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List of Acronyms

a	annum (lat. year)
ADP	abiotic depletion potential
ALO	agricultural land occupation
AP	acidification potential
C	carbon
CAN	calcium ammonium nitrate
CC	climate change
Cd	cadmium
CH_4	methane
CHP	combined heat and power unit
Cl	chlorine
CO_2	carbon dioxide
Cr	chromium
Cu	copper
DM	dry matter
DTT	distance-to-target
EEG	Renewable Energy Act
EMI	European Miscanthus Improvement
EoL	End-of-Life
EP	eutrophication potential
EU	European Union
FAETP	freshwater aquatic ecotoxicity potential
FE	freshwater eutrophication
FET	freshwater ecotoxicity
FFD	fossil fuel depletion
FU	functional unit
GHG	greenhouse gas
GW	gigawatt
GWP	global warming potential

ha	hectare
HCL	hydrogen chloride
HHV	higher heating value
HT	human toxicity
ILCD	International Reference Life Cycle Data System
iLUC	indirect land-use change
IPCC	Intergovernmental Panel on Climate Change
IR	ionizing radiation
IRENA	International Renewable Energy Agency
ISO	International Organization for Standardization
K	potassium
K_2O	potassium oxide
KTBL	Kuratorium für Technik und Bauwesen in der Landwirtschaft e.V.
kWh_{el}	kilowatt hour of electricity
LCA	Life-Cycle Assessment
LCI	Life-Cycle Inventory
LCIA	Life-Cycle Impact Assessment
LSF	Life Support Function
LUC	land-use change
ME	marine eutrophication
MET	marine ecotoxicity
Mg	megagram
MRD	mineral resource depletion
N	nitrogen
NF	normalisation factor
NH_3	ammonia
NLT	natural land transformation
N_2O	nitrous oxide
NO	nitric oxide
NO_3	nitrate
OD	ozone depletion
ODP	ozone depletion potential

OECD	Organisation for Economic Co-operation and Development
OPTIMISC	Optimizing Miscanthus Biomass Production
ORC	organic Rankine cycle
P	phosphor
Pb	lead
P_2O_5	phosphorus pentoxide
POCP	photochemical ozone creation potential
POF	photochemical oxidant formation
PMF	particulate matter formation
ppm	parts per million
S	sulphur
SO_2	sulphur dioxide
SOC	soil organic carbon
SOM	soil organic matter
SPML	species-poor marginal land
SRML	species-rich marginal land
TA	terrestrial acidification
TET	terrestrial ecotoxicity
TSP	triple superphosphate
ULO	urban land occupation
VDI	Verein Deutscher Ingenieure
VDLUFA	Verband Deutscher Landwirtschaftlicher Untersuchungs- und Forschungsanstalten
yr	year
WD	water depletion

Abstract

In a developing bioeconomy, the demand for biomass for industrial purposes is expected to increase significantly. This demand needs to be met in a sustainable way and without compromising food security. With this goal in mind, resource-efficient lignocellulosic crops, such as perennial energy grasses, are often cited as a biomass source with low negative impacts on the environment. Under European conditions, miscanthus is the leading perennial energy grass because of its high biomass and energy yield potential. It is a C4 plant, which achieves dry matter biomass yields of up to 20 Mg ha^{-1} yr^{-1} when harvested in later winter, and up to 30 Mg ha^{-1} yr^{-1} when harvested green in October. Currently the main utilization route of miscanthus is direct combustion for heat generation, but the biomass can also be used for various other applications, such as biofuels and insulation material. Several studies have analysed the environmental performance of perennial crop-based value chains, but most of these only assessed the Global Warming Potential (GWP). However, the GWP alone is not an adequate indicator for the holistic assessment of the environmental performance of such value chains. In addition, these studies often used generic data and applied varying assumptions, which makes a comparison of different value chains difficult.

The main goal of this thesis is to draw up recommendations for future assessments of the environmental performance of perennial crop-based value chains. For this purpose, five research objectives were formulated: 1) to identify the key parameters influencing the environmental performance of perennial crop-based value chains; 2) to analyse which impact categories are most relevant when assessing the environmental performance; 3) to assess the differences between various perennial-crop based value chains; 4) to assess the environmental performance of the utilization of marginal land to grow perennial crops for industrial purposes; and 5) to analyse and compare the environmental performance of annual and perennial crops in the example value chain ‘biogas production’.

To achieve these research objectives, the environmental performance of several perennial crop-based value chains was analysed in various impact categories applying the same underlying assumptions and using field data obtained under *ceteris paribus* conditions. The analysis was carried out using the globally recognised Life Cycle Assessment (LCA) methodology, which is standardized by two ISO norms (14040/44).

The results revealed that biomass yield is one of the most important parameters influencing the environmental performance of perennial crop-based value chains. An increase in yield of 50%, for instance, leads to an increase in carbon mitigation potential in a comparable range (46%). Furthermore, the marked influence on the environmental impact mitigation potential of both fertilizer-induced emissions and selection of the reference system was demonstrated. For example, if the reference system is changed from light fuel oil to natural gas, the substituting by heat generated from the combustion of miscanthus biomass increases the net impact in the category ‘particulate matter formation’ by 220%. The relevance of different impact categories was analysed for various perennial crop-based value chains using a normalisation approach. The results clearly indicated that a holistic assessment of the environmental performance of perennial crop-based value chains should at least include the impact categories ‘marine ecotoxicity’, ‘human toxicity’, ‘agricultural land occupation’, ‘freshwater eutrophication’ and ‘freshwater ecotoxicity’. In future assessments, it is recommended to include the impacts of land-use on both biodiversity (using species richness as an indicator) and soil quality (using SOM as an indicator). The comparison of the environmental performance of different perennial crop-based value chains revealed clear environmental advantages of the cascade use of biomass. An example is the production of miscanthus-based insulation material, which is first used as a building material and then incinerated to generate heat and electricity. The results also demonstrate that, despite low biomass yield on marginal land, miscanthus-based value chains have a substantial environmental impact mitigation potential when substituting a fossil-based reference system. Furthermore, the comparison of annual and perennials crops as biogas substrates showed that perennial crops, and in particular miscanthus, have a considerably better environmental performance in the impact categories ‘climate change’ (up to -73%), ‘fossil fuel depletion’ (up to -79%), ‘freshwater eutrophication’ (up to -69%), ‘marine eutrophication’ (up to -67%), and ‘terrestrial acidification’ (up to -26%).

In all four studies included in this thesis, it was observed that the data used for the biomass cultivation in particular, such as yield and fertilizer-induced emissions, have a considerable influence on the environmental performance. This data is highly site- and crop-specific and is strongly dependent on the agricultural management system applied. Based on the results of this thesis, the common practice of using generic data in assessments of the environmental performance of perennial crop-based value chains should be rejected. In order to obtain realistic results, the use of site- and crop-specific data is highly recommended.

Zusammenfassung

In einer wachsenden Bioökonomie steigt die Nachfrage nach Biomasse für industrielle Zwecke deutlich an. Diese Biomasse sollte nachhaltig produziert werden und dabei die Ernährungssicherheit nicht gefährden. Ressourceneffiziente, lignocellulosehaltige Pflanzen, wie beispielsweise mehrjährige Energiegräser, werden oft als Biomassequelle mit geringen negativen Auswirkungen auf die Umwelt angesehen. Unter europäischen Bedingungen ist Miscanthus aufgrund seines hohen Biomasse- und Energieertragspotentials das am weitesten verbreitete mehrjährige Energiegras. Miscanthus ist eine C4-Pflanze, die Trockenmasseerträge von bis zu 30 t ha^{-1} a^{-1} erreicht wenn ein Grünschnitt im Oktober erfolgt, und bis zu 20 t ha^{-1} a^{-1}, wenn Ende des Winters im März geerntet wird. Derzeit wird die Miscanthusbiomasse hauptsächlich zur Wärmeerzeugung genutzt, aber auch andere Verwertungsrichtungen sind möglich, wie zum Beispiel die Produktion von Biokraftstoffen oder die Herstellung von Dämmstoffen. In verschiedenen Studien wurde die Umweltwirkung von Wertschöpfungsketten, die auf mehrjährigen Pflanzen basieren, analysiert. Dabei wurde meist nur das Treibhauspotential betrachtet. Allerdings ist das Treibhauspotential allein kein hinreichender Indikator für eine ganzheitliche Bewertung der Umweltwirkung solcher Wertschöpfungsketten. Des Weiteren wurden in diesen Studien häufig Literaturdaten verwendet und sie stützen sich auf unterschiedliche Annahmen, beispielsweise in Bezug auf die Berechnung der Nitratauswaschung, was einen Vergleich verschiedener Wertschöpfungsketten erschwert.

Ziel der vorliegenden Thesis war es, Empfehlungen für zukünftige Studien auszuarbeiten, die die Umweltwirkung von auf mehrjährigen Pflanzen basierenden Wertschöpfungsketten analysieren. Hierfür wurden fünf Forschungsziele formuliert: 1) Ermittlung der wichtigsten Parameter, die die Umweltwirkung von mehrjährigen Kulturpflanzen beeinflussen; 2) Analyse der Relevanz verschiedener Wirkungskategorien für die Bewertung dieser Umweltwirkung; 3) Erfassung und Bewertung der Unterschiede zwischen verschiedenen auf mehrjährigen Pflanzen basierenden Wertschöpfungsketten; 4) Abschätzung der Umweltwirkung der Nutzung von Grenzertragsstandorten zum Anbau von mehrjährigen Biomassepflanzen für industrielle Zwecke; und 5) Vergleich der Umweltwirkung von einjährigen und mehrjährigen Kulturen im Rahmen der Biogasproduktion.

In dieser Thesis wurde die Umweltwirkung mehrerer auf mehrjährigen Pflanzen basierenden Wertschöpfungsketten in verschiedenen Wirkungskategorien analysiert. Die Abschätzung der

Umweltwirkung erfolgte mit Hilfe der weltweit anerkannten und durch zwei ISO-Normen (14040/44) standardisierten Life-Cycle Assessment (LCA) Methodik. Dabei wurden dieselben zugrunde liegenden Annahmen angewandt und unter *ceteris paribus* Bedingungen ermittelte Felddaten verwendet.

Die Ergebnisse dieser Studien zeigten, dass der Biomasseertrag einer der wichtigsten Parameter ist, der die Umweltwirkung von auf mehrjährigen Pflanzen basierenden Wertschöpfungsketten beeinflusst. Wenn der Ertrag beispielweise bei gleichbleibendem Ressourcenaufwand um 50 % gesteigert werden kann, dann steigt das CO_2-Minderungspotential um 46 % an. Darüber hinaus haben düngerbedingte Flächenemissionen und die Auswahl des fossilen Referenzsystems (d.h. das Produkt, welches in der Praxis durch die Miscanthus-basierte Wertschöpfungskette ersetzt wird) einen großen Einfluss auf die Umweltwirkung der jeweiligen Wertschöpfungskette. Dies wird im Folgenden am Beispiel von Wärme dargestellt, die durch die Verbrennung von Miscanthusbiomasse erzeugt wurde. Wird Erdgas anstelle von leichtem Heizöl als Referenzprodukt verwendet, erhöht dies die Umweltwirkung in der Wirkungskategorie ‘Bildung von Feinstaubpartikeln’ um 220 %. Dies liegt in der unterschiedlichen Menge an Feinstaubpartikeln begründet, die bei der Verbrennung dieser Energieträger gebildet werden.

Die Relevanz verschiedener Wirkungskategorien wurde anhand eines Normalisierungsansatzes für verschiedene auf mehrjährigen Pflanzen basierende Wertschöpfungsketten bestimmt. Die Ergebnisse zeigten eindeutig, dass für eine ganzheitliche Bewertung der Umweltwirkung dieser Wertschöpfungsketten zumindest die Wirkungskategorien ‘aquatische Ökotoxizität’ (bezogen sowohl auf Salzwasser als auch Binnengewässer), ‘Humantoxizität’, ‘Okkupierung landwirtschaftlicher Flächen’ und ‘Eutrophierung von Binnengewässern‘ analysiert werden müssen. Darüber hinaus sollten in künftigen Untersuchungen die Auswirkungen der Landnutzung auf die Biodiversität und die Bodenqualität miteinbezogen werden. Im Fall der Biodiversität empfiehlt sich dabei die Nutzung des Artenreichtums als Indikator. Um den Einfluss der Landnutzung auf die Bodenqualität abzuschätzen, stellt der Gehalt an organischer Substanz im Boden einen geeigneten Indikator dar. Der Vergleich der Umweltwirkung verschiedener auf mehrjährigen Pflanzen basierenden Wertschöpfungsketten zeigte deutlich die Vorteile der Kaskadennutzung von Biomasse. Ein Beispiel hierfür ist die Herstellung von Dämmmaterial aus Miscanthusbiomasse. Dieses wird zuerst als Baustoff verwendet und kann nach seiner Nutzenphase zur Erzeugung von Wärme und Strom verbrannt werden.

Im Rahmen der hier vorliegenden Thesis wurde auch die Umweltwirkung von Wertschöpfungsketten untersucht, die Miscanthusbiomasse, welche auf marginalem Land angebaut wurde, nutzen. Auf diesen Grenzertragsstandorten sind oft wesentlich niedrigere Biomasseerträge zu erreichen, im Vergleich zu fruchtbarem landwirtschaftlich genutztem Land. Trotzdem zeigten die Analysen dieser Wertschöpfungsketten ein erhebliches Potential Umweltwirkungen zu mitigieren, durch die Substitution eines auf fossilen Rohstoffen basierenden Referenzsystems. Im Vergleich zu einjährigen Pflanzen wiesen die mehrjährigen Kulturen als Substrat für die Biogasproduktion deutlich niedrigere Umweltwirkungen auf. Wenn Biogasmais durch Miscanthus ersetzt wurde, konnten das ‘Treibhauspotential’ um bis zu 73 %, der ‘Verbrauch fossiler Brennstoffe’ um bis zu 79 %, die ‘Eutrophierung von Binnengewässern’ um bis 69 %, die ‘Eutrophierung der Meere’ um bis zu 67 % und die ‘terrestrische Versauerung’ um bis zu 26 % reduziert werden

Alle in dieser Thesis enthaltenen Studien zeigten deutlich, dass insbesondere die für den Biomasseanbau benötigten Daten einen erheblichen Einfluss auf die Umweltwirkung von auf mehrjährigen Pflanzen basierenden Wertschöpfungsketten haben. Beispiele hierfür sind der Biomasseertrag und die Flächenemissionen, welche durch den Einsatz von Phosphor- und Stickstoffdüngern entstehen. Diese Daten sind sowohl pflanzen- als auch standortspezifisch und hängen stark von der jeweiligen landwirtschaftlichen Bewirtschaftung ab. Ausgehend von den Ergebnissen dieser Studien ist von der derzeit gängigen Praxis abzuraten, generische Daten für die Bewertung der Umweltwirkungen von solchen Wertschöpfungsketten zu verwenden. Um realistischere Ergebnisse zu erzielen wird ausdrücklich empfohlen auf standort- und pflanzenartspezifische Daten zurück zugreifen.

1. General Introduction

1.1. Driving forces of change towards a bioeconomy

The world in the 21st century is facing multiple environment-related challenges. Research based on the planetary boundaries framework suggests that, of the ten boundaries proposed, three have already been transcended. The planetary boundaries represent the range in which mankind can operate safely, without exceeding certain thresholds, which could lead to abrupt non-linear changes in the environment. One example of a boundary that has already been exceeded is climate change. Here, the threshold has been defined as an atmospheric CO_2 concentration of above 350 ppm (parts per million) (Rockstrom *et al.* 2009). Climate change is, to a large extent, induced by fossil CO_2 emissions from industrial processes, in particular the combustion of fossil fuels (IPCC 2014). In order to overcome these challenges and mitigate climate change, a comprehensive approach is required, which includes a shift towards low-carbon renewable energy sources, such as wind energy and biofuels, and towards renewable carbon sources in the form of biomass. The ongoing depletion of fossil resources reinforces this necessity. Fossil fuels reserves, with the exception of coal, could be depleted by the year 2042 (Shafiee & Topal 2009).

To tackle these problems and foster low-carbon growth, several countries (e.g. USA, Germany), the EU as a whole, and organisations such as the OECD, have proposed a transition from the current fossil-based economy towards a biomass-based bioeconomy (Staffas *et al.* 2013). McCormick & Kautto (2013) define the bioeconomy as an economy "*where the basic building blocks for materials, chemicals and energy are derived from renewable biological resources*". According to the European Commission, the bioeconomy involves "*the production of biomass and the conversion of biomass into value added products, such as food, feed, bio-based products and bioenergy*" (Ronzon *et al.* 2015). Thus it encompasses several sectors, including agriculture, forestry and fishery, the pulp and paper industry, and food production, but also parts of the energy, biotechnological and chemical industries (European Commission 2012). The aim of this shift towards a bioeconomy is to enable continued economic growth while simultaneously ensuring food security, the sustainable provision of bio-based resources, a reduction in dependence on fossil fuels and a minimization of negative impacts on the environment (Richardson 2012; McCormick & Kautto 2013; Staffas *et al.* 2013).

1.2. Biomass for a developing bioeconomy

According to McKendry (2002), the term *biomass* encompasses "*all organic material that stems from plants (including algae, trees and crops)*". However, in the following sections, the term is used to refer only to biomass for industrial purposes such as the production of energy or bio-based materials, and not, for example, biomass used for food and feed. In cases where food and feed are included, this is explicitly stated and then referred to as *total biomass*.

In the year 2013, total biomass demand in the EU-28 amounted to 1073 million tonnes of dry matter. Of this, 61% (mass based) was used for the production of food and feed, 18% for bioenergy generation and 18% for the production of bio-based materials. Of the biomass used for bioenergy generation, 87% was used for heat and power production and 13% for transportation biofuels (Ronzon *et al.* 2015). One important driver of the current demand for biomass is the EU renewable energy directive, which sets mandatory levels for the use of renewable energies. Twenty percent of total energy expenditure and ten percent of the energy used in the transport sector has to be produced from renewable sources (European Commission 2009).

