.38	12	53	600	1.31

[1] from CANDY database (Franko, 1996), [2] from (Schilling, 2000), [3] from (KTBL, 2012)

Losses during the fermentation process (*L*) were estimated using equation (6) and based on the assumption of 10% N losses during digestion (Vogt, 2008). *N* is the substrate specific N content (%).

$$L = \sum_x (FM_x \cdot DM_x \cdot N_x \cdot 0.1) \tag{6}$$

Consequently the N content of the biogas digestate (N_{BGD}) is also based on the N content of the biogas substrate and was estimated using the following equation:

$$N_{BGD} = \sum_x (FM_x \cdot DM_x \cdot N_x) - Lf \tag{7}$$

5.5 Estimation of the biogas fingerprint area

The BFA corresponds either to the area which is needed for the production of the biogas feedstock (A_{pr} in ha) or to the area needed for returning the digestate (A_{rc} in ha) when it exceeds the fertiliser demand of A_{pr}:

$$BFA = \max(A_{pr}, A_{rc}) \quad (8)$$

A_{pr} is calculated from the direct and indirect feedstock requirements of a biogas plant (see sections 5.4.1 and 5.4.2), considering typical ensiling losses Ls (Table 5-4) and the BPU specific yield Y (t ha^{-1}) of the relevant crops (x):

$$A_{pr} = \sum_{x} \left(\frac{(1 + Ls_x) \cdot FM_x}{Y_x} \right) \quad (9)$$

FM_x (in t) represents the feedstock requirement of energy crops or fodder crops. Grass silage demand is primarily provided by temporal grass crops and later by permanent grassland, if more substrate is required.

The area needed to recycle the digestate of a biogas plant (A_{rc} in ha) depends on the total N content of the digestate (N_{BGD} in t N) and application rates of N on arable land. We assume that the total amount of digestate-N applicable on A_{pr} (N_{pr} in t N) (1) compensates N offtake with harvested crops while (2) taking into account an application limit of 0.17 t N per ha given by legislation (DüV, 2017). If N_{BGD} exceeds N_{pr} the application area has to be extended by an additional area (A_{ex} in ha) for the disposal of the excess N (N_{ex} in t N):

$$A_{rc} = A_{pr} + A_{ex} \quad (10)$$

$$N_{ex} = N_{BGD} - N_{pr} \quad (11)$$

with:

$$N_{pr} = \min\left\{0.17 \cdot A_{pr}, \sum\nolimits_{x} (1 + Ls_x) \cdot FM_x \cdot Nc_x\right\} \quad (12)$$

where Nc_x is the N content in the fresh matter of the harvested yield of crop x.

If N_{BGD} is less than N_{pr} (N_{ex}<0), N_{BGD} will be evenly distributed on A_{pr}. If an additional area is required for digestate disposal (N_{ex} > 0), it is related to the average N removal by crop yield from the BPU area surrounding A_{pr} (N_{rem} in t N):

$$A_{ex} = \frac{N_{ex}}{N_{rem}} \quad (13)$$

BPUs where the local cultivation characteristics could not completely cover the feedstock demand of the corresponding biogas plants with respect to every type of substrate were excluded from the analysis. For example, some biogas plants at the Saxony border would require additional substrate from outside of the study region. This reduced the number of biogas plants included in the study from 183 to 121.

5.6 Carbon flows into soil

To characterize the impact of different land management systems on SOC we consider: (1) the total carbon flux from FOM into the soil as well as (2) the quality of different sources of FOM regarding the formation of new SOC. To assess the quality of the carbon flux from FOM to SOC, we use the “carbon reproduction flux” (C_{rep}), an indicator that aggregates the effect of different carbon sources on SOC storage (Brock et al., 2013; Franko et al., 2011; Kolbe, 2010; Küstermann et al., 2008).

The total carbon input from FOM, as well as the C_{rep} flux into soil were calculated in accordance with the approach of the carbon turnover models in CANDY (Franko et al., 1995) and CCB (Franko et al., 2011). In this approach the turnover of several FOM pools (C_{FOM}) results in a carbon flux to the atmosphere (mineralization) and a C_{rep} flux into the SOM pool. We calculated C_{FOM} and C_{rep} (in kg ha^{-1}) for different types of arable carbon sources: organic amendments (excrement, digestate), crop residues (roots and stubble) and crop by-products (straw and beat leaves) (Figure 5-3).

C_{FOM} flows were estimated using BPU specific yield data for each crop and application rates for organic amendments, as described in the previous sections. Parameterization of the different carbon sources and crops was taken from the CCB database. For the conventional agricultural carbon flows (residues, by-products, excrement) a more in-depth description is given by Franko et al. (2011). Regarding the matter flows from biogas digestate, equation (14) was used to calculate the carbon amount ($C_{FOM_{BDG}}$).

$$C_{FOM_{BDG}} = C_{FM} - C_{BG} \tag{14}$$

Here C_{BG} is the carbon equivalent of the produced biogas and C_{FM} is the total carbon amount of the biogas feedstock according to the material properties:

$$C_{FM} = \sum_{x} (FM_x \cdot DM_x \cdot C_x) \tag{15}$$

The carbon equivalent of the biogas C_{BG} was calculated using the molar volume of an ideal gas at 1 atmosphere of pressure V_m=22.42 l mol^{-1}, amount of biogas (BG_x), molar mass of carbon (M_C) in V_m depending on the methane share, biogas density ρ_x (kg m^{-3}) as sum over all added substrates x:

$$C_{BG} = \sum_x \left(\frac{BG_x * M_C}{\rho_x * V_m * 1000} \right) \tag{16}$$

For the calculations of C_{rep} every source of FOM has its specific substrate use efficiency parameter (η) characterizing the potential quality of the substrate for the formation of new SOC (Franko et al., 2011). The substrate use efficiency of biogas digestate was determined according to (Prays et al., 2017).

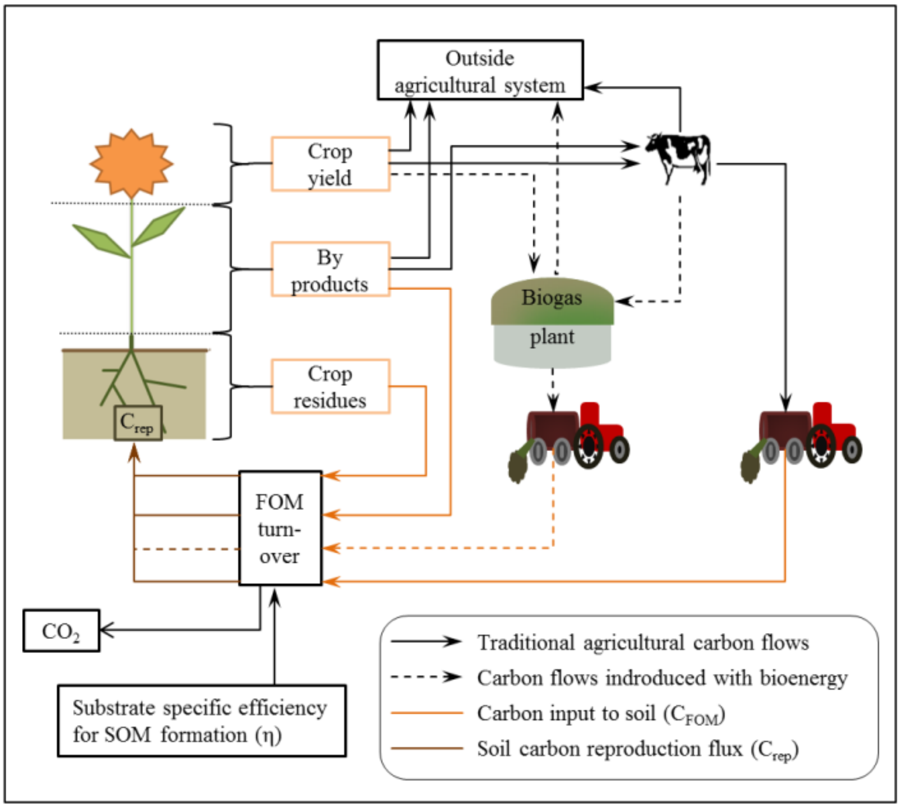

Figure 5-3: Carbon flows considered within the regional cycling of agricultural matter related to biogas production. Different pools of fresh organic matter (FOM) contribute to the total carbon flux to soil (C_{FOM}): crop residues, crop by-products, biogas digestate and livestock excrement. All sources of FOM have a different quality for the formation of new SOC. The C_{rep} flux is aggregating these differences and can be used as an indicator in a given environment to characterize the land use regarding SOC storage.

For the calculation of C_{FOM} and C_{rep} only arable land has been considered and permanent grassland has been left out. All carbon flows were calculated for two time steps, 2000 (without biogas) and 2011 (with biogas) for each BPU. For the year 2011, an additional analysis was performed for the BFA and for the area not affected by biogas production (see also section 5.3.1).

5.7 Results

5.7.1 Regional areal requirements of biogas production

5.7.1.1 BFAs and associated land use categories

The results of the model indicated that in 2011, the provision of biogas feedstock and distribution of digestate on average, affected 20.8% of the arable land within the BPUs. When considering the total agricultural land in Saxony (including permanent grassland) the BFA of the biogas plants covered 17.6% (Figure 5-4). Over 10% of all BPUs, were found to have a fingerprint area exceeding 40% of their BPU arable area.

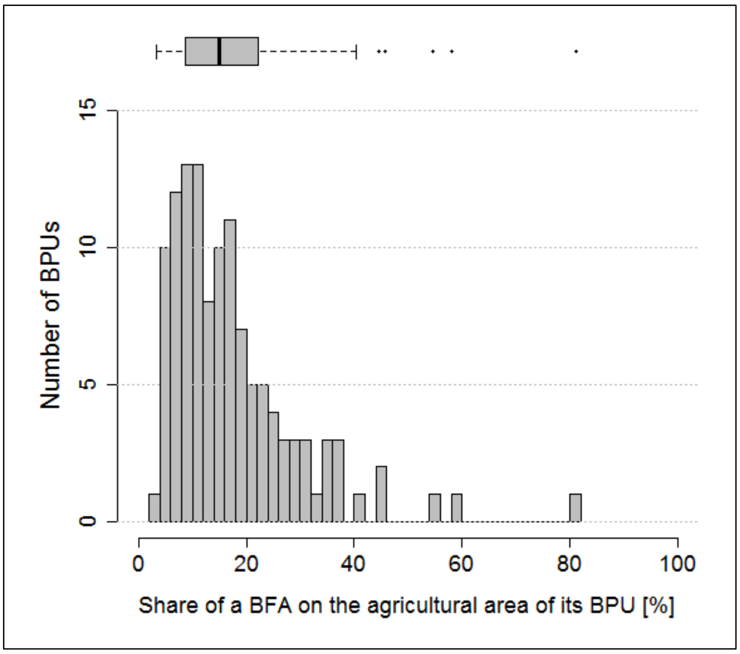

Figure 5-4 Share of the agricultural land of the BPUs in Saxony that is needed for the provision of biogas feedstock as well as for the distribution of digestate (BFA) in the year 2011.

The land use within BFAs was dominated by fodder crops on arable land (57.9%). The primary use of these areas is the production of meat and milk. The use of the livestock excrement for the production of biogas is a secondary and indirect use of these areas. The cultivation of energy crops on arable land covered 19.8% of the average BFA in Saxony and 7.1% was covered by permanent grassland. For most of the BFAs an additional area for the application of digestate was necessary. Digestate application to additional land outside the feedstock catchment accounted for 15.2% of an average BFA in Saxony.

5.7.1.2 Relationship between BFA and installed capacities

Relating the BFA to the installed electrical capacity of its biogas plant allows the different biogas systems to be compared with respect to the areal demand and hence areal efficiency per electrical energy output (ha kW_{el}^{-1}). On average for Saxony 2.0±0.4 ha kW_{el}^{-1} (± is the standard deviation) agricultural land was found to be influenced by biogas production. However, only 0.4±0.1 ha kW_{el}^{-1} from that was related to the cultivation of energy crops on arable land. The major part of the land demand consisted of fodder crops on arable land (1.2±0.3 ha kW_{el}^{-1}) for cattle supply, but also the additional area for digestate disposal was covering 0.3±0.3 ha kW_{el}^{-1}. To fulfill the demand for grass silage 0.1±0.1 ha kW_{el}^{-1} of permanent grassland was needed next to the use of field grass from arable land. Between individual BFAs the results differed due to regional differences in crop yields and livestock mix, as well as parameters of the specific biogas plant (e.g. installed capacity, feedstock mix).

The Saxon heath & pond landscape (1.8±0.3 ha kW_{el}^{-1}) as well as the loess region (1.9±0.2 ha kW_{el}^{-1}) showed significantly smaller area requirements than the low mountain range and foreland (2.6±0.4 ha kWel-1). Next to regional differences in crop yield this is a result of the greater number of smaller biogas plants in the low mountain range.

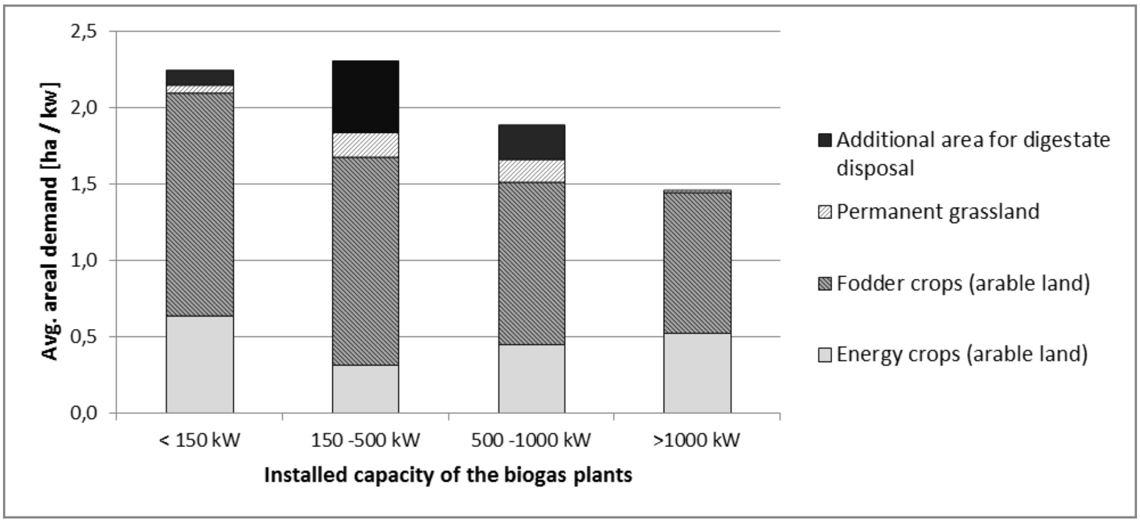

Figure 5-5: Average area demand (ha) per kilowatt installed electrical capacity of the biogas plants in Saxony. Biogas plants are differentiated by size classes. Area demand is separated by land use categories within a BFA.