In a developing bioeconomy, the demand for biomass will increase significantly. It is forecasted that the global total biomass demand will almost double from 2005 to 2050 and that agricultural production needs to be increased by 70-110% in the same timeframe to meet this demand (Mauser *et al.* 2015). In the study REmap 2030 (renewable energy roadmap), the International Renewable Energy Agency (IRENA) analysed how the biomass demand of selected sectors would change if the share of renewables in the global energy mix was doubled by 2030. The outcome of this study predicted an annual increase in biomass demand of for example, 9.7% for the sector "*Transport liquids production*" and 10% for the sector "*Power and heat generation*" (Nakada *et al.* 2014).

In addition to energy production, in future, a further increase in demand will come from the production of bio-based chemicals. Nearly all materials utilised in industry could be produced from biomass instead of fossil resources (Cherubini 2010; Jong *et al.* 2012). Until now, the higher costs of bio-based alternatives often hamper their further market penetration. However, in the coming years, legal requirements and the increase in fossil-based material prices could enhance their competiveness (Jong *et al.* 2012). Furthermore, whereas energy (e.g. power and heat) can also be produced from other renewable sources such as hydropower and photovoltaic plants, biomass is the primary renewable carbon source for the production of chemicals and materials and thus for the substitution of fossil carbon sources (Binder & Raines 2009). This increasing demand for biomass for bioenergy, bio-based chemicals and materials will coincide with the already challenging task of feeding the 9 billion people who will inhabit the Earth in the year 2050 (Godfray *et al.* 2010).

1.3. Perennial crops: A sustainable biomass resource?

As elaborated above, the demand for biomass will increase significantly in the next decades and there are several biomass resources potentially available to fulfil this demand for industrial purposes. In addition to dedicated biomass crops and fuel wood, several other resources can be used including wood, processing and harvest residues. Other potential resources are animal and household wastes. The REmap 2030 assumed that around 27-34% of the predicted biomass demand will be satisfied through the production of biomass crops (Nakada *et al.* 2014).

There are various types of dedicated biomass crops and several options for their categorisation. One possibility is to distinguish between *conventional crops*, such as wheat and maize, *woody bioenergy crops,* such as eucalyptus and willow, and *grasses,* such as miscanthus and switchgrass (Smeets *et al.* 2007). Whereas the first category consists of grain, seed and high-sugar crops, which could also be used as food or feed, the other two categories consist of lignocellulosic perennials not used for human or animal consumption (Karp & Shield 2008).

One important criteria for the selection of a suitable biomass crop - besides its ability to provide biomass in sufficient quantities - is that the biomass feedstock should compete with neither food nor feed production (Tilman *et al.* 2009). Furthermore, the biomass should be provided in an environmentally benign and sustainable way (Robertson *et al.* 2008).

As already mentioned, the category *conventional crops* consists of food and feed crops (Karp & Shield 2008) and their energetic utilization is strongly associated with negative impacts on food security (Mohr & Raman 2013). This contravenes the requirement specified above of the non-competition of biomass feedstocks with food and feed production. One solution could be to cultivate perennial lignocellulosic crops on marginal or degraded land which can no longer be used for agriculture (Tilman *et al.* 2009). For biophysical reasons, it is often not possible to grow conventional crops at all, or in an economically meaningful way, on such sites. As a result, there is no direct competition between biomass production and feed and food crops.

The second requirement mentioned above is that the biomass should be supplied in an environmentally sustainable way. For example, the European Commission has defined sustainability criteria for transportation biofuels and bioliquids. These environmental sustainability criteria are mainly based on CO_2 mitigation potential targets (European Commission 2009). First generation biofuels produced from conventional crops are already having problems fulfilling current mitigation targets, and these will become stricter in the future (Humpenöder *et al.* 2013). In addition, the cultivation and utilization of conventional crops can lead to other environmental problems such as soil erosion (Vogel *et al.* 2016) and nitrate leaching (Berenguer *et al.* 2009). Agricultural systems using perennial lignocellulosic crops are often stated as an opportunity to produce biomass with low impacts on the environment (Lewandowski *et al.* 2003b; Monti *et al.* 2009; Smeets *et al.* 2009; Blanco-Canqui 2010). Furthermore, such crops can have positive effects on soil properties and biodiversity (Rowe *et al.* 2009), especially in comparison to annual crops (Haughton *et al.* 2016).

Lignocellulosic perennial crops are either woody perennials, such as short rotation coppice (e.g. willow), or perennial grasses, such as miscanthus and switchgrass. In this thesis, the focus is on perennial grasses, for several reasons. One reason is that, in Europe, there is more land under perennial grass cultivation, in particular miscanthus, than short rotation coppice (Elbersen *et al.* 2012).The biomass of perennial grasses is also more suitable for certain utilization pathways such as bioethanol (Hamelinck *et al.* 2005) and has a wider range of possible applications, for instance as a substrate for biogas production (Kiesel & Lewandowski 2017). In addition, perennial grasses have a higher water use efficiency (Podlaski *et al.* 2017) and achieve higher biomass yields than woody perennials (Marsal *et al.* 2016; Amaducci *et al.* 2017). Further advantages are that the harvest is possible with more conventional harvesting technologies, the period until the first harvest is shorter and the harvest then takes place on an annual basis, which results in a more regular cash flow (Styles *et al.* 2008).

Perennial grasses can be divided into C3 plants, such as reed canary grass (*Phalaris arundinacea L.*) and giant reed (*Arundo donax L.*), and C4 plants, such as miscanthus (*Miscanthus spp.*) and switchgrass (*Panicum virgatum L.*). In general, the C4 grasses show higher yields on most sites in Europe, except where short vegetation periods or low winter temperatures favour the C3 grasses (Lewandowski *et al.* 2003b). The current study thus concentrates on the high-yielding C4 grasses, in particular miscanthus, which has a higher biomass and energy yield potential than switchgrass (Heaton 2004; Heaton *et al.* 2008).

1.4. Miscanthus – A promising bioeconomy crop for Europe

Miscanthus is a rhizomatous C4 grass which can yield up to 25 Mg ha^{-1} yr^{-1} (dry matter) in Central Europe and more than 30 Mg ha^{-1} yr^{-1} (dry matter) under irrigated conditions in the south of Europe (Lewandowski *et al.* 2000; Iqbal *et al.* 2015). As a perennial crop, after a two-year establishment phase, it can be harvested annually over a cultivation period of twenty years (Lewandowski *et al.* 2000; Christian *et al.* 2008). Miscanthus has a high nitrogen, land-use and energy efficiency and comparatively low amounts of pesticides and fertilizers need to be applied (Lewandowski & Schmidt 2006). One study has even suggested that there is no influence of nitrogen fertilizer on the yield (Christian *et al.* 2008). They recorded stable yields over 14 years without nitrogen fertilizer input. Several of the advantages of miscanthus mentioned above are due to its perennial nature. Soil cultivation and planting is only necessary in the first year of the 20-year cultivation period, which minimizes the energy requirements of the cultivation process. In addition, after the establishment phase, miscanthus suppresses weeds and thus the need for pesticides is reduced (Lewandowski *et al.* 2000).

In its area of origin, South-East Asia, miscanthus displays a huge genetic variation. In Europe, where it was introduced in 1935 via Denmark, the genotype *Miscanthus x giganteus* is mostly cultivated (Clifton-Brown *et al.* 2015). Considerable efforts have been made in the last decade to improve the agricultural management of the miscanthus cultivation process and breed new genotypes (Lewandowski *et al.* 2016; Clifton-Brown *et al.* 2017). These novel miscanthus germplasm types show higher abiotic stress tolerances and are suitable for cultivation on marginal land (Lewandowski *et al.* 2016).

This ability to grow on marginal sites and on contaminated land (Nsanganwimana *et al.* 2014; Pandey *et al.* 2016) is a further advantage of miscanthus. Land is often categorised as marginal based on its biophysical characteristics. These have an adverse effect on the cultivation of conventional crops, which is then not economically viable. That may means that the yield is not, or barely, high enough to cover the cost of production, or the land is too inaccessible, rendering the transport of the biomass to the market too expensive (Dale *et al.* 2010; Dauber *et al.* 2012). Therefore, one broad definition of marginal land in economic terms is "*any lands that are not in commercial use in contrast to lands yielding net profit from services*" (Dale *et al.* 2010). Through the use of such marginal land for the cultivation of perennial crops like miscanthus, it would be possible to increase biomass resource availability without coming into conflict with the cultivation of conventional food crops. This offers the opportunity to significantly reduce the risk of land use competition between food or feed and biomass production for industrial purposes.

However, there are several problems associated with the use of marginal land. Its cultivation often requires considerable effort and yields are usually low compared to good-quality sites (Dauber *et al.* 2012). That is why, even when it is possible to utilise these marginal lands, the question remains whether it makes sense from an economic as well as environmental point of view.

Miscanthus biomass can be used in various conversion pathways to produce energy carriers or bio-based materials. When harvested green in October, its biomass can be used for fermentation to biogas (Jankowski *et al.* 2016; Kiesel & Lewandowski 2017). When harvested in spring, it can be used to produce heat via combustion, and this is currently the most common utilization pathway (Iqbal & Lewandowski 2014). Advantages of a late harvest are the relocation of nutrients to the rhizomes and an improvement in biomass quality through the decrease in concentration of leachable elements and nitrogen in the aboveground biomass (Lewandowski *et al.* 2003a; Iqbal & Lewandowski 2014). A late harvest also leads to a better ash melting behaviour (Iqbal *et al.* 2016). Another possible energetic utilization pathway is fermentation to ethanol (van der Weijde *et al.* 2016). However, there are still some major technological constraints in this pathway, especially in the bioconversion of lignocellulosic biopolymers, rendering it uneconomical (Taha *et al.* 2016). Examples of miscanthus-based materials are bio-composites (Muthuraj *et al.* 2016; Ogunsona *et al.* 2017), insulation material (Uihlein *et al.* 2008) and miscanthus-based biochemicals, such as 2,3-butanediol (Lee *et al.* 2015) and furfural (Kim *et al.* 2016).

Evidence suggests that the cultivation of miscanthus has positive impacts on biodiversity, especially in comparison to annual plants. Examples can be seen in the abundance of invertebrate (Semere & Slater 2007) and farmland bird populations (Bellamy *et al.* 2009). Major reasons for this are the lower ground disturbance and overall management intensity, enabled by the perennial nature of the crop (Felten & Emmerling 2011). It has a lifetime of 20 years and soil cultivation only takes place in the first year. Especially if harvested in spring, the good soil cover consisting of litter and a mulch layer reduces run-off and erosion (McCalmont *et al.* 2017).

Another advantage of miscanthus is its ability to sequester carbon in the soil. McCalmont *et al.* (2017) showed that the transition from arable to perennial crops like miscanthus increase soil organic carbon (SOC) by 0.7-2.2 Mg C ha^{-1} yr^{-1}, which corresponds to 2.6-8.1 Mg CO_2 ha^{-1} yr^{-1}.

In addition to positive impacts on biodiversity mentioned above, miscanthus also has a beneficial environmental performance in other areas. Nitrate leaching for example, an important factor in marine eutrophication, is lower for miscanthus than for annual crops (Lesur *et al.* 2014). Several studies have also shown relatively low greenhouse gas (GHG) emissions for miscanthus cultivation (Drewer *et al.* 2012), electricity generation using miscanthus pellets (Sanscartier *et al.* 2014) and heat generation via combustion of miscanthus biomass (Godard *et al.* 2013) in comparison to other biomass sources or fossil references. Voigt (2015) summarizes these points in his conclusion on the environmental performance of *Miscanthus x giganteus,* stating that it "*should be regarded as the energy crop of choice*".

1.5. Assessing the environmental sustainability of perennial crop-based value chains

In the assessment of the environmental sustainability of perennial crop-based value chains, it is crucial, for several reasons, to analyse not only the cultivation, but also the subsequent utilization of the biomass and thus the whole value chain. It has been shown that the utilization can have hot spots in other impact categories than the cultivation (Jeswani *et al.* 2015). In addition, different mitigation potentials may be achieved utilising the same bio-based raw materials in different conversion pathways (González-García *et al.* 2012), as the mitigation potentials are strongly dependant on the fossil reference which is substituted.

In the context of sustainable biomass sources, it is often stated that perennial crops have a better environmental performance than annual crops (Monti *et al.* 2009; Jeswani *et al.* 2015). However, there are also some drawbacks in the utilisation of perennial lignocellulosic plants. They often require a more complex and thus more energy-intensive conversion process, as is the case with bioethanol (Nigam & Singh 2011). In addition, in the case of biogas, they only achieve a relatively low specific methane yield compared to annual plants and this has to be compensated by a higher total biomass yield (Whittaker *et al.* 2016). This further emphasizes the necessity to analyse the value chain as a whole in a holistic assessment of the environmental performance.

When substituting a fossil reference by a perennial crop-based value chain, the bio-based alternative can yield net benefits in several impact categories, such as global warming potential

(GWP). However, it can also lead to net impacts on the environment in other impact categories, such as acidification potential (Jeswani *et al.* 2015). It is therefore crucial in such an assessment to analyse the environmental impacts in relation to a comparable product. Only then it is feasible to analyse the differences between various perennial crop-based value chains or to compare such value chains with a fossil reference.

To summarize these requirements of an integrated environmental assessment, it can be concluded that such an assessment should be able to: 1) analyse the environmental impacts of whole value chains; 2) in various impact categories; and 3) in relation to a product to make it comparable with other bio-based value chains or with a fossil reference.

According to Ness *et al.* (2007), sustainability assessments can be categorised into those that analyse, for example, environmental performance on a regional or national level, and those that are product-related. In this thesis, the environmental performance of different bio-based value chains is assessed, and so a product-related approach is applied. The globally recognized technique Life-Cycle Assessment (LCA) is the most developed methodology in this category (Ness *et al.* 2007) and, for that reason, is used here to analyse the environmental performance of the cultivation and utilization of perennial crops. The LCA methodology is specified in the two ISO standards 14040 and 14044 (ISO 2006a, 2006b). According to ISO 14040, the LCA approach enables the assessment of "*potential environmental impacts (e.g. use of resources and the environmental consequences of releases) throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave)*" (ISO 2006a). The results of this approach can be used to inform stakeholders (politicians, governmental decision makers etc.), to improve environmental performance through the identification of hot spots, and for marketing purposes (ISO 2006a). The inclusion of the entire life cycle and the aggregation of environmental impacts into categories are characteristic of LCAs (Little *et al.* 2016). One example of such an impact category is the global warming potential (GWP). However, the ISO standards do not specify which Life-Cycle Impact Assessment (LCIA) methodology has to be followed or which impact categories have to be included (ISO 2006a, 2006b). The LCIA methodology ReCiPe, for example, consists of eighteen different impact categories. Apart from climate change – which corresponds to GWP – the methodology also includes other impact categories such as freshwater and marine eutrophication, and agricultural land occupation (Goedkoop *et al.* 2009).

Most previous studies analysing the environmental performance of miscanthus cultivation and utilization only assessed GWP (Styles & Jones 2008; Felten *et al.* 2013; Dwivedi *et al.* 2015; Roy *et al.* 2015). However, as explained above, GWP represents only one of several impacts on the environment. Recent studies have shown that GWP is unsuitable as an indicator of overall environmental performance when analysing complex issues (Kalbar *et al.* 2017). The emittance of toxic substances in particular is not correlated with GHG emissions (Laurent *et al.* 2010; Laurent *et al.* 2012).

1.6. Aim of the study

As elaborated above, the ISO standard-based Life-Cycle Assessment provides a commonly accepted technique for assessing potential impacts on the environment. Various studies have already been published, which at least partly assess the environmental performance of perennial crop-based value chains. However, there are still several research gaps, not only relating to the methodological approaches used, but also to the evaluation of the results of these assessments and thus of the environmental impacts themselves.

Based on an extensive literature research, the following five research questions were developed, which are examined in this thesis:

1. What are the key parameters, which influence the environmental performance of perennial crop-based value chains?
2. Which impact categories need to be included in a holistic assessment of the environmental performance of perennial crop-based value chains?
3. Several miscanthus-based value chains are analysed, including biomass cultivation, conversion, use, End-of-Life phase, as well as the substitution of a fossil reference. The main research question in this context is: which miscanthus-based utilization pathway has the lowest impact on the environment and what are the main differences between these pathways regarding environmental hot-spots and relevant impact categories?
4. Does it make sense from an environmental point of view to use marginal land for the cultivation of perennial crops and thereby reduce land use competition with food/feed crops? To examine this research question, an LCA was conducted using data from field trials on marginal land.
5. Does the cultivation and subsequent utilization of perennial crop biomass have lower environmental impacts than annual crop biomass? In order to examine this research

question, an LCA was conducted using input data from field trials of annual and perennial crops cultivated under *ceteris paribus* conditions.

The overall aim of this study is to analyse and evaluate key parameters and methodological approaches for the holistic assessment of the environmental performance of perennial crop-based value chains. In a second step, the selected approaches are applied in order to assess the environmental performance of different bio-based value chains using actually measured and comparable data for the cultivation processes.