Depending on the size classes of the biogas plants major differences in the total area demand and its composition have been observed (Figure 5-5). A constant decrease in the area requirements for the provision of biogas feedstock was found with increasing classes of plant size. While biogas plants with installed capacity < 150 kWel typically needed 2.1±0.3 ha kWel-1 for feedstock supply, plants > 1000 kWel only needed 1.5±0.1 ha kWel-1. This pattern was primarily caused by lower indirect feedstock requirements in the feedstock mix of larger biogas plants. But also the location distribution of the biogas plants and the subsequent agricultural yields are important factors. The area demand for the cultivation of direct feedstock requirements (energy crops) was lowest (0.3±0.1 ha kWel-1) for biogas plants in the size class 150-500 kWel. However, biogas plants in this capacity range showed the highest total areal demand per kWel due to large requirements regarding additional area for digestate disposal (0.5±0.3 ha kW-1). Input from energy crop cultivation was especially high within BPUs containing plants in the size classes <150 kWel and >1000 kWel. This was most of all due to a high share of energy crops (>1000 kWel) and especially cereals (<150 kWel) in the feedstock mix.

5.7.2 Regional carbon input to soil before and after implementation of biogas plants

The average carbon input into the arable soil of the Saxon BPUs was 2,905 kg C ha^{-1} in the year 2000 and increased slightly to 2,965 kg C ha^{-1} (+2.1%) in the year 2011, after the implementation of biogas plants. When considering the quality of different sources of FOM for the formation of SOC by using the indicator C_{rep} we observed an even higher increase of 2.8% (2000: 1,524 kg C_{rep} ha^{-1}; 2011: 1,567 kg C_{rep} ha^{-1}). Within the individual BPUs the changes between 2000 and 2011 are much more apparent, ranging from -388 kg C ha^{-1} to +576 Kg C ha^{-1} or -119 kg C_{rep} ha^{-1} to +297 kg C_{rep} ha^{-1} respectively.

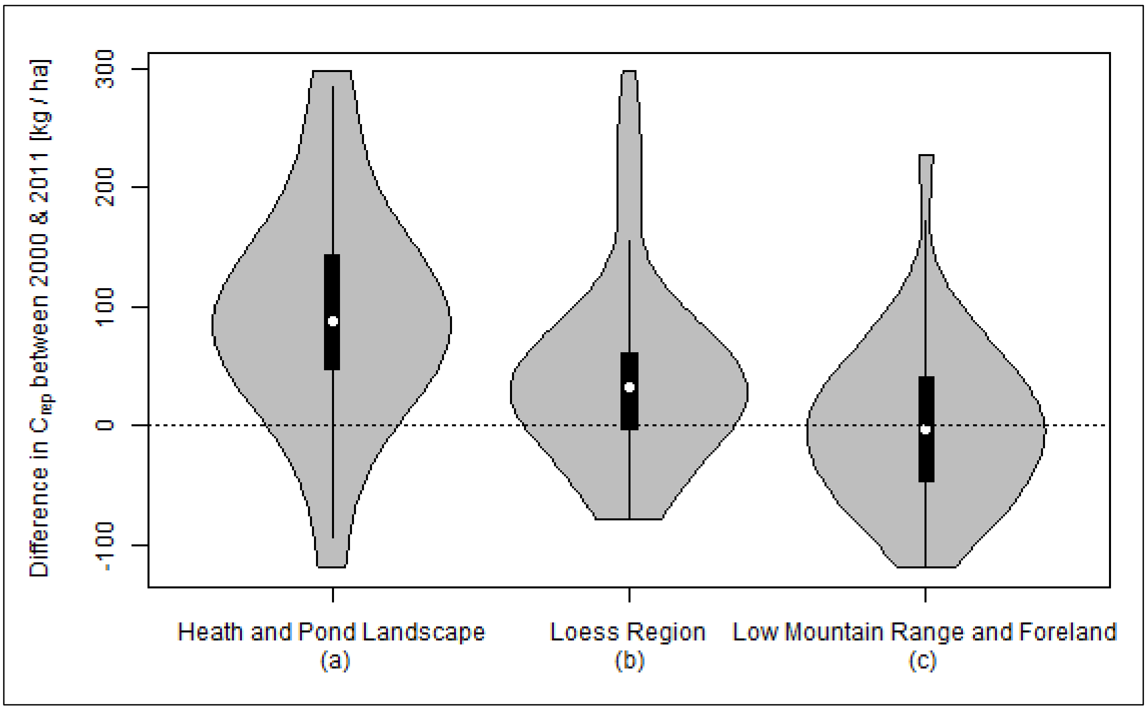

Figure 5-6: Violin plot showing the difference in C_{rep} between 2000 and 2011 on BPU-level, differentiated into the agro-economic regions within Saxony. The difference between the regions is significant (Welch t-test p-values: (a)-(b) 0.027, (a)-(c) <0.001, (b)-(c) 0.007). The Violin plot is combining a boxplot with a density plot.

The differences between individual BPUs were partly affected by their geographic location. On the level of agro-economic regions (Figure 5-5) significant differences in the temporal development of SOC input can be observed (ANOVA p-value: <0.001). Only limited statistical relationship between biogas plants capacity and SOC input have been found. BPUs having biogas plants in the power category 150-500 kW_{el} (+1.4% C_{rep}) contributed significantly less to the increase C_{rep} fluxes than the BPUs having biogas plants in all the other power categories (+4.0% C_{rep}) (Welch t-test p-value: 0.013).

5.7.3 Changes in carbon sources

5.7.3.1 SOC input from arable crops

The total crop based C flux into soil from all analyzed BPUs displayed a moderate increase (+5.8%) between 2000 (1,453 × 10^3 t C) and 2011 (1,538 × 10^3 t C). At the same time the contribution of the different cultivated crops changed greatly. Table 5-5 summarizes the quality adjusted C input (C_{rep}) from individual arable crops for the two time steps, 2000 and 2011. Winter rape, maize, winter wheat and sugar beet showed a high total increase in C_{rep}. A decline in C_{rep} contribution was observed for all cereals other than winter wheat. The contribution from fallow land was also seen to drop remarkably, as these areas went back into cultivation. The shift in C input to SOM of different crops is primarily caused by changes in cultivated area and less by changes in yield.

Table 5-5: Total soil carbon reproduction flux (C_{rep}) from the cultivation of different arable crops for the two years 2000 and 2011. C_{rep} is aggregating the carbon input to soil considering also the quality of different sources of FOM for the formation of SOC. Altering C_{rep} flows are caused by changes in the crop specific cultivated area and crop specific yields between the two time steps 2000 and 2011.

	Total C_{rep}		Differences between 2000 & 2011	
	2000 [10^3 t C]	2011 [10^3 t C]	Cultivated area [%]	Yield per area unit [%]
Winter Wheat	235.6	268.1 (+14%)	+13.8	+0.1
Winter Barley	121.0	103.4 (-15%)	-16.0	+2.4
Winter Rye & Triticale	98.1	62.4 (-36%)	-32.3	-4.4
Spring Cereals	49.3	45.5 (-8%)	-12.4	+7.0
Winter Rape	81.9	127.0 (+55%)	+43.7	+9.1
Maize for Silage	26.8	43.2 (+61%)	+61.4	-0.4
Grain Maize	12.2	21.7 (+78%)	+63.9	+12.2
Field Grass	21.7	26.4 (+22%)	+28.5	-9.9
Clover Grass	16.3	15.2 (-7%)	-6.2	-4.2
Sugar Beet	22.1	32.3 (+46%)	+23.4	+18.3
Other[1]	52.2	27.2 (-48%)	-32.2	-

[1] Fallow, vegetables, legumes, sunflower, potatoes, catch crops

5.7.3.2 SOC input from organic amendments

Our results indicate that major shifts in C flows on arable soils between 2000 and 2011 were associated with the type and contribution of organic fertilizers (i.e. animal excrement and biogas digestate). The total amount of regionally available C from livestock excrement declined from 295.8 × 10³ t in 2000 to 259.0 × 10³ t in 2011 (-12.4%) due to a reduction of livestock numbers. In the year 2000 all excrement were assumed to be applied to arable land, whereas in 2011 only 65% (167.2 × 10³ t) of the potential available C from livestock excrement could be used for this purpose. This was because the remaining part of livestock excrement (91.8 × 10³ t C) was used for the production of biogas. However, due to the usage of plant material (additional to the excrement) for biogas production, the C input to soil from biogas digestate (80.3 × 10³ t) compensates the livestock related C that was taken out of the traditional matter cycling. When considering the different quality of excrement and digestate for the formation of SOC the total contribution of organic amendments to C_{rep} fluxes decreased by only 5.1% in the period under study (2000: 180.4 × 10³ t C_{rep}; 2011: 171.1 × 10³ t C_{rep}) despite the reduction in livestock (-12.4%).

5.7.4 Carbon fluxes in- and outside of the BFA

Both C_{FOM} and C_{rep} were found to be lower on the arable land not needed for the provision of biogas feedstock and distribution of digestate (C_{FOM}: 2,956 kg ha^{-1}; C_{rep}: 1,518 kg ha^{-1}) than on the fingerprint areas of the biogas plants (C_{FOM}: 3,008 kg ha^{-1}; C_{rep}: 1,814 kg ha^{-1}). Indeed the C_{rep} fluxes were significant different (-16.3%; Welch t-test p-value: <0.001).

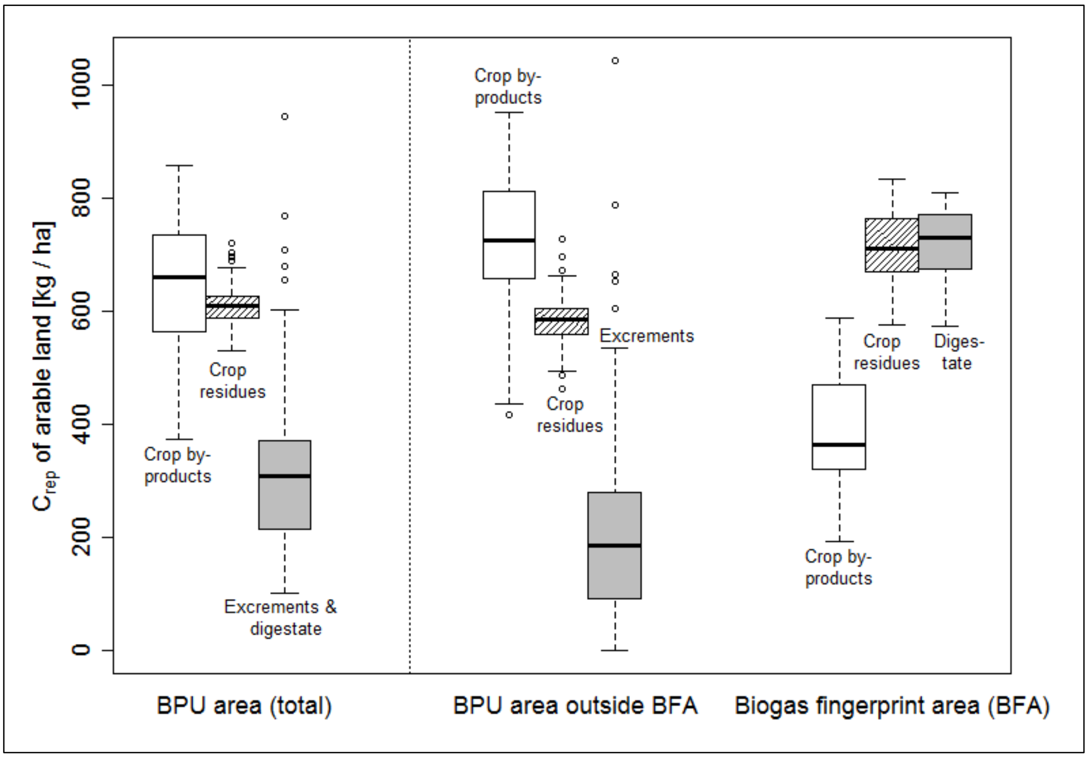

Figure 5-7: Soil-carbon reproduction fluxes (C_{rep}) of arable land for the year 2011 and with respect to different sources of carbon. C_{rep} aggregates the carbon input to soil considering also the quality of different sources of FOM for the formation of SOC. Regional basis are biomass providing units (BPU) as well as the two areal categories within a BPU: (1) biogas fingerprint area (BFA) and (2) BPU arable land outside of the BFA.

When analyzing the different carbon sources the BFAs showed a high carbon input to soil from crop residues and digestate application (Figure 5-7). The first is mainly due to a comparatively high share of agricultural grassland within the BFA which typically has higher amounts of residues (e.g. roots). The second is mainly due to the extensive application of digestate up to the limitation for organic N application. Furthermore, within the BFAs the amount of C from crop by-products (e.g. straw) is reduced, due to a lower share in cereal cultivation. In total the C_{rep} provision by arable crops (crop residues and crop by-products) is lower in the BFAs than in the surrounding BPU area.

5.8 Discussion

5.8.1 Influence of biogas production on land use

We developed a new approach of a 'biogas fingerprint area' to determine and characterize the agricultural areas affected by biogas production, due to their feedstock requirements and digestate recycling. This is in contrast to the concept of the 'ecological footprint' (Wackernagel and Rees, 1997). We deal only with the direct land area requirement for the production of biogas and disposal of digestate and within this, only the associated direct soil carbon fluxes. The BFA aggregates effects of location (e.g. crop yields) and management (e.g. feedstock mix of the biogas plant, fertilization practices). Therefore, the relationships between (1) the BFA and the total agricultural land of its BPU, as well as between (2) the BFA and the installed electrical capacity of its biogas plant are two valuable indicators for the analysis and differentiation of bioenergy production systems on larger scales. In this study the application of our methodology was successfully demonstrated for Saxony.