1.7. Publications

The four publications included in this thesis are structured into two chapters. The first chapter is entitled "*Evaluating key parameters for holistically assessing the environmental performance of perennial crop-based value chains*". This chapter examines which parameters have an important influence on the environmental performance of perennial crop-based value chains and which impact categories have to be included in an LCA study to holistically assess the net benefits and impacts of such value chains. In addition, the main environmental hot spots are specified. In this context, the main uncertainties involved in assessing the impacts on the environment are also determined. The publications included in this chapter are:

2.1 Meyer, F., Wagner, M. & Lewandowski, I. (2017) Optimizing GHG emission and energy-saving performance of miscanthus-based value chains. In: *Biomass Conversion and Biorefinery*, **7**(2), 139-152. doi:10.1007/s13399-016-0219-5

2.2 Wagner, M. & Lewandowski, I. (2017) Relevance of environmental impact categories for perennial biomass production. In: *GCB Bioenergy*, **9**(1), 215-228. doi:10.1111/gcbb.12372

The second chapter consists of two publications and is entitled "*Environmental impacts and benefits of perennial crop-based value chains"*. In the first publication, the relevance of different impacts categories are assessed in the broader context of an analysis of six different miscanthus-based value chains. Furthermore, the differences in the net benefits and impacts of different utilization pathways on six locations in Europe are analysed. Miscanthus was cultivated on marginal land at two locations to analyse the impact on the environmental performance. In the second paper, the environmental performance of the utilization of perennial plants is compared to that of annual plants in one exemplary value chain. The following two publications are included in this chapter:

3.1 Wagner, M., Kiesel, A., Hastings, A., Iqbal, Y. & Lewandowski, I. (2017) Novel miscanthus germplasm-based value chains: A Life Cycle Assessment. In: *Frontiers in Plant Science*. doi: 10.3389/fpls.2017.00990

3.2 Kiesel, A., Wagner, M. & Lewandowski, I. (2017) Environmental performance of miscanthus, switchgrass and maize: can C4 perennials increase the sustainability of biogas production? In: *Sustainability*, **9**(1), 5. doi:10.3390/su9010005

1.8. References

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2. Evaluating key parameters for holistically assessing the environmental performance of perennial crop-based value chains

2.1 Optimizing GHG emission and energy-saving performance of miscanthus-based value chains

In this sub-chapter the global warming potential and the fossil fuel depletion of three different miscanthus-based value chains (combustion, second-generation bioethanol, and production of insulation material) were assessed for various miscanthus genotypes grown on five different locations across Europe. Hereby, with regard to the first research question, a strong focus was on the influence of genotype, location and selected value chain on the environmental performance.

This sub-chapter is published in the journal *Biomass Conversion and Biorefinery* (accessible online at https://link.springer.com/journal/13399), as:

Meyer, F., Wagner, M. & Lewandowski, I. (2017) Optimizing GHG emission and energy-saving performance of miscanthus-based value chains. *Biomass Conversion and Biorefinery*, **7**(2), 139-152. doi:10.1007/s13399-016-0219-5

2.2 Relevance of environmental impact categories for perennial biomass production

This sub-chapter analysed the relevance of different impact categories for the assessment of the environmental performance of perennial crop-based value chains addressing thereby the second research question of this thesis. A Life-Cycle Assessment was conducted to analyse the environmental performance of the cultivation of miscanthus and willow biomass and the subsequent utilization in a biomass heater in eighteen impact categories. In order to assess the relevance of the different impact categories a normalisation approach was used.

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Relevance of environmental impact categories for perennial biomass production

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Abstract

The decarbonization of the economy will require large quantities of biomass for energy and biomaterials. This biomass should be produced in sufficient quantities and in a sustainable way. Perennial crops in particular are often cited in this context as having low environmental impacts. One example of such crops is miscanthus, a tall perennial rhizomatous C4 grass with high yield potential. There are many studies which have assessed the global warming potential (GWP) of miscanthus cultivation. This is an important impact category which can be used to quantify the environmental benefit of perennial crops. However, the GWP only describes one impact of many. Therefore, the hypothesis of this study was that a holistic assessment also needs to include other impact categories. A life cycle assessment (LCA) with a normalization step was conducted for perennial crops to identify relevant impact categories. This assessed the environmental impact of both miscanthus and willow cultivation and the subsequent combustion for heat production in eighteen categories using a system expansion approach. This approach enables the inclusion of fossil reference system hot spots and thus the evaluation of the net benefits and impacts of perennial crops. The normalized results clearly show the benefits of the substitution of fossil fuels by miscanthus or willow biomass in several impact categories (e.g. for miscanthus: climate change −303.47 kg CO_2 eq./MWh_{th}; terrestrial acidification: −0.22 kg SO_2 eq./MWh_{th}). Negative impacts however occur, for example, in the impact categories marine ecotoxicity and human toxicity (e.g. for miscanthus: +1.20 kg 1.4-DB eq./MWh_{th} and +68.00 kg 1.4-DB eq./MWh_{th}, respectively). The results of this study clearly demonstrate the necessity of including more impact categories than the GWP in order to be able to assess the net benefits and impacts of the cultivation and utilization of perennial plants holistically.

Keywords: combustion, environmental performance, global warming potential, life cycle assessment, miscanthus, normalization, perennial crop, willow

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Introduction

In 2009, the European Commission set mandatory targets for the production and promotion of energy from renewable resources. The EU renewable energy directive stipulates that, by the year 2020, 20% of total EU energy consumption should come from renewable sources and at least 10% of petrol and diesel consumption for transport should be supplied through biofuels (European Commission, 2009). The European Commission expects the use of renewable energy to increase considerably over the next decades and its proportion of gross final energy consumption to reach values of up to 55% by the year 2050. In the energy roadmap 2050, the European Commission also emphasizes the need for large quantities of biomass for heat, electricity and transport to achieve the goal of the decarbonization of the economy (European Commission, 2011). There is a wide range of biomass resources available, which can be potentially exploited for bioenergy production. Of these, dedicated energy crops such as miscanthus have emerged as a promising future feedstock for biomass-based energy production. For this reason, miscanthus was chosen as the main representative perennial crop for this study. In addition to miscanthus, willow short rotation coppice was included in this study to examine whether there are any differences between woody perennials and perennial grasses.

Miscanthus is a tall perennial rhizomatous C4 grass, which can yield up to 25 Mg ha^{-1} yr^{-1} (dry matter) in Central Europe after a two-year establishment period and can be harvested annually over a twenty-year cultivation period (Lewandowski *et al.*, 2000; Christian *et al.*, 2008; Felten *et al.*, 2013; Iqbal *et al.*, 2015). It is a low-input crop with a high nitrogen, land-use and energy efficiency (Lewandowski & Schmidt, 2006) and has the potential to remove CO_2 from the atmosphere through

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carbon sequestration (Clifton-Brown *et al*., 2007). In an editorial regarding the environmental benefits of miscanthus, Voigt (2015) recommends *Miscanthus x giganteus* 'as the energy crop of choice'. In the context of sustainability requirements, it is important to assess the performance of each crop in economic, social and ecological terms. This study focuses on evaluating the ecological performance of the utilization of perennial energy crops. One option available for such an evaluation is life cycle assessment (LCA). Life cycle assessment is a method which is standardized by two ISO norms – 14040 and 14044 (ISO, 2006a,b). In the last ten years, several papers have been published which use LCA to assess the potential environmental impacts and benefits of miscanthus. The impact categories examined in these are presented in Table 1.

As shown in Table 1, most of the studies carried out to evaluate the environmental performance of miscanthus focus on one impact category – the global warming potential (GWP). In the EU, political support for bioenergy aims at reducing greenhouse gas (GHG) emissions from fossil fuels. Therefore, any LCA study on bioenergy includes an assessment of the GWP.

Agriculture contributes significantly to GWP. The agricultural sector is responsible for about 10–12% of total anthropogenic emissions of greenhouse gases globally (Smith *et al*., 2007). However, agriculture also has an influence on other impact categories, such as eutrophication potential (EP) and acidification potential (AP) (EEA, 2005; Rice & Herman, 2012). In the European Union, the agricultural sector is responsible for 93.3% of ammonia emissions (Eurostat, 2015), which are a main driver of AP. In addition, the use of mineral and organic fertilizers on agricultural land leads to a gross nitrogen surplus of 51 kg nitrogen ha^{-1} yr^{-1} and a gross phosphorus surplus of 2 kg P ha^{-1} yr^{-1} (Eurostat, 2012, 2013). These nutrients can enter groundwater, for example, through nitrate leaching and lead to marine eutrophication. High concentrations of nutrients in water can pose health risks for humans (Di & Cameron, 2002). Nitrate leaching is only one example of the manifold emissions released by the entire agricultural value chain and the subsequent biomass utilization. From this, it can be concluded that, when trying to assess the environmental performance of perennial crops, the estimation of GWP alone is too simplistic. An analysis of the studies on the environmental performance of perennial crops listed in Table 1 confirms this conclusion. Jeswani *et al.* (2015) found that, when considering the GWP of second-generation biofuels, the production of the feedstock for the ethanol plant – the cultivation of the biomass – is the most important hot spot. However, the influence of the feedstock on other impact categories is relatively small. For the impact categories abiotic resource depletion (ADP, elements), AP, EP and freshwater aquatic ecotoxicity potential (FAETP), the main driver is the subsequent conversion of the biomass. Considering GWP alone grossly underestimates the

Table 1 Impact categories used in LCA studies on miscanthus

Authors	Method	GWP	EP	AP	ADP	POCP	ODP	TET	FET	MET	HT
Jeswani *et al.* (2015)	CML	x	x	x	x	x	x	x	x	x	x
Monti *et al.* (2009)	CML	x	x	x	x	–	x	x	x	x	x
Godard *et al.* (2013)	CML; USES-LCA 2.0; CED	x	x	x	x	x	x	x	–	–	–
Styles *et al.* (2015)	CML	x	x	x	x	–	–	–	–	–	–
Nguyen & Hermansen (2015)	EDIP 97; IPCC; Impact 2002+	x	x	x	–	–	–	–	–	–	–
Murphy *et al.* (2013)	CML	x	x	x	–	–	–	–	–	–	–
Brandão *et al.* (2011)	CML	x	x	x	–	–	–	–	–	–	–
Tonini *et al.* (2012)	EDIP 2003	x	x	–	–	–	–	–	–	–	–
Sanscartier *et al.* (2014)	IPCC	x	–	–	–	–	–	–	–	–	–
Styles & Jones (2008)	IPCC	x	–	–	–	–	–	–	–	–	–
Felten *et al.* (2013)	IPCC	x	–	–	–		–	–	–	–	–
Dwivedi *et al.* (2015)	n.a.	x	–	–	–	–	–	–	–	–	–
Brandão *et al.* (2010)	IPCC	x	–	–	–		–	–	–	–	–
Scown *et al.* (2012)	n.a.	x	–	–	–	–	–	–	–	–	–
Roy *et al.* (2015)	n.a.	x	–	–	–	–	–	–	–	–	–
Wang *et al.* (2012)	n.a.	x	–	–	–	–	–	–	–	–	–
Iqbal *et al.* (2015)	IPCC	x	–	–	–	–	–	–	–	–	–
Parajuli *et al.* (2015)	Stepwise 2006	x	–	–	–	–	–	–	–	–	–
Smeets *et al.* (2009)	IPCC	x	–	–	–	–	–	–	–	–	–

GWP, global warming potential; EP, eutrophication potential; AP, acidification potential; ADP, abiotic depletion potential; POCP, photochemical ozone creation potential; ODP, ozone depletion potential; TET, terrestrial ecotoxicity; FET, freshwater ecotoxicity; MET, marine water ecotoxicity; HT, human toxicity.

influence of the conversion stage on the environmental performance of the second-generation biofuels. Godard *et al.* analysed the environmental performance of heat produced from different feedstocks (flax shives, miscanthus, cereal straw, linseed straw and triticale as whole plant). Using economic allocation, heat produced from miscanthus has the lowest GWP, but scores worse in all the other impact categories in comparison with flax shives as feedstock. If an allocation based on mass is used, heat produced from miscanthus has the best environmental performance in all selected impact categories (Godard *et al*., 2013). If GWP alone is analysed, it is not only impossible with economic allocation to select the feedstock with the best environmental performance, but it is also impossible to thoroughly analyse the impact of different allocation procedures. The comparison of different perennial crops revealed that marine water ecotoxicity is the most affected impact category after normalization. It is 20–30 times higher than the other categories. Switchgrass, for example, achieved very low values in this important impact category. For this reason, it is a very suitable crop for sites near rivers or coastlines (Monti *et al*., 2009). In order to select the biomass crop best adapted to specific conditions, it is essential to have a complete picture of the environmental performance of each crop. Various studies on short rotation coppice (poplar and willow), which analysed the environmental performance of the cultivation and utilization in several impact categories, confirm the hypothesis that a number of categories need to be assessed. González-García *et al.* (2012a) showed that, for poplar plantations, in addition to GWP, the impact categories ADP, AP, EP, FE and ME were the most significant after a normalization step. Further results showed that the selection of the most environmentally friendly energy conversion pathway for willow chips largely depends on which impact categories analysed (González-García *et al*., 2012b). The same applies to the management practice of willow plantations (González-García *et al*., 2012c).

In this study, an LCA was conducted according to the ISO standards 14040 and 14044 to analyse the environmental performance of miscanthus cultivation and utilization in eighteen different impact categories in comparison with a fossil reference (ISO, 2006a,b). This was done employing the widely used ecoinvent database (version 3.1) and openLCA, an open source LCA software. One objective was to identify those impact categories that need to be included in a holistic assessment of environmental impacts and benefits of the production and utilization of perennial crops, such as miscanthus. To compare the importance of the different impact categories analysed, a normalization step was carried out. According to ISO, normalization is defined as 'calculation of the magnitude of category indicator results relative to reference information' (ISO, 2006a). Normalization factors were taken from the ReCiPe methodology. The result for each impact category is divided by the respective emissions caused by an average European citizen in the year 2000. This results in values without units, which show the calculated emissions as a proportion of the emissions of an average European citizen. Through this additional calculation, it is possible to compare the importance of different impact categories (Goedkoop *et al*., 2008). A hot spot analysis was also conducted. It reveals which processes are responsible for the largest share of emissions in each impact category.

Through the normalization and the hot spot analysis, it is possible to determine not only the relevant impact categories in the cultivation and utilization of perennial crops, but also which processes or emission sources are most important for each category.

This study aimed to provide guidelines for future research on the environmental performance of perennial crops, with regard to both the choice of relevant impact categories and the focus on data for the most important processes and emission sources.

Material and methods

Scope and boundaries

The scope of this study is a cradle-to-grave analysis of the environmental performance of the cultivation of miscanthus (*Miscanthus x giganteus*) and willow (*Salix viminalis*) short rotation coppice (variety 'Tora') and subsequent combustion in a biomass-fuelled boiler. In order to compare this performance with a fossil reference (heat produced through combustion of light fuel oil), a system expansion approach was applied. This approach enables the inclusion of fossil reference system hot spots. The outcome of this analysis shows the net benefits and impacts through the substitution of fossil fuel by the energetic utilization of miscanthus and willow chips. One megawatt hour of heat (MWh_{th}) was chosen as the functional unit. These systems are described in Fig. 1. The system boundaries include the production of the mineral fertilizers and the pesticides used, the production of the propagation material (miscanthus rhizomes and willow cuttings) and the land management (soil preparation, planting, mulching, fertilizing, spraying of pesticides, harvesting, recultivation) over a twenty-year cultivation period. The miscanthus was mulched in the first year and harvested from the second year onwards; the willow plantation was harvested from the fourth year on and then in three-year cycles. Both crops were harvested with a self-propelled forage harvester. The biomass is then transported to a biomass heater where it is combusted to produce heat. The coarse ash is rich in potassium and phosphorus and is used as fertilizer. The fly ash is disposed of in landfill.

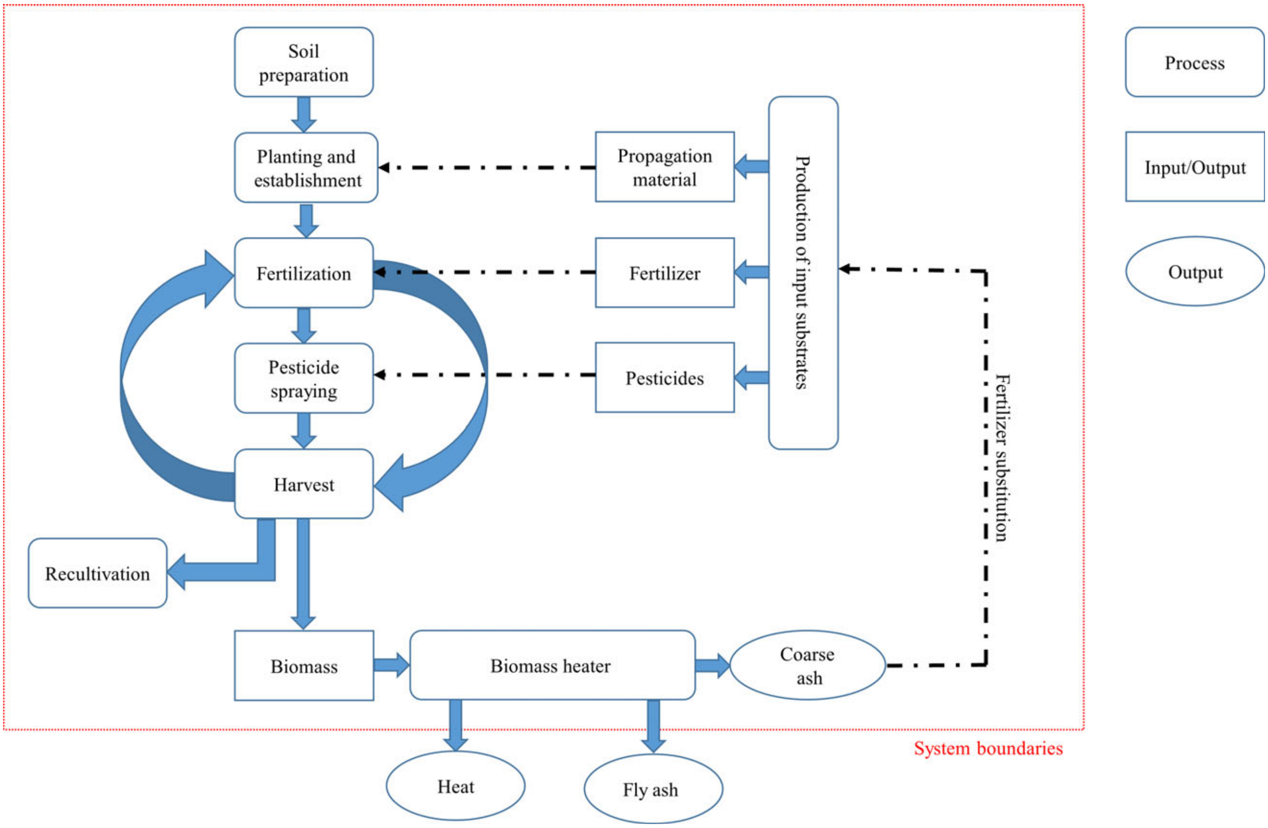

Fig. 1 System description and boundaries for miscanthus and willow biomass production and subsequent utilization in a biomass heater.

Life cycle inventory

The data for the cultivation process used in this LCA study was obtained from a multiannual field trial at Ihinger Hof, a research station of the University of Hohenheim. The Ihinger Hof is located in southwest Germany (48.75°N and 8.92°E). The soil belongs to the soil class Haplic Luvisol. The mean annual temperature for the measurement period was 9.2 °C, and the average annual rainfall was 707.5 mm. The experimental design of the trial is described in Iqbal *et al.* (2015). Data on cultivation practices, fertilizer and pesticide inputs as well as the yields was available for a 10-year period from 2002 to 2012. For both perennial crops, three different fertilizer regimes were applied: N1 with 0 kg of nitrogen, N2 with 40 kg nitrogen and N3 with 80 kg nitrogen per year and hectare in the form of calcium ammonium nitrate. Potassium and phosphate fertilizer levels were the same in all three application regimes. For miscanthus, herbicides only were applied (described in Iqbal *et al.*, 2015). For willow, one insecticide (Karate Zeon, Syngenta, active ingredient 100 g l^{-1} lambda-cyhalothrin) was applied in 2004 at a rate of 0.075 l ha^{-1}. Three herbicides (3 l ha^{-1} Durano, Monsanto, active ingredient 360 g l^{-1} glyphosate; 5 l ha^{-1} U-46 M-Fluid, Nufarm, active ingredient 500 g l^{-1} MCPA; and 2 l ha^{-1} Starane 180, Syngenta, active ingredient 180 g l^{-1} fluroxypyr) were applied in 2006. In the following years, no pesticides were applied. The principle data for the cultivation of miscanthus and willow used in this analysis is summarized in Table 2. As yield data was only available for the first ten years, it was predicted for the rest of the 20-year cultivation period. For willow, the average of the three measured harvests (years 4, 7 and 10) was taken to estimate the yield for years 11 to 20. For miscanthus, the average of year four to ten was taken for this prediction. The yields of the first three years were excluded in the estimation because the crop was still in its establishment period and has lower yield than after full establishment. However, the yield data inputted into the LCA is the average yield over the whole cultivation period including the establishment phase. Background data for the environmental impacts associated with the production of the input substrates and the cultivation processes (soil preparation, harvesting) was taken from the ecoinvent database version 3.1 (Weidema *et al.*, 2013).

Direct N_2O and NO emissions from mineral fertilizers were estimated according to Bouwman *et al.* (2002). Indirect N_2O emissions from mineral fertilizers and N_2O emissions from harvest residues were calculated according to IPCC (2006). Ammonia emissions were estimated using emission factors from the Joint EMEP/CORINAIR Atmospheric Emission Inventory Guidebook (EMEP/CORINAIR, 2001). Nitrate leaching to groundwater was calculated according to the SQCB – NO_3 model described in Faist Emmenegger *et al.* (2009). Phosphate and phosphorus emissions to surface water

Table 2 Summary of in- and outputs of each perennial crop

	Miscanthus			Willow		
Values in kg yr^{-1} ha^{-1}	N1	N2	N3	N1	N2	N3
N	0	40	80	0	40	80
K_2O	128	128	128	64	64	64
P_2O_5	32	32	32	32	32	32
Pesticides	1.375	1.375	1.375	0.504	0.504	0.504
Dry Matter Yield	16404	19684	20333	16013	17755	20583

and groundwater as well as heavy metal emissions to agricultural soils were calculated according to Nemecek & Kägi (2007). The nitrogen, phosphorus and heavy metal emissions are summarized for the respective crops and fertilizer levels in Table S1.

As no data for the transport of the input substrates (fertilizer, pesticides and propagation material) to the farmer and the biomass to the biomass heater were available, a transport distance of 150 km for the input material and 50 km for the biomass, both by truck, was assumed. The average field-to-farm distance was assumed to be 2 km. The emission stage for the truck used was EUR5. The process data for the transportation of the input material and the biomass was taken from the ecoinvent database (Weidema *et al*., 2013).

The biomass heater used in this LCA study is a furnace of 300-kW capacity for heat production. The background data for the emissions associated with the combustion of the different biomasses is taken from the ecoinvent database. The data set is based on a Froling Turbomat 320-kW woodchip boiler. The thermal efficiency is assumed to be 75%. As stipulated in the process description of the ecoinvent database, this thermal efficiency is lower than in the technical specification, because it represents the average annual operation, including start and stop phases (Weidema *et al*., 2013). As there is not enough specific information available regarding the emissions from the combustion of miscanthus, a straw combustion process was used as a worst-case assumption. Where miscanthus-specific emissions factors were available, the straw combustion process was adapted accordingly. This was the case for carbon monoxide, sulphur dioxide, hydrogen chloride, nitrogen oxides and particulates. The emission factors are based on Dahl & Obernberger (2004). A scenario analysis with an improved emission setting was performed to analyse the impact of this assumption.

Miscanthus has a water content of around 15% at the time of harvest, so a further drying process was not necessary. This corresponds to a calorific value of 4.3 kWh kg^{-1} fresh biomass. The wood chips have a water content of 50% at the time of harvest. The chips are then stored on the farm where natural drying is employed. This process results in a water content of around 20%, which corresponds to a calorific value of 3.86 kWh kg^{-1} fresh biomass.

For all willow fertilization levels and the N2 and N3 miscanthus variants, the use of the coarse ash as fertilizer allows the crops to be cultivated without additional input of mineral phosphate or potassium fertilizers. Therefore, the boundaries applied in this study only include nitrogen fertilizer. For the N1 miscanthus variant, an additional input of 4 kg P_2O_5 and 14 kg K_2O was necessary. Information on ash content, amount of fly and coarse ash and the nutrient as well as the heavy metal content of the coarse ash can be found in Table S2. The fly ash is disposed to landfill.

Choice of impact categories

This LCA study used the life cycle impact assessment method ReCiPe, which consists of eighteen different impact categories (Goedkoop *et al*., 2008). All mid-point indicators described in the ReCiPe methodology were included. The following impact categories were considered: climate change (CC), which corresponds to global warming potential (GWP); ozone depletion (OD); terrestrial acidification (TA); freshwater eutrophication (FE); marine eutrophication (ME); human toxicity (HT); photochemical oxidant formation (POF); particulate matter formation (PMF); terrestrial ecotoxicity (TET); freshwater ecotoxicity (FET); marine ecotoxicity (MET); ionizing radiation (IR); agricultural land occupation (ALO); urban land occupation (ULO); natural land transformation (NLT); mineral resource depletion (MRD); fossil fuel depletion (FD); and water depletion (WD). Characterization and normalization factors were taken from Goedkoop *et al.* (2008). A normalization factor for the impact category water depletion is not available in the ReCiPe methodology. For this reason, only absolute values are given for this impact category.

Results

Life cycle impact assessment (LCIA)

Table 3 presents the environmental impact in the different impact categories per MWh_{th} of the miscanthus and willow cultivation and subsequent combustion of the biomass. The results are shown for the N2 fertilization level. With these data, it is possible to compare the environmental performance of cultivation and combustion for the two perennial crops in different impact categories. However, due to the different reference units, it is not possible to compare the significance of the different impact categories themselves. For that, a normalization step is necessary.

Table 3 LCIA of the combustion of miscanthus and willow (fertilization level N2) per MWh_{th}

Impact category	Miscanthus	Willow	Reference unit
Fossil fuel depletion	7.3780	7.2425	kg oil eq.
Agricultural land occupation	135.7814	168.6545	m^2*a
Photochemical oxidant formation	0.5002	0.7597	kg NMVOC
Particulate matter formation	0.2008	0.6375	kg PM_{10} eq.
Marine ecotoxicity	1.7175	1.4394	kg 1,4-DB eq.
Natural land transformation	0.0072	0.0074	m^2
Ozone depletion	2.91E-06	0.0829	kg CFC-11 eq.
Terrestrial ecotoxicity	0.1781	0.0030	kg 1,4-DB eq.
Freshwater eutrophication	0.0220	0.0258	kg P eq.
Freshwater ecotoxicity	0.7532	1.5774	kg 1,4-DB eq.
Mineral resource depletion	2.2788	2.8192	kg Fe eq.
Urban land occupation	0.5211	0.5168	m^2*a
Human toxicity	84.3484	6.8041	kg 1,4-DB eq.
Water depletion	97.6863	103.8735	m^3
Marine eutrophication	0.1626	0.2581	kg N eq.
Ionising radiation	4.3592	4.5147	kg U235 eq.
Climate Change	37.4125	40.1806	kg CO_2 eq.
Terrestrial acidification	0.5058	0.5015	kg SO_2 eq.

Comparison of the environmental performance of the cultivation and combustion of miscanthus and willow

Figure 2 presents a comparison between the environmental performance of the cultivation and utilization of miscanthus and willow. For each crop, the LCIA results of the fertilizer level N2 (40 kg nitrogen) are shown. In fifteen of the eighteen impact categories analysed, there are no differences in the rankings of the impact categories between the two perennial crops. The exceptions are human toxicity, terrestrial ecotoxicity and particulate matter formation. In the case of human toxicity potential, the values for willow are significantly lower than for miscanthus. This is in part due to the higher uptake of heavy metals by willow than by miscanthus. These are partially removed from the system through the disposal of the fly ash (which is rich in heavy metals) to landfill. Another reason is the fact that the combustion process of miscanthus produces higher emissions. These are also the main cause of the significantly higher terrestrial ecotoxicity. Differences in the emissions from the combustion of the biomasses are also responsible for the lower particulate matter formation with miscanthus than with willow. Heat produced

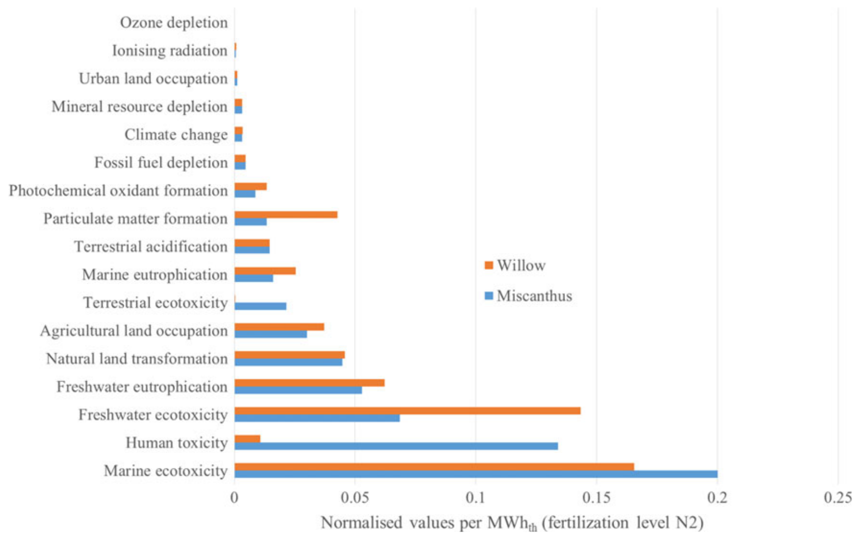

Fig. 2 Assessment of the environmental performance of the cultivation and utilization of miscanthus and willow.

by the combustion of willow chips also has a higher marine eutrophication caused by slightly higher nitrate leaching rates during cultivation. Nitrate is a main source of the marine eutrophication potential. A further difference in the environmental performance of the utilization of the two crops is the freshwater ecotoxicity. Here, the differences in emissions from the combustion process are the main driver.

Life cycle emissions of miscanthus cultivation and combustion using a different combustion scenario

The LCIA results reveal that most of the main differences seen in the comparison of the cultivation and utilization of the two biomasses stem from the combustion process rather than the cultivation phase. The results for the comparison between miscanthus and willow cultivation and combustion using a second combustion scenario for the miscanthus biomass are presented in Fig. 3. A scenario analysis was performed with a boiler with emission characteristics comparable to wood combustion. Under this setting, the differences between willow and miscanthus are much less pronounced.

Life cycle emissions of miscanthus cultivation and combustion using a system expansion approach

In Fig. 4, the results of the LCIA of the miscanthus cultivation and utilization are shown for the three different fertilization regimes using a system expansion approach. The values include the emissions avoided through the substitution of heat produced from a conventional furnace using light fuel oil by heat produced from a biomass heater. In this approach, negative values represent burdens avoided by the substitution of fossil by renewable fuels, while positive values represent an additional impact due to the use of the biomass heater. The substitution of fossil fuels by miscanthus biomass leads to burdens avoided especially in the impact categories fossil fuel depletion, climate change and terrestrial acidification. However, it causes additional impacts in the categories marine ecotoxicity, human toxicity, agricultural land occupation, freshwater eutrophication and terrestrial ecotoxicity. It should be noted that these results are strongly depending on the fossil reference used. In the case of the substitution of heat produced from hard coal instead of light fuel oil, the use of miscanthus biomass would lead to an avoided burden in the impact category human toxicity instead of an additional impact (data not shown).

In most impact categories analysed, there are only small differences between the three fertilization levels. These are mainly caused by differences in yield and the amount of fertilizer used. Marine eutrophication, for which nitrate leaching is an important driver, increases significantly from N1 to N3 due to the higher nitrate leaching through the additional input of nitrogen fertilizer.

Life cycle emissions of willow cultivation and combustion using a system expansion approach

The substitution of heat produced from the combustion of light heating fuel by heat produced from willow chips leads to burdens avoided especially in the impact categories fossil fuel depletion, climate change and

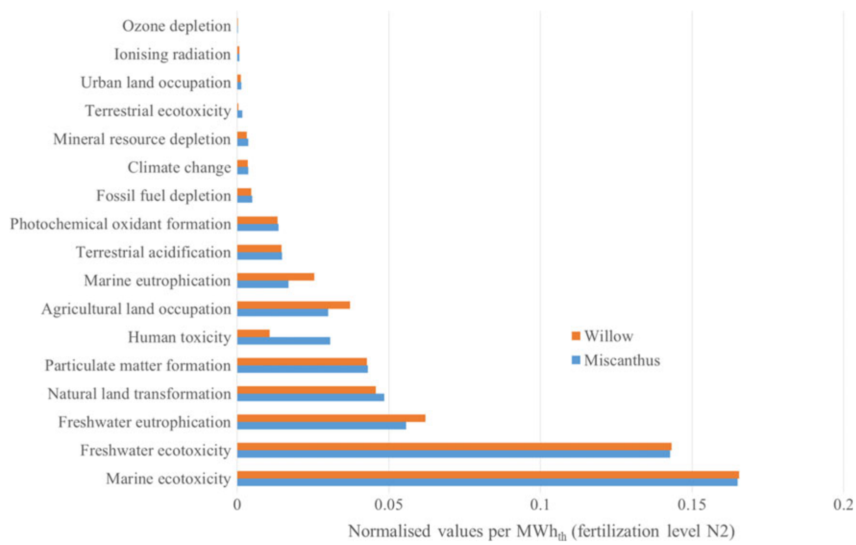

Fig. 3 Scenario analysis of the environmental performance of combustion of miscanthus and willow.

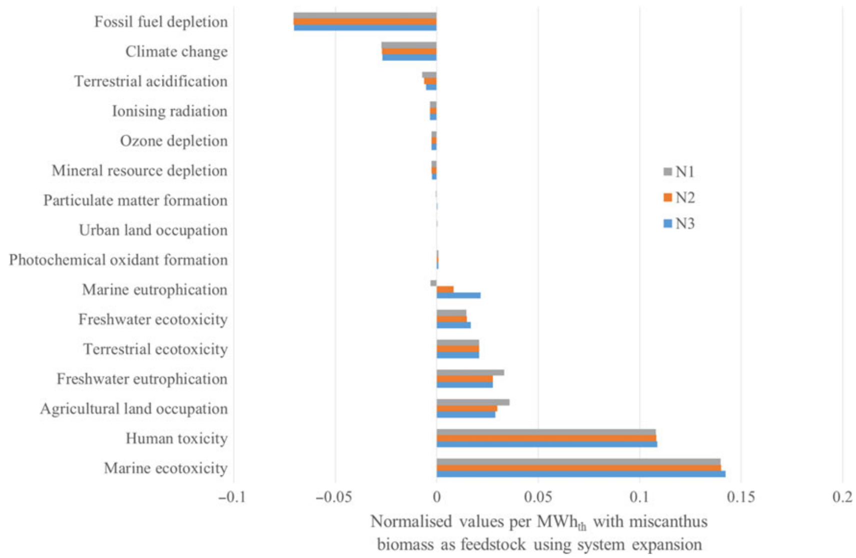

Fig. 4 Assessment of the environmental performance of the cultivation and utilization of miscanthus using a system expansion approach.