The need to establish a greater understanding of the relationship between power supply and area requirements of different renewable energy sources has already been identified (Evans et al., 2009; Lechon et al., 2011; Popp et al., 2014; Scheidel and Sorman, 2012; Wüstemann et al., 2017). Therefore, the results of this study contribute to a better understanding of this in relation to biogas production on the regional scale. We found that biogas production consumed the harvested crop yield from 4.1% of the BPUs arable land due to their direct feedstock requirements. This corresponds to on average 0.4 ha of energy crops from arable land per 1 kW_{el} installed capacity of an average biogas plant in Saxony. A similar range has been discussed in other studies analyzing the area demand of biogas plants in German study regions (Delzeit et al., 2011; Hartmann, 2008). But we also showed that the total area requirements of the biogas production systems in Saxony, including indirect feedstock requirements and the area needed for disposal of excess digestate, are many times larger than the area strictly dedicated to energy crop cultivation. Within the study period considered the harvest areas of the different cultivated crops changed considerably. While for the majority of crops, the increase in cultivated area may have been influenced by bioenergy production, the observed changes are also influenced by general changes in agricultural management (e.g. rotations).

Soil, climate and agricultural structure are important factors which distinguish the agro-economic regions in Saxony (StLa, 2004; StLa, 2012b) and effected the management (e.g. feedstock mix) and area requirements of the biogas plants. For example a large variability has been found with respect to the area needed for the disposal of digestate. But the regional properties (e.g. livestock numbers, yield potential) also affect the biogas plants themselves, e.g. with respect to the choice of power category that has been build. Other studies have shown, that there is an incentive to build larger biogas plants in areas having high yield expectations as this limits transportation distance and costs (Delzeit, 2009; Delzeit et al., 2012). It is important to understand these relations to be able to give scientifically substantiated recommendations on how to improve the management of those complex agricultural systems. The indicators developed in this study can help to identify critical hot-spots, where an increased competition for agricultural area and harvested crop yield may occur on one side between neighboring biogas plants, but also between biogas production and food production.

5.8.2 Biogas driven modifications in SOC input

The approach presented in this study can show the differences in carbon fluxes into soil between conventional agricultural systems and those with integrated biogas production. This is important, as the effects of bioenergy feedstock cultivation on SOC storage is a key factor in determining the sustainability of bioenergy (Anderson-Teixeira et al., 2009; Schrama et al., 2016; Tiemann and Grandy, 2015). Our results indicate that biogas production can be a win-win strategy that substitutes fossil fuel and leads to a positive effect on regional SOC input in Saxony. This also applies when considering the quality of the different sources of FOM for the formation of SOC.

The observed temporal shifts in carbon fluxes cannot be used to predict changes in long term SOC stocks, as they also depend on regional turnover conditions and the historical SOC development of the site. Large scale detailed monitoring data of SOC stocks in agricultural soils would allow quantifying actual changes in SOC storage. However this kind of monitoring for the whole Saxon study region has yet to be conducted. For future studies it may be an option to initialize regional SOC levels based on interpolation of available site measurements (Li, 2010; Mishra et al., 2010; Schloeder et al., 2001).

We propose to use the difference between the carbon input fluxes inside and outside of the BFA as indicator to characterize the sustainability of biogas production in terms of SOC storage. In Saxony average C fluxes to soil have been higher in the BFAs than in the arable land outside the BFAs. However, BFAs had a very low carbon input to soil from crop by-products (e.g. straw), due to a low share in cereal cultivation. The BFAs benefited from the extensive application of digestate, as well as from a high area share of agricultural grassland which typically has higher amounts of residues (e.g. roots). The effect of feedstock mix on the sustainability of biogas production has already been recognized by policy measures (EEG, 2017). Policy measures addressing the feedstock mix can effectively control the use of substrates and would affect the area of crop cultivation (Britz and Delzeit, 2013).

The calculated amounts of organic fertilizers applied on arable land (excrement and digestate) are about 3% lower than reported in official statistics on the application of organic amendments in Saxony (StLa, 2011) but are within a reasonable range (9.6×10^6 t compared to 9.9×10^6 t). For all BPUs analyzed, the carbon input from organic fertilizers changed considerably within the observation time. While the application of livestock excrement on arable land was strongly reduced due to the use for biogas production and the reduction in livestock numbers, the application of digestate could almost completely compensate this. Here the higher quality of digestate for the formation of SOC is important. The digestate based carbon is essential to compensate the low crop based carbon fluxes within the BFAs. Most of the biogas plants needed more area for the application of digestate than for feedstock supply. More practical field research is required to determine the effects of applying digestate, as of yet this knowledge base is sadly lagging behind what is known about application of animal slurries.

It must be pointed out that the analysis focused only on the biogas catchments and does not consider any indirect effects on SOC outside of Saxony due to imported fodder. But for this study these possible drawbacks are quite low as the rate of internal fodder production in Germany kept at about 90% between 2000 and 2011 (Deutscher Bundestag, 2012). Another uncertainty is the exact regional distribution of the livestock related organic fertilizers due to the spatial resolution of the initial data. However, the assumptions were consistent across the entire region and suitable for a relative comparison across the region.

5.8.3 Conclusions

The proposed modeling approach outlined in this paper has the benefit to provide better insight into agricultural carbon and matter fluxes, as well as regional area requirements related to biogas production. It is an attempt to understand the complexity of this system. It was shown that in the study region Saxony biogas plants can be operated sustainably with regard to SOC recycling. The total carbon flux into soil kept stable, with a slight tendency for an increase during the time period of the establishment of the biogas industry. On average, 17.6% of the agricultural land in Saxony was determined to supply the biogas plants and dispose of their digestate in 2011. The comparison of carbon fluxes inside and outside of this biogas fingerprint areas is an easily applicable instrument to assess the influence of biogas production on the region's SOC input.

Areas affected by biogas production showed a high carbon input to soil, but this was very reliant on the application of digestate. It could be beneficial if governments would develop "good farm practices" for agricultural systems operating biogas plants. Furthermore an adequate farm management planning has to be developed to deal with this different type of fertilizer.

5.9 Acknowledgments

We would like to thank the LfULG (Sächsisches Landesamt für Umwelt, Landwirtschaft und Geologie) and GALF (Gesellschaft für Angewandte Landschaftsforschung bR) for the provision and preprocessing of the data on crop harvest areas and yields within the framework of the research project "Dynamische Bilanzierung von Humushaushalt und Nährstoffaustrag im regionalen Maßstab im Kontext von Landnutzungs- und Klimawandel".

5.10 Supplementary material

Table A 1: Characterization of the agro-economic regions in Saxony. Average values for the specified periods.

	Heath & Pond Landscape	Loess Region	Low Mountain Range & Foreland
Temperature[1] [°C]	9,6	9,3	7,8
Precipitation[1] [mm]	736	770	961
Clay content[2] [%]	4,4	9,5	13,8
Silt content[2] [%]	22,8	65,3	58,2
Stone content[2] [%]	10,4	7,4	16,2
Arable land [%]	31,9	52,5	28,4
Grassland & pasture [%]	8,1	9,8	16,7
Catch crops[3] [%]	4,5	4,0	4,6

[1] Period 1990-2014, [2] of agricultural land (topsoil), [3] Period 2000-2012

6 Synthesis and conclusions

6.1 General discussion

6.1.1 Yield formation

The first objective of this thesis was to contribute to the knowledge of the yields and yield formation caused by fertilization with separated and unseparated BGRs from agricultural as well as industrial and municipal wastes.