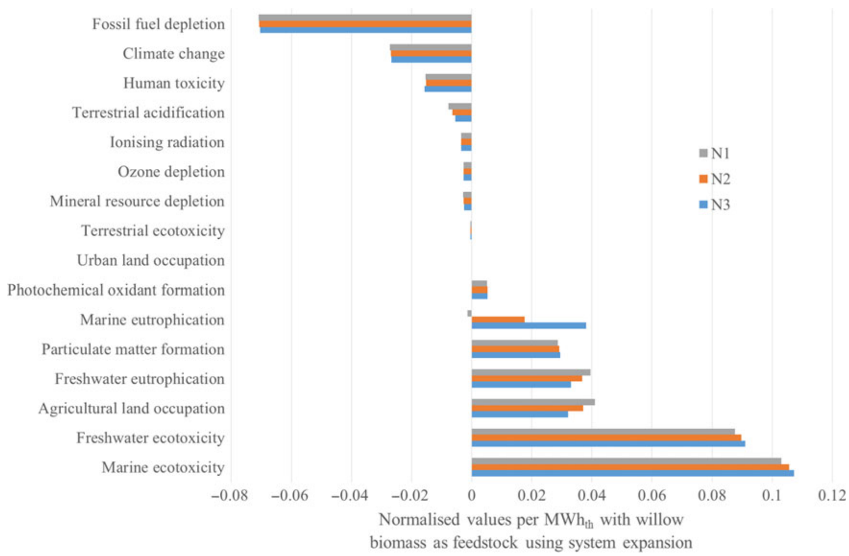

Fig. 5 Assessment of the environmental performance of the cultivation and utilization of willow using a system expansion approach.

human toxicity (see Fig. 5). However, it also causes additional impacts in the categories marine ecotoxicity, freshwater ecotoxicity, agricultural land occupation, freshwater eutrophication and particulate matter formation.

The differences between the results of the three fertilization levels (e.g. in marine eutrophication) can be explained in the same way as for miscanthus. The results for the impact category urban land occupation are mapped in Fig. 5; however, the normalized values are too small to be visible (−9.6 E-05).

Comparison of the environmental performance of the cultivation and combustion of miscanthus and willow using a system expansion approach

Figure 6 shows a comparison of heat produced from miscanthus and willow biomass at fertilization level N2 using a system expansion approach. Because the fossil reference is identical for both crops, the reasons for the differences in environmental performance between heat produced from miscanthus and willow are the same as for the normalized values without system expansion.

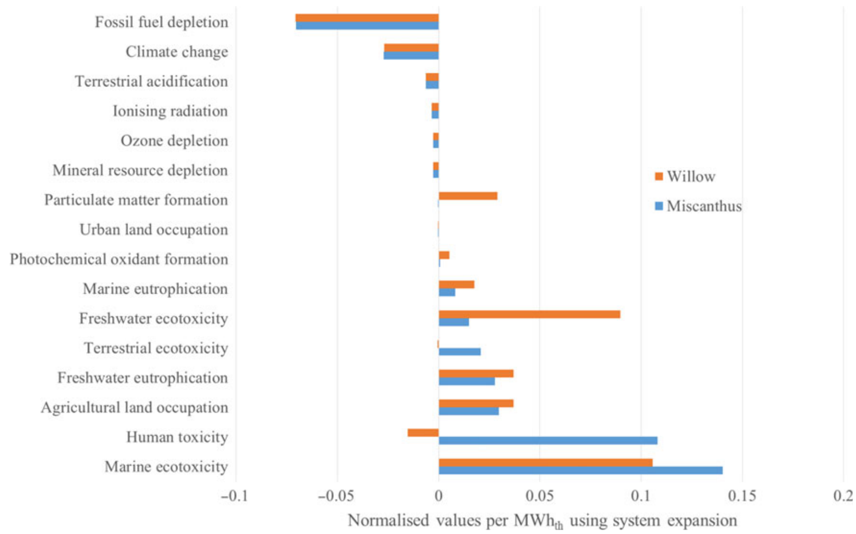

Fig. 6 Comparison of the environmental performance of the cultivation and utilization of miscanthus and willow (fertilization level N2) using a system expansion approach.

While the results for both crops are very similar in impact categories such as climate change, fossil fuel depletion and ionizing radiation, there are substantial differences in other impact categories, especially freshwater ecotoxicity and human toxicity.

Results for natural land transformation (data not shown) using a system expansion approach are lower than −0.7 (in normalized values) and are therefore much lower than the results of all other impact categories.

Hot spot analysis

The hot spot analysis reveals which processes are responsible for the largest share of emissions in each impact category. The grouping 'cultivation' summarizes all cultivation steps up to and including harvest, the production and transport of input substrates (e.g. mineral fertilizers), and the transport of the coarse ash from the biomass heater back to the field. The transport of the input substrates and ash each accounts for <1% of the total emission of the respective impact category. For this reason, they are not represented individually in the hot spot analysis (Figs 7–9). The grouping 'biomass transport' represents the environmental impacts of the transport of the biomass from the field to the biomass heater. The grouping 'combustion' indicates the proportion of total emissions associated with the combustion process. Overall, the results of the hot spot analysis show no large differences between the utilization of miscanthus and willow for most of the impact categories.

In each of the impact categories shown in Fig. 7, the combustion of the biomass has a share of over 90% of total emissions. The negative values seen for the cultivation of willow can be explained by the higher uptake of heavy metals. The emissions associated with biomass transport have a substantial impact especially on the impact categories natural land transformation, urban land occupation, fossil fuel depletion and mineral resource depletion (see Fig. 8). The combustion process has an impact of over 50% on almost all impact categories shown in Fig. 8, in particular terrestrial acidification and photochemical oxidant formation. The differences in impact of the cultivation stage of miscanthus and willow on the photochemical oxidant formation are much smaller in absolute than in percentage terms.

The cultivation stage has a substantial impact on the categories shown in Fig. 9. In the case marine eutrophication, nitrate leaching is the main cause. Nitrous oxide emissions from the use of mineral nitrogen fertilizer are also an important driver of climate change. The impact of cultivation on freshwater eutrophication is mainly due to phosphor emissions to ground water and surface water. Its impact on ozone depletion stems from mineral fertilizer production and agricultural management. The cultivation stage is also responsible for over 99% of agricultural land occupation.

Discussion

Normalization and system expansion

The normalization of the results is a useful way of assessing the importance of different impact categories. It shows the impact of perennial crop production and utilization in each category and thus helps in the selection of the relevant ones for an assessment of their

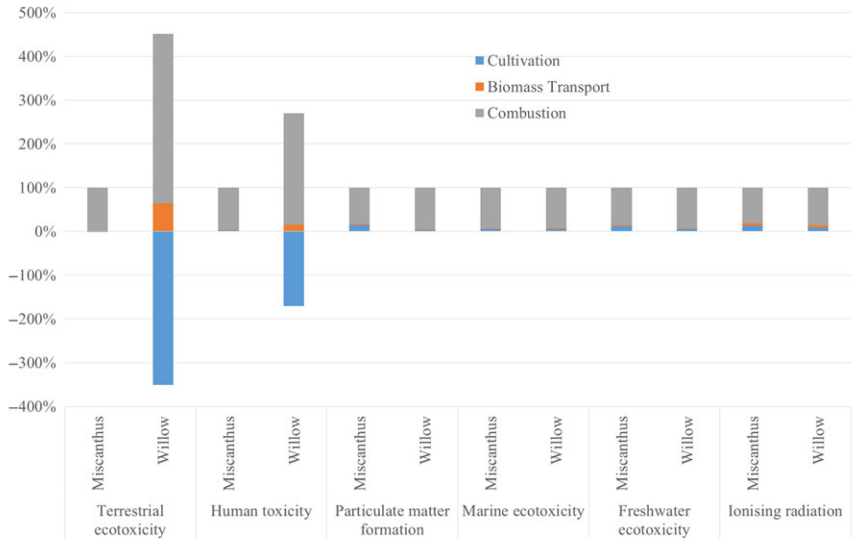

Fig. 7 Hot spot analysis of environmental impacts of cultivation and combustion of two perennial crops. Bars show the contribution of the processes to the overall emissions in each impact category.

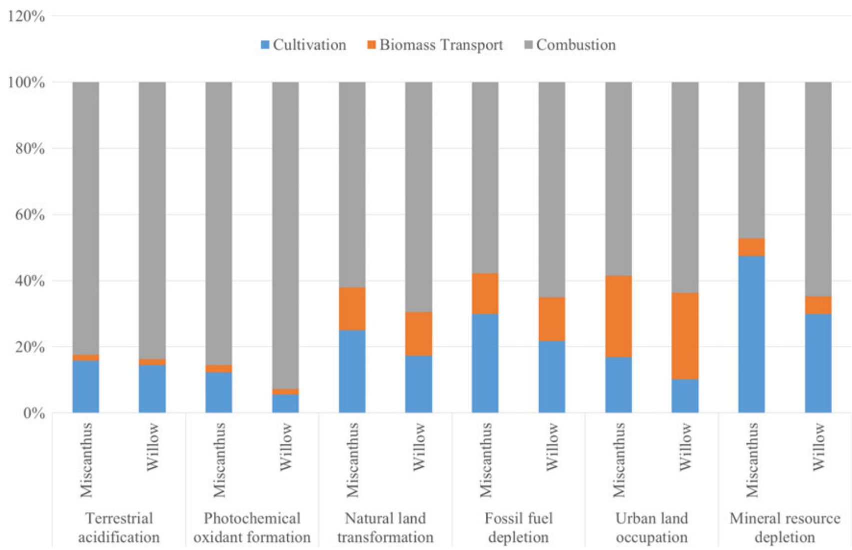

Fig. 8 Hot spot analysis of environmental impacts of cultivation and combustion of two perennial crops. Bars show the contribution of the processes to the overall emissions in each impact category.

environmental performance. However, a system expansion approach is necessary to reveal the net benefits and impacts of biomass utilization. Impact categories with a low ranking before a system expansion approach was applied, for example climate change and terrestrial acidification, show substantial benefits in avoided burdens after system expansion. In impact categories such as marine ecotoxicity and freshwater ecotoxicity, which have relatively high normalized values, the substitution of fossil fuels leads to additional impacts on the environment. If the normalized values alone are analysed, without a comparison to a fossil reference, the benefits of the utilization of perennial crops are substantially undervalued.

Choice of relevant impact categories

The hypothesis of this study was that a holistic assessment of the environmental performance needs to include other impact categories than just global warming potential (climate change). As presented in Fig. 6, the cultivation and utilization of the two analysed crops show no significant differences in the impact category climate change. In order to choose a biomass or utilization pathway on the basis of its environmental performance, it is also necessary to compare other impact categories. As shown in this study, the substitution of fossil fuel by miscanthus or willow chips leads to net

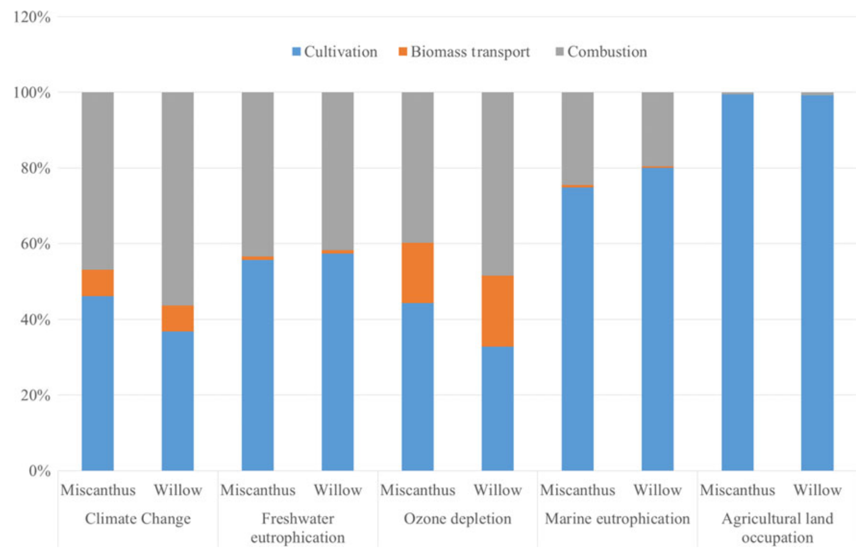

Fig. 9 Hot spot analysis of environmental impacts of cultivation and combustion of two perennial crops. Bars show the contribution of the processes to overall emissions in each impact category.

benefits in the impact category climate change. However, this substitution also leads to additional impacts on the environment in other categories. If climate change alone is assessed, other substantial environmental burdens are ignored. This again emphasizes the need to include more impact categories.

In order to assess the environmental performance of the cultivation and utilization holistically, the impact categories that show substantial benefits and those that show strong negative impacts on the environment need to be included. While there are only small differences between the two crops in the impact categories with net benefits (fossil fuel depletion, climate change, terrestrial acidification), there are substantial differences in the categories with the strongest impact (e.g. human toxicity, freshwater ecotoxicity). This is mainly due to differences in the heavy metal uptake (both in the amount and in the kind of heavy metal) of the crops and differences in emissions associated with the combustion process. This emphasizes the difficulties in preselecting impact categories and the need to analyse several impact categories when assessing the environmental performance of perennial crop-based value chains.

The assessment of the environmental impact of the cultivation and utilization of perennial crops carries a risk of double-counting emissions. For example, particulate matter formation has a strong impact on human toxicity and there is an overlap between mineral resource depletion and fossil fuel depletion. Nevertheless, as shown in Fig. 6, the normalized results for human toxicity and particulate formation can differ substantially. For this reason, both impact categories should be included, despite the double counting. However, the correlation between them should be clearly stated and integrated in the evaluation of their respective relevance.

The normalized results, however, are not necessarily the sole indicator in the assessment of impact categories' relevance. The normalization does not, for example, include social preferences or specific perspectives of the company commissioning the study. Another important point not included in the normalization is the preload of the specific environment. For example, at a site where the initial acidification is low and the buffer capacity of the soils is high, terrestrial acidification might not be the most urgent issue. The selection of impact categories is thus always dependent on the specific conditions and the questions to be answered by the study. Nevertheless, normalization of the life cycle impact assessment results is a crucial step in the assessment and comparison of the magnitude of different impact categories for biomass production and utilization.

Uncertainties in assessing the environmental performance of the cultivation and utilization perennial crops

1. Yield. One important influence on the environmental performance of perennial crops is the yield. With increasing yields, the environmental impact per tonne biomass is decreasing if the input of fertilizers and pesticides remains the same. There are only few field data for miscanthus yield performance over a ten-year or longer period. Those reports on long-term yields which are available indicate an entire range of developments: from stable yields over long periods;

through year-to-year variations; to yield decreases after an early peak (Gauder *et al.*, 2012; Iqbal *et al.*, 2015). More long-term field trials at different locations are necessary to get a reliable data basis. In the present study, the uncertainty caused by the yield was reduced through the availability of yield data for both crops under very similar conditions from field trials over a 10-year period.

An important aspect in environmental impact assessment of perennial energy crops is the accounting for yield variations over plantation time and the length of the productive period. Low yields in the first years and the fact that woody perennials only are harvested every third year are also an important reason why the whole cultivation period of perennial crops should be considered instead of only one year. The first harvest of miscanthus is in the second year and for willow only feasible from the fourth year onwards. Therefore, the environmental impact of the establishment period is broken down on the subsequent years. If the cultivation period is shorter, the total impact of the establishment on the environmental performance of the harvested biomass is increasing.

Besides that, there are uncertainties regarding the influence of nitrogen fertilizer in the yield development of miscanthus. In the field trial, which provided the underlying data for this study, there were significant differences in the yield of miscanthus between the three nitrogen fertilizer levels (Iqbal *et al.*, 2015). Similar results were found in field trials with miscanthus in the US Midwest, where the yield increased significantly with nitrogen fertilization (Arundale *et al.*, 2014). In contrast, multiannual trials in England showed no significant yield differences under different nitrogen fertilization levels (Christian *et al.*, 2008). If it would be possible to maintain high yields while decreasing the inputs of mineral fertilizers, it would improve the environmental performance significantly. On a location with good soils with a high nitrogen content and a high nitrogen deposition rate, it is reasonable to assume that no nitrogen fertilizer is applied. On the other hand, on poor sites the nitrogen fertilizer use should be included. Therefore, a recommended approach for LCA in perennial energy crops is to calculate with fertilization levels that are equivalent to the withdrawal of nutrients by the biomass.

2. Emission factors and calculation models. The hot spot analysis in this study revealed that the emissions associated with the use of mineral fertilizer – especially nitrogen fertilizers – have a huge impact on the environmental performance of the cultivation stage. The nitrate emissions, for example, are a main driver for the marine eutrophication potential. However, recent studies show that the nitrate leaching under perennial plants is much lower than under annual crops (Lesur *et al.*, 2014; Pugesgaard *et al.*, 2015). These results suggest that the data for nitrate leaching used in this study – which were calculated with a common agricultural model for nitrate leaching – are probably higher than actually experienced in perennial energy crop production. This emphasizes the need for emission models, which are adapted to the distinctive features of perennial crops.
3. CO_2 sequestration. In the last years, there were several papers published which highlighted the potential of miscanthus to sequester carbon in the soil (Kahle *et al.*, 2001; Clifton-Brown *et al.*, 2007; Brandão *et al.*, 2011; Felten & Emmerling, 2012). However, there are still huge uncertainties regarding the amount of CO_2, which will be sequestered, and the time frame of the sequestration (Harris *et al.*, 2015). Due to these uncertainties, the sequestration of carbon in the soil through the cultivation of perennial plants was not included in this study. Other LCA studies, which included the sequestration, showed that the cultivation of miscanthus could, under certain conditions, act as a real carbon sink (Brandão *et al.*, 2011; Godard *et al.*, 2013). Even if the carbon is only sequestered for the cultivation process and released again after the recultivation of the site, there still is a positive environmental impact in perennial crop production, which should be accounted for through the GWP based on a 20-year horizon (20-year GWP).
4. Missing impact categories. Miscanthus cultivation has a positive impact on the biodiversity with more weed vegetation and open-ground bird species (Semere & Slater, 2007a) and on the abundance of invertebrate populations (Semere & Slater, 2007b). The prolonged fallow period of perennial crops improves the soil quality, and the soil cover over winter reduces the erosion. However, it is not possible yet to include these positive effects of perennial crops in a LCA study. Therefore, approaches should be further developed for including these equally important environmental impacts into a holistic impact assessment. There are already some approaches to include these impacts. Oberholzer *et al.* (2012), for example, developed a method to include the impact of agricultural practice on soil quality in LCA. However, to date, this has only occasionally been used in LCA studies due to its complexity and huge data requirements. There are also approaches to integrate biodiversity aspects in LCA. Finnan *et al.* (2012) included a biodiversity indicator for miscanthus in their assessment of the environmental impacts of bioenergy plans. However, there

are still several shortcomings in the biodiversity indicators presently available for LCA (Souza *et al.*, 2015). Therefore, further research is necessary to allow a realistic assessment of the impact of agriculture or land use in general on biodiversity.