To address this issue, a field experiment with soil samplings and laboratory analyses was conducted. The field experiment was an appropriate method to reach the objective, although only one vegetation period was observed. The main uncertainties of the method were the influence of the weather and water regime.

With a field experiment the first two hypotheses were confirmed, namely

1) separated BGRs sourced from biodegradable household and industrial wastes result in similar yields to those obtained by mineral fertilizers and that

2) the liquid fraction of BGRs results in higher yields than the solid fraction due to its high nitrogen (N) availability.

The results showed that the initial short-term fertilization effect of soil treated with BGRs from municipal and industrial wastes is similar to agricultural BGRs and mineral fertilizer. The yields measured in the experiment corresponded to average yields for the study region in 2011 (59 dt ha^{-1}). Fertilization with BGRs was based on TKN which is not completely directly available to plants. Nevertheless, no significant differences in the grain yields of plots fertilized with BGRs and mineral fertilizer were found. These results are in line with the majority of published investigations. Other authors also confirmed BGR's significance in improving crop yield (Bath and Ramert, 1999; Odlare et al., 2008; Rivard et al., 1995; Svensson et al., 2004) and grain quality (Svensson et al., 2004). A study by Rivard et al. (1995) showed that dried and composted BGR produced from municipal solid waste induced an increase in biomass (e.g. corn) and plant yield in direct proportion to the BGR application rate. Additionally, BGR was equally good or better than cow manure, pig slurry and mineral fertilizer in terms of fertilization of agricultural crops (Odlare et al., 2008). In general, the potential of biogas residue as a crop fertilizer and soil conditioner appears predominantly positive. However, the application of the BGRs to cropland requires rigorous monitoring to detect early perturbations in soil quality; potential physical, chemical and biological soil contaminations as well as atmospheric and water pollutions should be considered.

The second hypothesis assumes that the liquid fraction of BGRs results in higher yields comparing to solid fraction of the same BGR due to its high nitrogen availability. This assumption was confirmed by the results of the field experiment (chapter 2). A high fertilizing potential in liquid variants is associated with NH_4-N content which is immediately plant available after application. Nitrogen in solid variants has to be mineralized prior to being taken up by plants. In this regard, soil N_{min} in liquid variants was high at the beginning of the vegetation period and decreased until the end of the vegetation period due to plant uptake. Soil N in plots fertilized with solid variants was low throughout the observation period. The plant availability of N in these plots started only during the grain filling phase. As a result, grain N of solid variants was equal to or significantly higher than liquid variants.

Thus, the thesis shows that the liquid fraction of BGRs provides agricultural benefits and helps to close nutrient cycles. This is also confirmed by other current research results. Sigurnjak et al. (2017) state that the liquid fraction gives the same fertilization performance as mineral fertilizer. In those 3-year field trials, liquid fraction of BGR as a nitrogen source in treatments with animal manure or BGR had similar effects on biomass yields and soil properties as the conventional fertilization regime which uses animal manure and mineral nitrogen fertilizers.

The results of the field experiment show that BGRs affect the yield formation. Liquid fraction alone or in combination with nitrification inhibitor induced fewer plants per m² than corresponding solid and complete variants or variants without the inhibitor. One possible explanation is the high potential phytotoxicity of NH_3. Other authors also pointed to possible phytotoxicity and wrote that BGR may not be a suitable soil improver in its basic form, due to possible phytotoxicity (Alburquerque et al., 2012b; Schittenhelm and Menge-Hartmann, 2006; Teglia et al., 2011). Nevertheless, in the field experiment barley compensated the initial disadvantages over the vegetation period with a higher number of ears per plant and grains per ear. In other studies (e.g. on the farm scale) the phytotoxicity was not observed and also not reported by the farmer.

The last hypothesis concerns the yields increase due to the usage of nitrification inhibitor compared to the same BGR without inhibitor. This hypothesis had to be rejected. The assumption that the usage of nitrification inhibitor leads to higher grain yields compared to the same BGR without inhibitor was not confirmed. The yields on plots with nitrification inhibitor added to the BGR insignificantly differ from those fertilized with the same BGR without the inhibitor.

6.1.1 Soil organic carbon stocks

The next objective of this study was to determine the impact of the implementation of biogas production on the SOC and carbon fluxes in soil. The process model CANDY and the SOM model CCB were used as tools to reach this objective. Prior to modeling it was crucial to define the carbon turnover parameters of BGR degradability and their efficiency for creating new SOM. As a result, this thesis shows a possible way of parameter estimation using incubation experiments.

Additionally, a linear relationship between the turnover parameter k and the chemical property pH as well as between the synthesis coefficient η and C/N_{org} ratio of BGRs was found. After gaining this basic information about the BGR parameters for process modeling, the next step was to apply them within a model. It was assumed that due to carbon offtake during the anaerobic digestion process the replacement of undigested organic fertilizers with BGRs will lead to SOC decrease. The comparison of the organic matter parameters of different organic fertilizers could give the first information which fertilizer will have the most contribution to the SOM. Nevertheless, the model calculations were necessary to determine how much more or less SOC would result from BGR application compared to the application of the animal manures. The results in this thesis imply that the BGRs contribute more to the formation of SOM than cattle slurry. These preliminary results are in line with the current research, hence they were derived only from six BGRs. Heintze et al. (2017) showed that the application of cattle slurry results in significantly higher CO_2 mineralization fluxes compared to the application of BGR. Up to 60% less carbon remains in the BGR compared to the untreated waste and/or manure and/or slurry as a result of conversion to CH_4 during the degradation process (Ernst et al., 2008; Weiland, 2003). Hence easily degradable carbon compounds are decomposed in the digestion process and a more recalcitrant form of carbon is present in the BGR compared to untreated organic biomass (Ernst et al., 2008; Kirchmann and Witter, 1992).

In the next step these parameters were applied for modeling SOC development on the farm scale and for calculating C_{rep} on the landscape scale. The results show that biogas production has no negative impact on SOC development on the farm scale and within the regional study on the landscape scale. In case of the farm study no decrease of the SOC concentration was observed after 10 years of BGR application and no decrease until 2050 was predicted through the model. This is in agreement with Cong et al. (2017) who modeled the soil carbon stock and calculated carbon increase in soil in 100 years after BGR application. The authors found that the soil carbon stock could increase with low (52,310 t) / high (124,770 t) biomass supply potentials (measured as remaining carbon in soil in 100 years after digestate application into soil). Other authors also stated, that BGRs were effective in increasing SOM in short term respect to control, due to their high content in organic matter (Muscolo et al., 2017). However, the result in this thesis is restricted by the low number of studied fields and low number of SOC observations per field used for the validation. Thus, the results can be used as an indicator, but should be verified with further research.

In the regional study the results on the farm scale were confirmed. Values of C_{rep} and C fluxes were calculated. The inclusion of biogas production led to increase in the total carbon flux into the soil within the observation time and change of its contribution from different carbon sources. Areas affected by biogas production showed higher SOC reproduction rates than the surrounding agricultural land due to high contributions from digestate and crop residues. A 'biogas fingerprint' was calculated. It corresponds either to the area which is required for the production of the biogas feedstock or to the area required or returning the digestate when it exceeds the fertilizer demand of the biogas feedstock production area. Carbon reproduction proved to be higher within the biogas fingerprint area than outside the fingerprint area. Thus, the hypothesis which predicted decreasing carbon after the BGR application instead of undigested fertilizer was not confirmed.

A further hypothesis was that the biogas production would lead to a change of the cropping system with impact on carbon sequestration after its implementation.

The detailed studying of the farm management data as well as of the communal statistics data on the regional scale showed changes within the cropping system. On the farm level significant increase of crop yields (e.g. winter wheat and silage maize) as well as changes of cropping rotation were observed during 45 years of observation time. On the landscape scale considerable changes from 2000 to 2011 were observed. However, these changes in both studies are not necessarily driven only by the implementation of biogas plants. They are also caused by general developments in the agricultural management leading to increased yields or changes in crop shares. Hence the hypothesis about the cropping system changes cannot be conclusively confirmed.

The next objective of the thesis was to identify the share of agricultural land required for biogas production. This objective was only analyzed in the regional study. On the landscape scale the impact of each biogas plant was calculated in the form of areal demand. The hypothesis was that the specific agricultural area required for each biogas plant can be determined and characterized with respect to its spatial extent and the related soil carbon fluxes. Moreover, this area demand can be used as an indicator to assess the influence of biogas production on SOC reproduction. The system boundaries were chosen in a way which allows considering every biogas plant separately. The results show that approximately 17.6% of agricultural land are required for biogas production. This is also the approximate share which is used for the BGP supply on the studied farm.

6.2 Limitations of the thesis

Biogas residues vary greatly in their composition so that it is difficult to make overall and undifferentiated statements or classifications of BGRs and their effects on soil and crops. I used for experiments BGRs of different origin: untreated agricultural BGRs as well as treated BGRs from industrial and municipal wastes. Additionally, different methods were used: field experiments, laboratory analyses and process modeling. Thus the results of this thesis are subjected to different (conceptual and methodological) limitations.

The results on yield formation are supported by a study which was conducted during only one vegetation period. These first results are a good reference for short-term effects and can indicate the direction of further research. For more reliable data and better statistics it could be beneficial to conduct a field experiment over minimum two vegetation periods on different fields. For more comprehensive results depending on the issue, it could be even reasonable to conduct an experiment on different soils.

Furthermore, the field study was focused only on the treated (separated) BGRs from industrial and municipal wastes, but only one agricultural BGR was used. By contrast, the following modeling studies focused on the agricultural BGRs. The elemental composition of BGRs varies greatly from BGP to BGP and depends greatly on the feedstock material and the kind of fermentation procedure (Zirkler et al., 2014). Thus it could be helpful to complement the field experiment with few agricultural BGRs and the modeling study with some industrial and municipal BGRs to have a possibility to make more statistically reliable statements.

In view of the great variability of BGRs it is also necessary to determine specific SOM turnover parameters for each BGR by means of incubation experiments. However, analyses of every BGR applied to soils are expensive and time-consuming and cannot always be realized. These parameters are the decisive factor for the SOC predictions. Our modeling results on SOC development in soil affected by BGR are caused to a large extent by the results of the organic turnover parameters. This thesis provides the first approach to estimate BGR organic matter turnover parameters from simple and cost-efficient estimations. Nonetheless it is based on results from only six BGRs. Published results of the experiment, not specifically designed to parametrize that model, were used by reason of the lack of other possibilities. For example only total C was measured and thus inorganic origin of C evolution could not be excluded. In future studies the organic and inorganic C should be measured. Considering a bride variability of BGR feedstocks and consequently BGRs Statistical quality of these results is to be improved. Thus, further examinations (incubation experiments) are required to confirm or to refute the preliminary results about organic matter turnover parameters. Furthermore for better connecting of the modeled soils and BGRs on farm scale it could be worth to conduct incubation experiments with BGR and soils which are represented on the farm. Afterwards these recognitions can be better transferred to other regions, soils and BGRs.

The estimated parameters which were used for SOC modeling on the farm scale are only one factor of the limitations of the results. One another factor is the initial value for SOC in soil which was chosen for the model initialization. Model initialization was performed by adjusting the initial value of SOC manually during the spin-up run to fit the SOC values to measured values. It is a common procedure, but measured initial values would improve the accuracy and quality of the results. Additionally, the analysis is restricted by the low number of SOC observations per field used for the validation. More validation points would improve the results.

On the farm scale study only one farm with homogeneous soil conditions was modeled. Thus, the result can be a representative for Central Germany, were loess soils are widespread, but not e.g. for sandy soils which are less beneficial for agriculture and which probably react more sensitively on changes of organic matter supply. Apart from that, only SOC was measured in the field. Other soil properties for modeling were inferred from a soil map or calculated from these inferred properties. These uncertainties can influence the modeling result, but they were not quantified within the study.

The investigations on the C fluxes and the areal demand of the BGPs in Saxony on the landscape scale are limited to the data quality and system boundaries. The smaller the resolution of the data about soils, BGPs, weather conditions, applied fertilizers and crop rotation, the exacter are the modeling results. One uncertainty in this thesis is that due to the spatial resolution of the initial data it was not possible to distribute the livestock-related organic fertilizers in the region exactly. As it was the first attempt to quantify C fluxes which are affected by a BGP, it was necessary to keep the system as simple as possible and regard closed matter cycles. The boundary conditions were chosen in a way that all feedstock is produced on farm. Thus any indirect effects on SOC outside of Saxony due to imported fodder were not considered and the analysis focused only on the biogas catchments. Other uncertainties are the substrate mix for each BGP and the fodder ration of animals. These parameters influence substantially the area demand of each BGP. However, the assumptions are suitable for a relative comparison across the region whereas they were consistent across the entire region.

6.3 Further research

A number of objectives were reached in this thesis. First, the stated hypotheses have been verified. In general, the potential of BGRs as a crop fertilizer and soil conditioner appears predominantly positive. Additionally, this thesis provides preliminary results where no negative effects of BGRs on the SOC were observed or predicted. The areal demand of the BGPs in Saxony does not exceed the available area for the biogas production. Nevertheless, there are limitations of the thesis that may be a subject of further research.

Different types of organic wastes will still be important for the biogas production. Thus, the future research on the impact of BGRs on yield formation and yields should equally include both types of BGRs from agricultural as well as from industrial and municipal wastes. It can be substantial to use field experiments for reliable SOC measurements, which can be used for modeling. Experiments on longer term (> 3 years) may be required to fully evaluate the effects of continuous application of separated BGRs on crop growth and soil fertility.

The variability of the BGR properties is still a challenge for future predictions of biogas production effects on soils. Further examinations in the form of incubation experiments are required to collect more reliable information about the chemical BGR characteristics and the SOM turnover parameters. Furthermore more systematical research of the feedstock influence on BGR properties is required.