5. Indirect land-use change. While there are many positive effects of perennial biomass crops on the environment, the expansion of their cultivation area still bears a risk. If their cultivation is not restricted to marginal or unused land, an increase in their production area can lead to food production displacement. These food crops then need to be produced elsewhere. This indirect land-use change can lead to substantial negative impacts on the environment.
6. Utilization. As shown in the hot spot analysis, the emissions associated with the combustion process have a substantial impact on the different categories. While the data basis for the combustion of willow chips is adequate, there is insufficient information available on emissions from the combustion of miscanthus. In this study, a straw combustion process was taken as a worst-case assumption. In practice, the combustion of miscanthus would produce less emissions than shown here and have a lower impact on the environment. A reason for that is the higher chloride and sulphur content of straw in comparison with miscanthus biomass. These elements lead to harmful emissions in the combustion process (Spliethoff & Hein, 1998; Iqbal & Lewandowski, 2014)

A holistic environmental impact assessment of perennial biomass crops requires additional impact categories other than climate change (GWP). The GWP is a basic impact category due to the importance of climate change and the fact that the reduction in GHG emissions is one of the positive environmental contributions of perennial crop production. Based on the results of this study, an assessment of the environmental performance of the cultivation and combustion of perennial crops should also include the impact categories fossil fuel depletion, terrestrial acidification, freshwater ecotoxicity and human toxicity. However, the results of the study also show that the relevance of impact categories can differ depending on the crop and the utilization pathway. In order to resolve this issue, the choice of the relevant impact categories should be an iterative process. The first step is to analyse the relevance of the different impacts categories for the respective study goals and boundaries using initial data, a normalization step and a system expansion approach. After the determination of the relevant impact categories, the quality of the data important for these categories can be improved and the goal and scope adapted if necessary.

The choice of impact categories, emission models and data basis should also consider the special features of perennial crops in order to be able to assess the environmental benefits of their production. These include, for example, reduction in nitrate leaching, soil carbon sequestration and maintenance of biodiversity. Therefore, it is recommended that impacts on soil quality and biodiversity are included in future environmental impact studies and methodologies for integrating them into LCA are developed. In addition, nitrogen emission models currently available need to be adapted to actual data of perennial crop performance.

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Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Data S1. Fertilizer induced emissions and heavy metal soil balance.
Data S2. Ash characteristics.

3. Environmental impacts and benefits of perennial crop-based value chains

3.1 Novel miscanthus germplasm-based value chains: A Life-Cycle Assessment

In this sub-chapter the environmental performance of miscanthus biomass grown on six different locations across Europe and the subsequent utilization in six different pathways was assessed in eighteen impact categories. The six utilization pathways were: 1) small-scale combustion (heat) – chips; 2) small-scale combustion (heat) – pellets; 3) large-scale combustion (CHP) – biomass baled for transport and storage; 4) large-scale combustion (CHP) – pellets; 5) medium-scale biogas plant – ensiled miscanthus biomass; and 6) large-scale production of insulation material. The different pathways were compared with respect to their impact on the environment, their environmental hot-spots and the relevant impact categories. As two from the six sites were classified as marginal, the findings of this study also addresses the issue raised in research question four regarding the environmental performance of miscanthus cultivated on marginal land.

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Novel Miscanthus Germplasm-Based Value Chains: A Life Cycle Assessment

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In recent years, considerable progress has been made in miscanthus research: improvement of management practices, breeding of new genotypes, especially for marginal conditions, and development of novel utilization options. The purpose of the current study was a holistic analysis of the environmental performance of such novel miscanthus-based value chains. In addition, the relevance of the analyzed environmental impact categories was assessed. A Life Cycle Assessment was conducted to analyse the environmental performance of the miscanthus-based value chains in 18 impact categories. In order to include the substitution of a reference product, a system expansion approach was used. In addition, a normalization step was applied. This allowed the relevance of these impact categories to be evaluated for each utilization pathway. The miscanthus was cultivated on six sites in Europe (Aberystwyth, Adana, Moscow, Potash, Stuttgart and Wageningen) and the biomass was utilized in the following six pathways: (1) small-scale combustion (heat)—chips; (2) small-scale combustion (heat)—pellets; (3) large-scale combustion (CHP)—biomass baled for transport and storage; (4) large-scale combustion (CHP)—pellets; (5) medium-scale biogas plant—ensiled miscanthus biomass; and (6) large-scale production of insulation material. Thus, in total, the environmental performance of 36 site × pathway combinations was assessed. The comparatively high normalized results of human toxicity, marine, and freshwater ecotoxicity, and freshwater eutrophication indicate the relevance of these impact categories in the assessment of miscanthus-based value chains. Differences between the six sites can almost entirely be attributed to variations in biomass yield. However, the environmental performance of the utilization pathways analyzed varied widely. The largest differences were shown for freshwater and marine ecotoxicity, and freshwater eutrophication. The production of insulation material had the lowest impact on the environment, with net benefits in all impact categories expect three (marine eutrophication, human toxicity, agricultural land occupation). This performance can be explained by the multiple use of the biomass, first as material and subsequently as an energy carrier, and by the substitution of an emission-intensive reference product. The results of this study emphasize the importance of assessing all environmental impacts when selecting appropriate utilization pathways.

Keywords: miscanthus, biobased value chains, LCA, environmental performance, normalization, impact categories

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INTRODUCTION

The developing European bioeconomy will lead to an increasing demand for sustainably produced biomass in the near future. Miscanthus is one of the leading candidate biomass crops and has the advantage that it can also grow under marginal site conditions (Lewandowski et al., 2016). It is a perennial rhizomatous C4 grass originating from Southeast Asia, where it shows large genetic diversity. Miscanthus was introduced into Europe in 1935, where the genotype *Miscanthus × giganteus* is predominately cultivated (Clifton-Brown et al., 2015). It is a resource-efficient, low-input crop, which can achieve yields of well above 20 Mg ha^{-1} a^{-1} (dry matter) in Central Europe (Lewandowski and Schmidt, 2006; Iqbal et al., 2015) and more than 30 Mg ha^{-1} a^{-1} (dry matter) in southern Europe under irrigated conditions (Lewandowski et al., 2000). As a perennial crop, miscanthus can be harvested over a 15–20-year cultivation period (Lewandowski et al., 2000; Christian et al., 2008). Due to its perennial nature and its high nitrogen- and water-use efficiency, miscanthus has a comparatively low impact on the environment as a biomass crop (Lewandowski et al., 2000; Voigt, 2015; McCalmont et al., 2017).

Miscanthus biomass can be used in several different utilization pathways. When harvested green in the period September to October, it can be used as a biogas substrate (Whittaker et al., 2016; Kiesel and Lewandowski, 2017). When harvested in early spring, it is suitable for combustion (Dahl and Obernberger, 2004; Iqbal and Lewandowski, 2014), as a late harvest leads to a lower water and mineral content (Lewandowski et al., 2000). In addition, miscanthus biomass can be fermented to ethanol (van der Weijde et al., 2016) or used as a raw material for the production of insulation material (Uihlein et al., 2008) or bio-composites (Muthuraj et al., 2015).

However, despite these diverse potential applications, there is currently low implementation of miscanthus cultivation as several major barriers hinder its utilization in practice (Clifton-Brown et al., 2016). To overcome these barriers, considerable efforts have been made in the last years in (a) development of new genotypes, tailored to different, especially marginal, site conditions in Europe, and different biomass uses; (b) the optimization of miscanthus management (Clifton-Brown et al., 2016; Lewandowski et al., 2016).

The objective of this study is to assess the environmental performance of various miscanthus-based energetic and material value chains using the most up-to-date genotype as well as management options. Most previous studies used cultivation and yield data from the standard genotype *Miscanthus × giganteus* to analyse environmental performance. However, as explained above, in the last years there have been substantial efforts especially in the breeding of new genotypes. The inclusion of this progress in the current study will allow a more realistic assessment of the environmental impact and mitigation possibilities of miscanthus-based value chains.

Several studies have already evaluated the environmental performance of miscanthus-based value chains in different impact categories. These studies encompass the utilization of miscanthus as a biogas substrate (Kiesel et al., 2016), for electricity generation (Sanscartier et al., 2014), as feedstock for bioethanol (Jeswani et al., 2015), and as fuel for heat generation (Wagner and Lewandowski, 2017). However, most of these studies examine only one single utilization pathway or assess only a few impact categories (Meyer et al., 2016).

The various assumptions, system boundaries and methodologies used in these studies makes a comparison of the results very difficult. Therefore, the second objective of the current study is to assess the environmental sustainability of different miscanthus utilization pathways in several impact categories under the same assumptions and underlying conditions. This is done in order to enable the comparison of the environmental performance of different miscanthus-based value chains.

For this purpose, an attributional Life Cycle Assessment (LCA) was conducted according to the ISO standards 14040 and 14044 (ISO, 2006a,b). The energetic and material utilization pathways assessed in this study are: (1) small-scale combustion (heat)—chips; (2) small-scale combustion (heat)—pellets; (3) large-scale combustion (CHP)—biomass baled for transport and storage; (4) large-scale combustion (CHP)—pellets; (5) medium-scale biogas plant—biomass ensiled; and (6) large-scale production of insulation material—biomass baled for transport and storage. These pathways were assessed for miscanthus biomass cultivated from different genotypes on six climatically different sites across Europe: Aberystwyth (UK), Adana (Turkey), Moscow (Russia), Potash (Ukraine), Stuttgart (Germany), and Wageningen (Netherlands). Data for the cultivation of the biomass were provided through the EU-funded research project OPTIMISC (Optimizing Miscanthus Biomass Production) (Lewandowski et al., 2016). The environmental performance of each of the six utilization pathways was assessed for each site in 18 impact categories using the life-cycle impact assessment methodology ReCiPe (Goedkoop et al., 2008). To assess the mitigation potential of the analyzed pathways in the different impact categories, a system expansion approach was chosen. This approach enabled the assessment of the net benefits and impacts of the different pathways on the environment through the substitution of a chiefly fossil-based reference product with a miscanthus-based one.

In addition, a normalization step was applied. This allows the relevance of the analyzed impact categories for each utilization pathway to be assessed (Wagner and Lewandowski, 2017). The normalization factors used in this study were taken from the ReCiPe methodology (Goedkoop et al., 2008).

MATERIALS AND METHODS

Scope and Boundaries

The scope of this study is a cradle-to-grave analysis of the environmental performance of miscanthus cultivation at six sites in Europe and the subsequent utilization in six pathways. In total, 36 site × pathway combinations were assessed. In order to include the substitution of a reference product, a system expansion approach was applied. This allows the impact of the substitution of a reference product (e.g., heat produced by the combustion of natural gas) through the utilization of 1 ha miscanthus (e.g., heat produced by the combustion of miscanthus chips) to be included in the assessment for each value chain. Thus, negative values represent burdens avoided by such a substitution, while positive values represent an additional impact through the use of miscanthus biomass. This is the case when the production and utilization of the reference products emits less than the substituting miscanthus-based product.

The functional unit (FU) as well as main and co-products for the six utilization pathways are shown in **Table 1**. In addition, for each product, the substituted reference product is indicated. One hectare was chosen as functional unit to assess the annual net benefit or impact of substituting a reference product by the energetic or material utilization of miscanthus. On the cultivation sites Aberystwyth (UK), Moscow (Russia), Potash (Ukraine), Stuttgart (Germany), and Wageningen (Netherlands), the genotype OPM-06 was used, a *M. sinensis* × *M. sacchariflorus* hybrid. On the Adana site in Turkey, the genotype *M* × *giganteus* (OPM-09) was used. These two were preselected from 15 assessed genotypes, because they were the most suitable for the location and utilization pathway in terms of biomass quality and yield. The data on the cultivation process and choice of genotypes are based on multi-location field trials described in Lewandowski et al. (2016). The sites in Adana, Potash, Stuttgart and Wageningen are mostly on land previously used as agricultural land, whereas the sites in Aberystwyth and Moscow are on marginal land. In Aberystwyth, the miscanthus was cultivated on land which was previously low-quality grassland. At the Moscow site, harsh winters lead to non-ideal growing conditions (Lewandowski et al., 2016).

The agricultural system is described in **Figure 1**. The system boundaries include the production of input substrates (e.g., fertilizers, propagation material) and the whole cultivation process (from soil preparation through planting and establishment to harvest over a twenty-year cultivation period) to subsequent recultivation. For all utilization pathways, the miscanthus is mulched in the first year and harvested from the second year onwards. In pathways 2, 3, 4, and 6, it is mowed and then pressed into bales; in 1 and 5 it is harvested with a self-propelled forage harvester in the form of chips. For the combustion pathways 2 and 4, the miscanthus bales are then further processed to pellets.

The utilization pathways 1 to 5 are shown in **Figure 2**. In all four combustion pathways (1, 2, 3, and 4), the handling of the ash is the same. It is assumed that both the fly and bottom ash is disposed of in landfill. The fly ash in particular has high levels of heavy metals. In utilization pathway 1, the miscanthus biomass is used on-farm in a small combustion unit to generate heat. In utilization pathway 2, miscanthus biomass in the form of pellets instead of chips is utilized in a small combustion unit to generate heat. The reference product of the utilization pathways 1 and 2 is heat produced by combustion of light fuel oil. This reference product was chosen, because it is produced in a comparable small-scale combustion unit. A sensitivity analysis was performed with heat produced by combustion of natural gas as a reference product to analyse the impact of this assumption.

In utilization pathway 3, miscanthus bales are combusted in a combined heat and power unit (CHP) to generate heat, with electricity as a co-product. In pathway 4, miscanthus pellets are utilized in the CHP instead of bales. Heat was specified as the main and electricity as the co-product in accordance with the description in the ecoinvent database (Weidema et al., 2013). The electricity produced is assumed to substitute the European electricity mix. The heat generated substitutes heat produced by the combustion of natural gas in a CHP. Natural gas was chosen in this case as a reference product, because it is a relative clean energy source (May and Brennan, 2006). This assumption reduces the risk of overestimating the net environmental benefit of the miscanthus-based alternative.

Utilization pathway 5 includes the fermentation of green-harvested miscanthus biomass to biogas and subsequent combustion to generate electricity, with heat as a co-product. Electricity was selected as main product in accordance with Bacenetti et al. (2016) and the European electricity mix was chosen as reference product. The heat generated as co-product substitutes heat produced by the combustion of natural gas in a CHP. The residues of the fermentation process are rich in

TABLE 1 | Utilization pathways assessed in this study, the functional unit, their outputs and the reference products.

No.	Utilization pathway	Biomass used	FU	Output	Main product	Co-product	Reference product
1	Small-scale combustion	Chips	1 ha	Heat	*		Heat produced by combustion of light fuel oil
2	Small-scale combustion	Pellets	1 ha	Heat	*		Heat produced by combustion of light fuel oil
3	Large-scale combustion (CHP)	Bales	1 ha	Heat	*		Heat produced by combustion of natural gas in a CHP
				Electricity		*	European electricity mix
4	Large-scale combustion (CHP)	Pellets	1 ha	Heat	*		Heat produced by combustion of natural gas in a CHP
				Electricity		*	European electricity mix
5	Biogas plant	Silage	1 ha	Electricity	*		European electricity mix
				Heat		*	Heat produced by combustion of natural gas in a CHP
6	Production of insulation material	Bales	1 ha	Insulation material	*		Glass wool

**Indicates if the product is the main- or the co-product.*

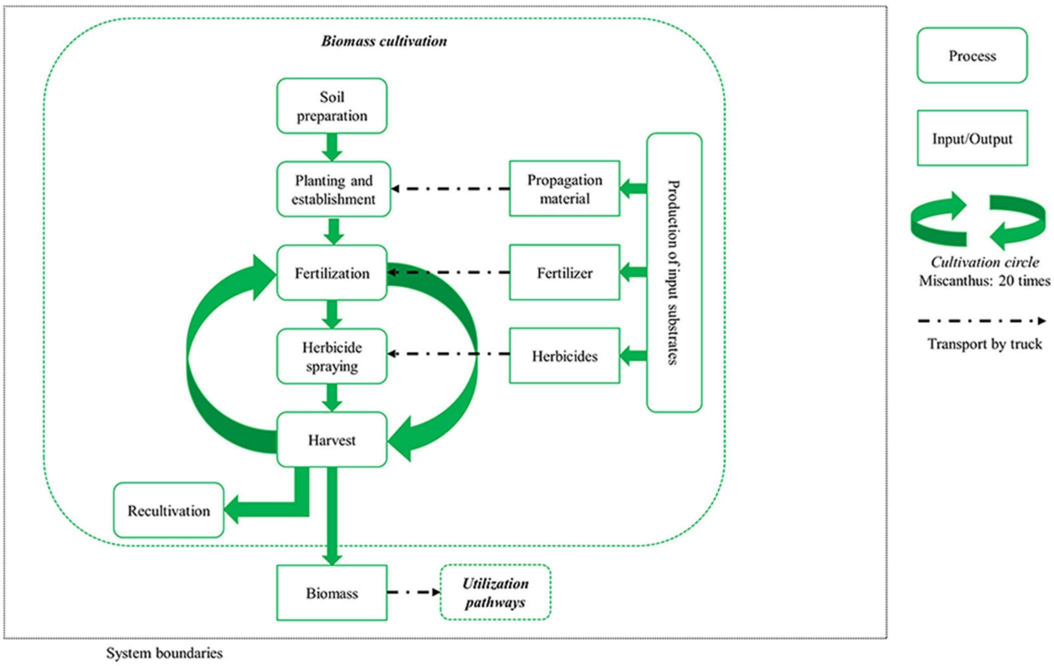

FIGURE 1 | System description and boundaries for miscanthus biomass cultivation.

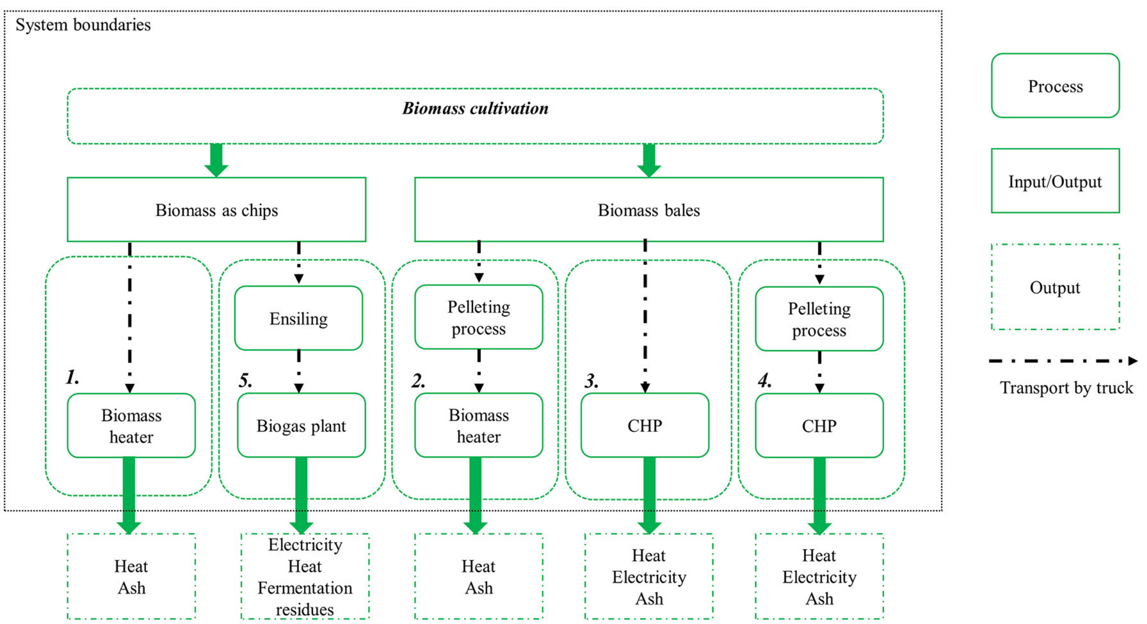

FIGURE 2 | System description and boundaries for the energetic utilization pathways 1–5.

nutrients (see Table S1) and can be used to substitute mineral fertilizer.

Utilization pathway 6, which is displayed in **Figure 3**, is the production of insulation material from miscanthus biomass. The miscanthus fibers are separated via steam explosion, dried, and mixed with additives. Insulation material is then produced through hot pressing. The reference product for 1 m^3 miscanthus-based insulation material is 110 kg glass wool mats with comparable characteristics (Meyer et al., 2016). The End-of-Life of the miscanthus- and the fossil-based pathways are

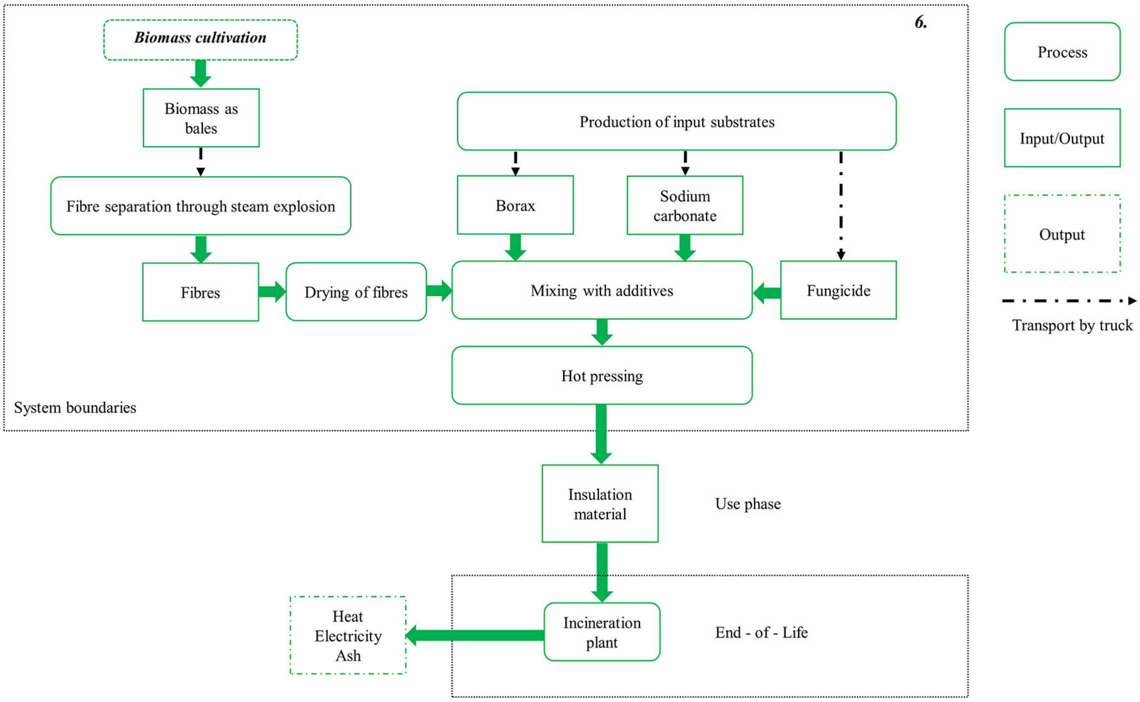

FIGURE 3 | System description and boundaries for the material utilization pathway 6.

included in the assessment. The glass wool is treated as inert waste and disposed of to landfill. After its use phase, it is assumed that the miscanthus-based insulation material is incinerated, generating heat and electricity (see **Figure 3**). The electrical and thermal efficiencies of the incineration plant are comparable to the CHP plant used in the utilization pathways 3 and 4.

Life Cycle Inventory

Agricultural System

The data used in this Life Cycle Assessment for the cultivation phase of miscanthus were obtained from multi-location field trials conducted within the OPTIMISC project (Lewandowski et al., 2016). **Table 2** shows the main inputs and outputs at the different sites for the pathways using biomass harvested in spring (combustion, production of insulation material), or autumn (biogas substrate). Field data for pathway 5 was only available for the Adana, Moscow and Stuttgart sites (see **Table 2**).

In addition to the inputs shown in **Table 2**, the field trials in Adana were irrigated with 976.75 m^3 water per hectare and year, independent of harvest date.

Nitrogen was applied as calcium ammonium nitrate, potassium as potassium sulfate and phosphate as triple superphosphate. Herbicides are only necessary in miscanthus cultivation in the preparation of the sites, in the first two cultivation years, when miscanthus is unable to compete with weeds, and in the recultivation process. Over the twenty-year cultivation period, a total application of 16.2 l herbicides ha^{-1} were applied: 10 l ha^{-1} Round up (Monsanto, active ingredient 360 g l^{-1} glyphosate); 3.5 l ha^{-1} Stomp Aqua (BASF, active ingredient 455 g l^{-1} pendimethalin); 1.5 l ha^{-1} Calisto (Syngenta, active ingredient 100 g l^{-1} mesotrione); 0.2 l ha^{-1} Arrat (BASF, active ingredient 100 g l^{-1} tritosulfuron and 500 g l^{-1} dicamba); and 1 l ha^{-1} Dash, (BASF, an emulsifiable concentrate). This corresponds to an average of 0.81 l or 0.93 kg ha^{-1} yr^{-1} herbicides.

The yield data in **Table 2** is shown per year. However, these yield data are based on the whole cultivation period including the establishment phase. In the first year, the biomass is not harvested but mulched, and the full yield is only achieved from the third year onwards (Lewandowski et al., 2003). The calculation for the early spring harvest is given in Equation 1 and for the autumn harvest in Equation 2.

$$\textit{Mean yield spring}\ \left[t\ DM\ ha^{-1}yr^{-1}\right] = \frac{yield\ (2.\ year\_spring\ +\ 3.year\_spring*18)}{20} \tag{1}$$

$$\textit{Mean yield autumn}\ \left[t\ DM\ ha^{-1}yr^{-1}\right] = \frac{yield\ (2.\ year\_autumn\ +\ 3.year\_autumn*18)}{20} \tag{2}$$

Table 3 shows the agricultural operations applied during miscanthus cultivation including frequency. These are shown for two harvest procedures: in the chopping line, the biomass is processed to chips to be used in the utilization pathways 1 and 5; and in the baling line, it is baled (utilization pathways 2, 3, 4, and 6).

The background data for the environmental impacts associated with the cultivation processes (e.g., plowing, mowing)

TABLE 2 | Summary of the main inputs and outputs of the spring and the autumn harvests.

Values in kg yr^{-1} ha^{-1}	Adana	Aberystwyth	Moscow	Potash	Stuttgart	Wageningen
			Harvest Feb./Mar.			
N	60	60	60	60	60	60
K_2O	120	120	120	120	120	120
P_2O_5	30	30	30	30	30	30
Herbicides	0.93	0.93	0.93	0.93	0.93	0.93
Dry matter yield	12,600	9,745	9,734	16,065	15,316	10,320
			Harvest Sept./Oct.			
N	140	n.a.	140	n.a.	140	n.a.
K_2O	200	n.a.	200	n.a.	200	n.a.
P_2O_5	30	n.a.	30	n.a.	30	n.a.
Herbicides	0.93	n.a.	0.93	n.a.	0.93	n.a.
Dry matter yield	19,365	n.a.	15,568	n.a.	23,624	n.a.

TABLE 3 | Agricultural operations applied during 20 years of miscanthus cultivation with frequency.

Agricultural operations	Frequency per cultivation period	
	Chopping line	**Baling line**
Rotary harrow	2	2
Plowing	1	1
Planting	1	1
Mulching—first year	1	1
Spraying	5	5
Fertilizing	19	19
Mowing	0	18
Swath	0	18
Chipping	18	0
Baling	0	18
Mulching—final year	1	1
Chisel plow	1	1

and the production of the input substrates were taken from the ecoinvent database version 3.3 (cut-off system model) (Weidema et al., 2013). The energy demands of the harvesting processes (chopping and baling) and the pelleting process are based on Hastings et al. (under review).

N_2O emissions from harvest residues and indirect N_2O emissions from nitrogen fertilizer were estimated using emission factors based on IPCC (2006). Direct N_2O and NO emissions from nitrogen fertilizer were calculated according to Bouwman et al. (2002). Ammonia emissions were calculated using emission factors from EMEP/CORINAIR (2001). Phosphate and phosphorus emissions to surface and groundwater, and heavy metal emissions to agricultural soil were estimated based on Nemecek and Kägi (2007). Nitrate leaching to groundwater was calculated according to the SQCB—NO_3 model described in Faist Emmenegger et al. (2009). All pesticide applied have been modeled completely as emission to agricultural soil in accordance to Nemecek and Schnetzer (2011). The ecotoxicity values of this emission are based on the ecoinvent database (Weidema et al., 2013).

Several recent publication have demonstrated the ability of miscanthus to sequester CO_2 in the soil through an increase in soil organic carbon, especially in comparison to annual plants (Gauder et al., 2016; McCalmont et al., 2017). However, these changes in soil organic carbon are highly dependent on the previous crop and thus contain a high degree of uncertainty (Harris et al., 2015). Because of this, carbon sequestration in the soil was not included this assessment.

Table 4 gives the farm-to-field distances and truck transport distances for the different utilization pathways. No data were available for the transport distances of input substrates (e.g., fertilizer) or propagation material. Therefore, a transport distance of 150 km for the input material by a EUR5 truck was assumed. The background data associated with the transportation of the input material and biomass were taken from the ecoinvent database (Weidema et al., 2013).

There are considerable differences in transport density between chips, bales and pellets. To account for these differences, the emission data from the ecoinvent database used for the transport process (Weidema et al., 2013) was adapted in accordance with Hastings et al. (under review).

Utilization Pathways

The following section describes the life cycle inventories for the different utilization pathways. The modeling of the pathways included the emissions associated with the construction of the conversion plants (e.g., CHP unit, biogas plant) and necessary infrastructure, based on background data from the ecoinvent database (Weidema et al., 2013).

The biomass heater used for utilization pathways 1 and 2 is a furnace with a heat generation capacity of 300 kW. The background data for the emissions associated with combustion is taken from the ecoinvent database. This data is based on a Froling Turbomat 320 kW woodchip boiler with a thermal efficiency of 75%. This is lower than in the technical specification, because it represents the average annual operation, which includes start and stop phases (Weidema et al., 2013).

The background emission data for utilization pathways 3 and 4 [combined heat and power unit (CHP)] are based on

TABLE 4 | Transport distances for the utilization pathways.

Process	Unit	Utilization pathways					
		(1)	(2)	(3)	(4)	(5)	(6)
Truck transport of input substrates	km	150	150	150	150	150	150
Farm-field distance	km	2	2	2	2	15	2
Truck transport of bales	km	–	100	400	100	–	400
Truck transport of pellets	km	–	400	–	400	–	–

TABLE 5 | Technical characteristics of the biogas plant used in the analysis.

Technical characteristics		Unit
Full load hours	7,800	H
Plant output electrical	500	kWh_{el}
Plant output total	1,351	kWh
Electrical efficiency	37	% of plant total output
Thermal efficiency	53	% of plant total output
Inherent heat demand	20	% of total heat production
Inherent power consumption	12	% of total power production

the ecoinvent process *"heat and power co-generation, wood chips, 6,667 kW, state-of-the-art 2014."* According to the process description in the ecoinvent database, an organic rankine cycle (ORC) steam generator with an electrical efficiency of 15% and a thermal efficiency of 45% is used (Weidema et al., 2013).

As there is insufficient specific information available on emissions from miscanthus combustion, all four utilization pathways are based on wood combustion processes. Miscanthus-specific emission factors for carbon monoxide, sulfur dioxide, hydrogen chloride, nitrogen oxides, and particulates were taken from Dahl and Obernberger (2004). At the time of harvest, miscanthus biomass has a water content of around 15% (Lewandowski et al., 2016). A further drying process is therefore not necessary. A mean calorific value of 4.3 kWh kg^{-1} fresh biomass was calculated based on the model of Jiménez and González (1991).

The miscanthus biomass used in the biogas plant is harvested in autumn and then ensiled. Dry matter losses of 12% were assumed during the ensilage process. The silage is subsequently fermented to biogas. The methane hectare yield [m^3 CH_4 yr^{-1} ha^{-1}] for the Adana site was 4,676, for the Moscow site 4,194, and for the Stuttgart site 6,495 (Kiesel et al., 2017). The methane yield was measured as described in Kiesel and Lewandowski (2017). A biogas batch test was performed for 35 days in mesophilic conditions (39°C) according to VDI guideline 4,630. The approach of the biogas batch test was certified by the KTBL and VDLUFA interlaboratory comparison test 2014 and 2015. Each sample was assessed in four technical replicates. Methane losses of 1% were assumed in the biogas plant based on Börjesson and Berglund (2007). The biogas is combusted in a CHP unit to generate heat and power. The electricity is fed into the grid. Twenty percent of the heat produced is used internally for the heating of the fermenter. In this study, it was assumed that 50% of the remaining heat (that is 40% of the total heat produced) is used to heat nearby residential buildings and so substitute heat produced from fossil sources. The other 50% of the remaining heat is not used and thus is excess heat that escapes into the atmosphere. The technical characteristics of the CHP used in this study are shown in **Table 5** (Uihlein et al., 2008). Both the emissions associated with biogas combustion in the CHP unit and the construction of the biogas plant are based on the ecoinvent database (Weidema et al., 2013).

To produce 1 m^3 of insulation material, 194.3 kg dry-matter miscanthus biomass is required. This corresponds to 228.6 kg fresh biomass at a moisture content of 15%. The additives consist of 3.85 kg borax, 3.85 kg sodium carbonate and 1.1 kg of the fungicide thiocarbamate (Velásquez et al., 2003). The energy required for the production process is shown in **Table 6**.

Choice of Impact Categories

The life cycle impact assessment methodology ReCiPe was used in this LCA study (Goedkoop et al., 2008). All 18 mid-point indicators described in this methodology were included: climate change (CC), which corresponds to global warming potential (GWP); ozone depletion (OD); terrestrial acidification (TA); freshwater eutrophication (FE); marine eutrophication (ME); human toxicity (HT); photochemical oxidant formation (POF); particulate matter formation (PMF); terrestrial ecotoxicity (TET); freshwater ecotoxicity (FET); marine ecotoxicity (MET); ionizing radiation (IR); agricultural land occupation (ALO); urban land occupation (ULO); natural land transformation (NLT); mineral resource depletion (MRD); fossil fuel depletion (FFD); and water depletion (WD). The results are shown as normalized values. This means, that the results of each impact category are divided by the respective emissions caused by an average European in the year 2000. The resulting values show the calculated impact as a proportion of the emissions of an average European citizen. The characterization and normalization factors are based on Goedkoop et al. (2008). No normalized values are given for the impact category "water depletion," as no normalization factor is available in the ReCiPe methodology for this impact category (Goedkoop et al., 2008).

RESULTS

The results are presented as normalized values. These show the net benefits and impacts of the utilization of 1 ha miscanthus for all six sites and for all six utilization pathways (see **Figures 4–9**). The absolute values per ha for all utilization pathways on all sites analyzed are given in the Supplementary Material (Tables S2–S7). In addition, they are shown per MJ_{th} for the utilization pathways 1, 2, 3, and 4 (Tables S2–S5), in MJ_{el} for utilization pathway 5 (Table S6) and in m^3 insulation material for utilization pathway 6 (Table S7).