In this thesis, CANDY process model, presuming SOM subdivision into pools, was used. It is assumed that after FOM is applied on soil, SOM is build up during ‘humification’ process when carbon moves between the pools. In literature, Lehmann and Kleber (2015) criticize that such process models does not represent the characteristic processes of carbon transformation, such as adsorption and physical protection, desorption, and microbial activity. In the research aimed at reliable predictions of SOM turnover, they advise to focus on investigating its spatial arrangement within the mineral matrix, the fine-scale redox environment, microbial ecology and interaction with mineral surfaces under moisture and temperature conditions observed in soils. Thus, a comparison of different SOM turnover models results could bring a light in the discussion and could be beneficial for overall process understanding.

On the landscape studies further studies have to show if the relations observed in Saxony are valid for other regions as well. Therefore the system boundary conditions concerning the substrate mix, the fodder ration and the regional organic fertilizer distribution could be refined.

Further new studies can also deal with the improvement of the SOC data for model applications, including N investigations, to have a possibility to make significant statements about the environmental effects of BGRs. One of the biggest present problems of the BGRs is their potential environmental effect on the ground water. Biogas residues came into the focus of public attention through negative headlines because of their impact on the ground water (BMUB and BMEL, 2017). Only 50 mg/l of nitrate are allowed according to the German drinking water regulations (TrinkWV, 2001). But in areas with widespread organic fertilization (including BGR applications) in the agriculture, values over 50 mg/l are not rare. Additionally, one of the problems for the farmers and also for BGP operators lies in the efficient use of the produced BGRs, especially with regard to the new fertilization ordinance regulation (DüV, 2017). Maximal amounts and times of BGR application were restricted, so that a better planning of the fertilization would become more important.

Furthermore, new studies can help to develop “good farm practices” for the agricultural systems operating biogas plants. These practices could help to deal with the BGRs as fertilizer and to plan an adequate farm management.

7 Danksagung

Es ist eine Freude all denen zu danken, die mich auf verschiedenste Weise unterstützt und mich bis zur Fertigstellung der Dissertation begleitet haben.

Mir liegt es sehr am Herzen, zu Beginn meinen beiden Betreuern, Prof. Martin Kaupenjohann und Dr. Uwe Franko, meinen besonderen Dank auszusprechen. Ich möchte mich einerseits für die wunderbare, fachliche Betreuung bedanken, die durch anregende und konstruktive Diskussionen geprägt war und die mir des Öfteren half, meine Gedanken zu ordnen bzw. Sachverhalte neu zu denken. Neben dieser kompetenten, fachlichen Unterstützung barg meine Betreuung andererseits auch eine sehr menschliche und kollegiale Komponente, die mich als Persönlichkeit wachsen ließ. Ich bedanke mich für all das Vertrauen, die Zuversicht und die offenen Ohren, die mir entgegengebracht wurden. Selbst in Phasen, in denen mir das Promovieren schwer fiel, konnte ich auf den Zuspruch meiner beiden Betreuer zählen. Vielen Dank für die Motivation und den Glauben, die mich in Tiefphasen gerettet haben. Ich habe viel von Euch beiden gelernt – Dankeschön!

Zudem möchte ich an dieser Stelle auch Herrn Prof. Christof Engels danken, der sich bereiterklärt hat, der dritte Gutachter meiner Arbeit zu sein.

Mein nächster Dank geht an meine wunderbaren Kollegen von der Arbeitsgruppe C und N Dynamik: Dr. Ralf Gründling, Eric Bönecke, Katharina Meurer, Felix Witting, Petra Peterson und Sylvia Schmögner. Sie haben mich immer mit positiven Emotionen versorgt und mich tatkräftig im Büro, im Labor sowie im Feld unterstützt und stets Lösungen für größere und kleinere Probleme gefunden. Die Montagsbesprechungen, die Kuchenrunden, die CANDY-Sitzungen, die Kneipengänge – unsere kleine CANDY-Welt wird mir für immer warm in der Erinnerung bleiben. Außerdem vielen Dank an Martina Stange für die unvergessliche Unterstützung im Labor. Meiner studentischen Hilfskraft Raik Petermann möchte ich danken, dass er während meiner Schwangerschaft im Feld sowie im Labor stets ohne Widerstand alle meine Arbeitsaufträge erfüllte und zusätzlich mich mit Wild versorgte.

Ein besonderer Dank gebührt meiner wundervollen Bürokollegin Anika Klotzbücher für die schönsten Bürostunden. Wir haben viel gelacht, viel geleistet und sind durch dick und dünn gegangen. Vielen Dank an Pauline Winkler, die mich auf letzten Metern immer dienstags und sonntags in die Bibliothek physisch und mental begleitet und mit mir „mitgelitten“ hat. Meiner Freundin Alena Roos danke ich dafür, dass sie mir beibrachte, dass die Sprache Kunst ist. Und dafür, dass sie sich in jede meiner Lagen einfühlen konnte, mich bei allen meinen emotionalen Stürmen wieder zu beruhigen wusste und stets eine Lösung (mit Zucker und /

oder Worten) parat hatte. Ebenfalls ein großes Dankeschön an Dasha Frankiv für die sehr hilfreicheTextkorrektur und emotionale Unterstützung.

Vielen Dank ebenfalls an meine neuen Kollegen, insbesondere Dr. Hans-Joachim Berger und Eyk Hasselwander, die die berufsbegleitende Fertigstellung meiner Dissertation gewünscht und vorangetrieben haben. Insgesamt habe ich in der G.U.T. mbH mehr als nur gute Kollegen gefunden. Ich freue mich auf meinem weiteren Weg solche wunderbaren Menschen um mich zu haben.

Für die finanzielle Unterstützung möchte ich mich bei dem Impuls- und Vernetzungsfonds der Helmholtz-Gesellschaft; bei dem Department Bioenergie, dem Department Bodensystemforschung sowie der Graduiertenschule HIGRADE am Helmholtz-Zentrum für Umweltforschung GmbH – UFZ bedanken.

Ein großer Dank geht an meine Eltern, die mir liebend und beratend zur Seite standen, die sich viele Sorgen um meine Zukunft machten, aber mich immer meine Entscheidungen machen ließen.

Und schließlich möchte ich mich bei meinem Mann Aleksey Prays und meinen Kindern Luka und Jakob für ihre Liebe, Verständnis, Geduld und stetige Ermunterung danken. Ohne Eure Rückenstärkung, ständiges „unter die Arme greifen“ im Familienalltag sowie das Abpuffern aller emotionalen Hochs und Tiefs würde ich die Promotion nie realisieren können. Vielen Dank, dass ich immer die Möglichkeit und Freiheit habe den Weg zu gehen, den ich gehen will und die Dinge tun, die ich tun möchte!

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9 Liste der Veröffentlichungen

Folgende Veröffentlichungen (Paper) sind in meiner Dissertation enthalten:

1. Prays, N., and Kaupenjohann, M. (2016): Initial Effects of Differently Treated Biogas Residues from Municipal and industrial Wastes on Spring Barley Yield Formation. PLoS ONE 11, https://doi.org/10.1371/journal.pone.0154232 (postprint)

2. Witing, F., Prays, N., O'Keeffe, S., Gründling, R., Gebel, M., Kurzer, H.-J., Daniel-Gromke, J., Franko, U., 2018. Biogas production and changes in soil carbon input - A regional analysis. Geoderma 320, 105-114, https://doi.org/10.1016/j.geoderma.2018.01.030 (postprint)