The normalized net benefits and impacts per ha in the impact categories TA, FE, and ME, MRD and FFD, and CC are shown in **Figure 4** for the sites Adana, Stuttgart and Moscow and in **Figure 5** for the sites Aberystwyth, Potash, and Wageningen. Utilization pathway 6 (production of insulation material) has the largest net benefits in the categories TA, FE, MRD, and CC on all

TABLE 6 | Energy consumption for the production of miscanthus-based insulation material.

Energy consumption	Unit	Per kg dry-matter miscanthus biomass	Per m^3 insulation material
Steam explosion	MJ_{th}	1.452	282.085
	MJ_{el}	0.073	14.104
Drying of fibers	MJ_{th}	1.493	290.111
	MJ_{el}	0.075	14.506
Mixing and hot pressing	MJ_{th}	0.824	160.103
	MJ_{el}	0.042	8.161
Total	MJ_{th}	3.769	732.299
	Mj_{el}	0.19	36.771

sites. This is due to the substitution of the reference product glass wool, which has a very emission-intensive production process. All utilization pathways perform negatively in the category ME. This is largely caused by nitrogen-fertilizer-induced nitrate emissions in the miscanthus cultivation process. Utilization pathways 1 and 2 (both small-scale combustion) also have a negative impact in FE, which is mainly caused by phosphate-fertilizer-induced emissions. The production process of the reference product of utilization pathways 1 and 2 (heat generated through the combustion of light fuel oil) has a low FE. For this reason, the substitution caused a net negative impact on the environment in this category. Differences between the utilization pathways 1 and 2, as well as 3 and 4 are due to differences in transport distance and the additional pelleting process. As a result, pathway 1 has lower environmental impacts than pathway 2, and pathway 3 lower environmental impacts than pathway 4. This applies to all impact categories.

The normalized net benefits and impacts per ha in the impact categories PMF, HT, MET, FET, and TET for the sites Adana, Stuttgart and Moscow are shown in **Figure 6**, and for the sites Aberystwyth, Potash and Wageningen in **Figure 7**. The utilization pathway 5 (medium-scale biogas plant) had relatively high environmental benefits in HT, MET, and FET (see **Figure 6**). These can be explained by the emission-intensive production process of the substituted reference product, the European electricity mix. Utilization pathway 6 showed low environmental impacts in the category PMF compared with the other utilization options. This is due to the high impact of the substituted reference product glass wool in this impact category, in particular its production process. All other utilization pathways had a comparatively negative performance in all impact categories depicted in **Figures 6**, **7**. The net impacts in ME, FE and especially HT in the utilization pathways 1 to 4 result from the treatment of the bottom and fly ash, which incur in the combustion process.

The normalized net benefits and impacts per ha in the impact categories IR, POF, OD, ALO, and ULO are shown in **Figure 8** for the sites Adana, Stuttgart, and Moscow, and in **Figure 9** for the sites Aberystwyth, Potash, and Wageningen. Naturally, all biomass-based utilization pathways perform negatively in the category ALO. Utilization pathway 6 shows a comparatively large net benefit in the category POF. This is again caused by the substitution of the reference product. The net benefit of utilization pathways 1 and 2 in the category OD result from the emission-intensive generation of the reference product (heat generated by the combustion of light fuel oil). All utilization pathways had a comparatively large net benefit in the impact category natural land transformation (data not shown). The normalized results range from −6.15 for utilization pathway 5, to −42.86 for utilization pathway 1. In all utilization pathways, this is caused by the substituted reference products, which have a strong negative impact in this category. For clarity of presentation, these results are not included in **Figures 8**, **9** due to their considerably higher values.

DISCUSSION

The first part of the discussion focuses on the normalized values shown in **Figures 4–9**, including a critical reflection on the influence on the final results of reference product selection and credits given for co-products. In addition, the impact of the End-of-Life phase of the products is elaborated. The second part discusses the relevance of the impact categories for the various utilization pathways analyzed in this study. The final part gives recommendations for improving the environmental performance of the biobased value chains and considers the implications of the results for future biomass use.

Determinants of Environmental Benefits and Impacts

Figures 4–9 show the normalized values for the environmental benefits and impacts per hectare (including the cultivation of the biomass and subsequent utilization) minus the substitution of a reference product and the credits given for co-products.

A comparison of the normalized results from this study with results from reference literature is only partially possible due to different assumptions, system boundaries and methodologies used. Wagner and Lewandowski (2017) analyzed the relevance of various impact categories for a small-scale combustion chain using miscanthus and willow cultivated under three nitrogen fertilizer regimes. The results of their study show strong similarities with those of the current assessment, in particular with regard to the question of which impact categories are relevant and which not.

In general, the utilization pathways 5 (fermentation of miscanthus in a biogas plant and subsequent utilization in a CHP) and 6 (production of insulation material) had the lowest impacts on the environment. They had considerably larger net benefits, especially in the impact categories MET and FET, and FE. The results of the small-scale combustion chains again emphasized the necessity of including more impact categories than just climate change when analyzing and comparing the environmental performance of biobased utilization pathways (Jeswani et al., 2015; Wagner and Lewandowski, 2017). The small-scale combustion chains had advantages in the impact categories OD and FFD, and achieved the highest climate change saving potential of all energetic value chains (1, 2, 3, 4,

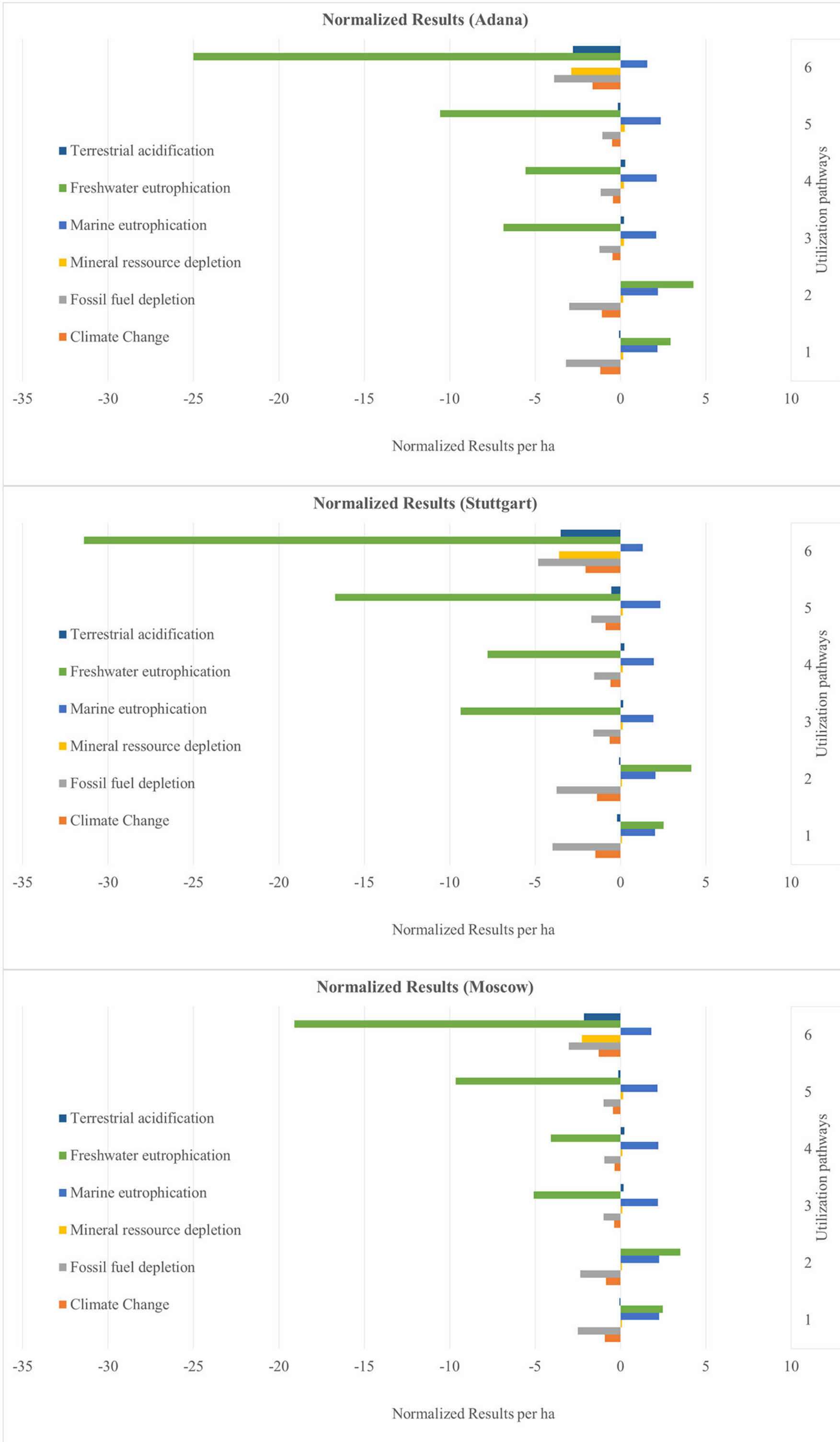

FIGURE 4 | Normalized results per ha for the sites Adana, Stuttgart, and Moscow—Part 1. Utilization pathways: 1. Small-scale combustion—chips; 2. Small-scale combustion—pellets; 3. Large-scale combustion—biomass baled for transport and storage; 4. Large-scale combustion—pellets; 5. Medium-scale biogas plant—biomass ensiled; and 6. Large-scale production of insulation material—biomass baled for transport and storage.

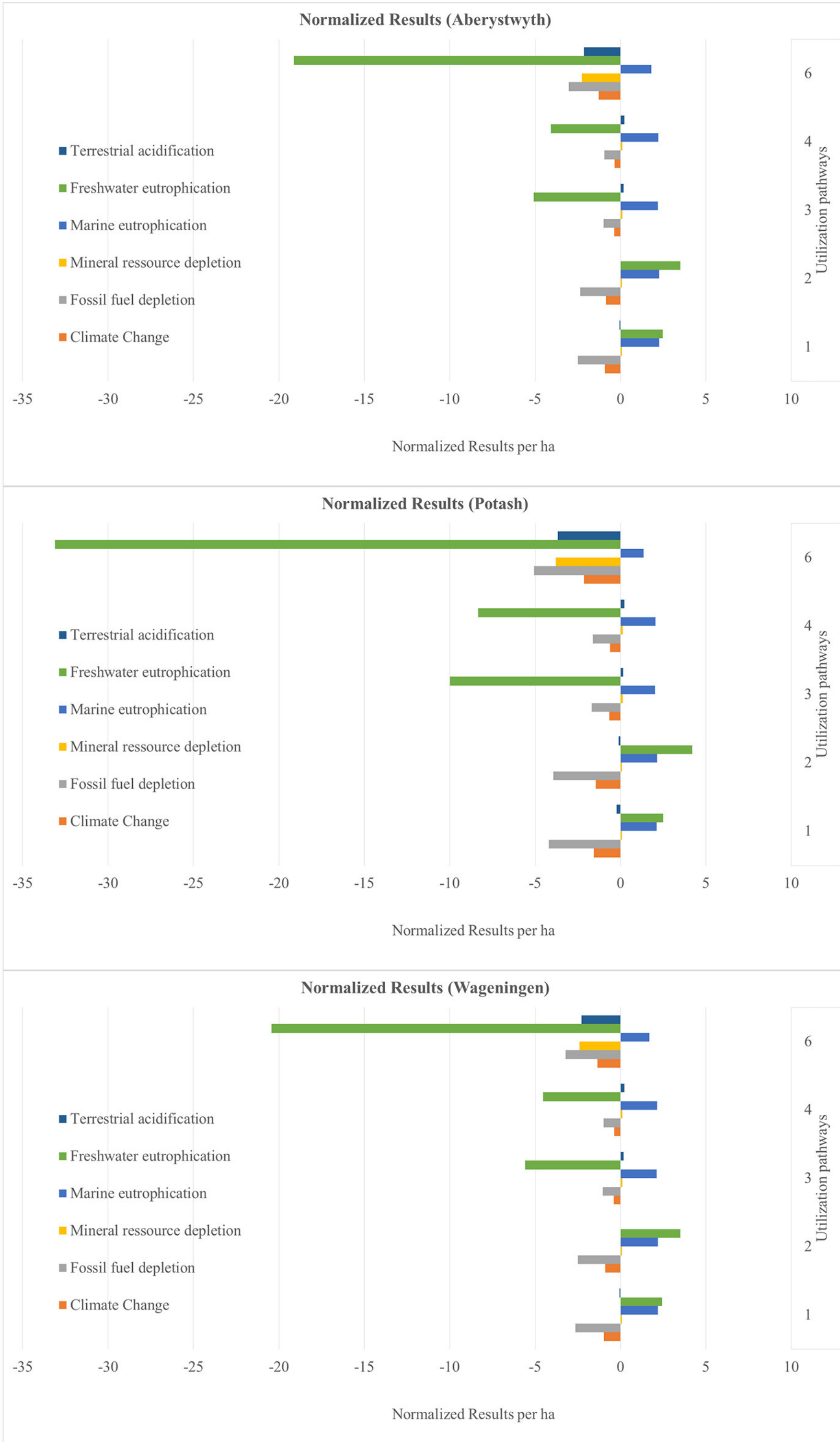

FIGURE 5 | Normalized results per ha for the sites Aberystwyth, Potash, and Wageningen–Part 1. Utilization pathways: 1. Small-scale combustion—chips; 2. Small-scale combustion—pellets; 3. Large-scale combustion—biomass baled for transport and storage; 4. Large-scale combustion—pellets; and 6. Large-scale production of insulation material—biomass baled for transport and storage.

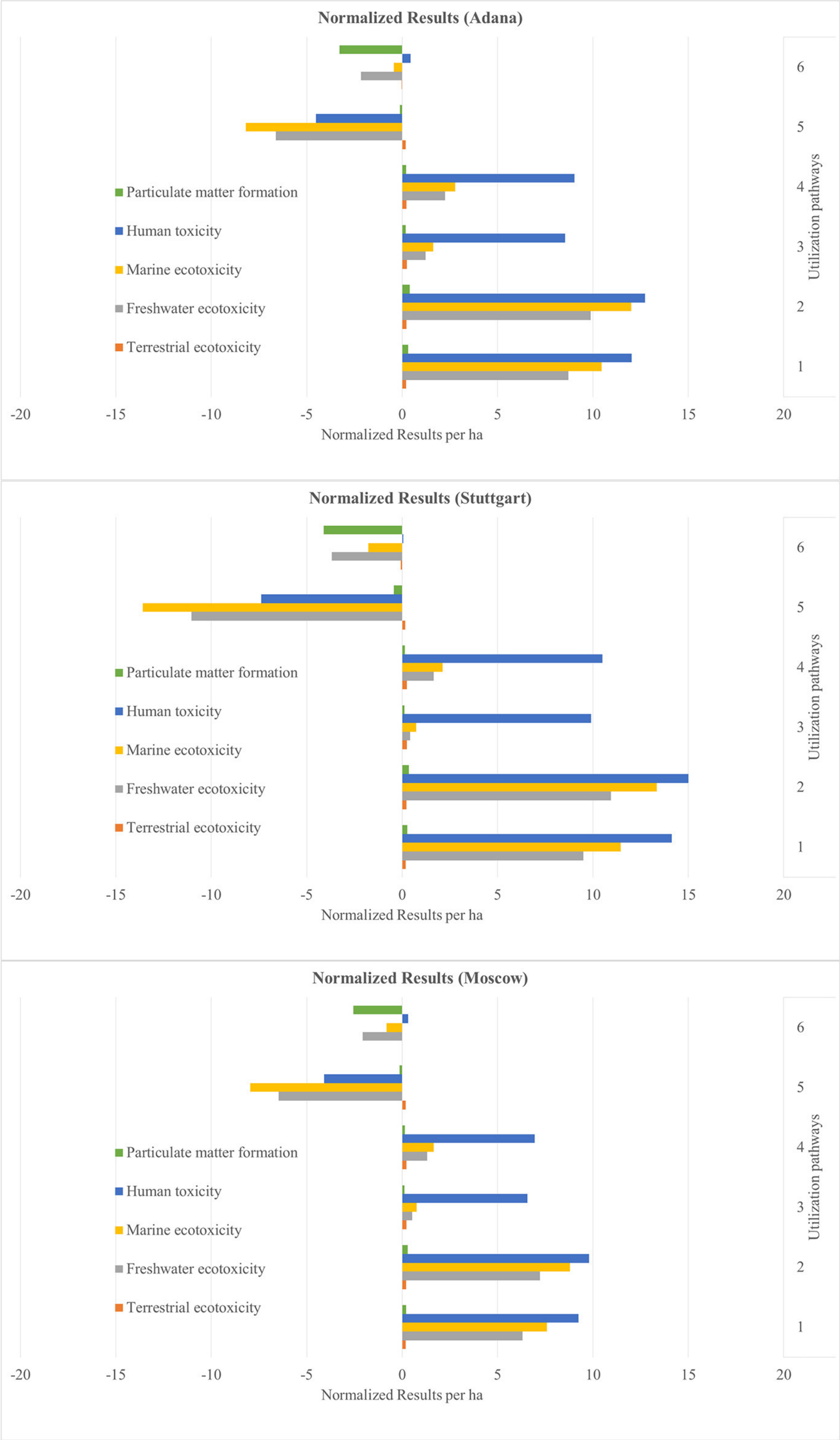

FIGURE 6 | Normalized results per ha for the sites Adana, Stuttgart, and Moscow—Part 2. Utilization pathways: 1. Small-scale combustion—chips; 2. Small-scale combustion—pellets; 3. Large-scale combustion—biomass baled for transport and storage; 4. Large-scale combustion—pellets; 5. Medium-scale biogas plant—biomass ensiled; and 6. Large-scale production of insulation material—biomass baled for transport and storage.

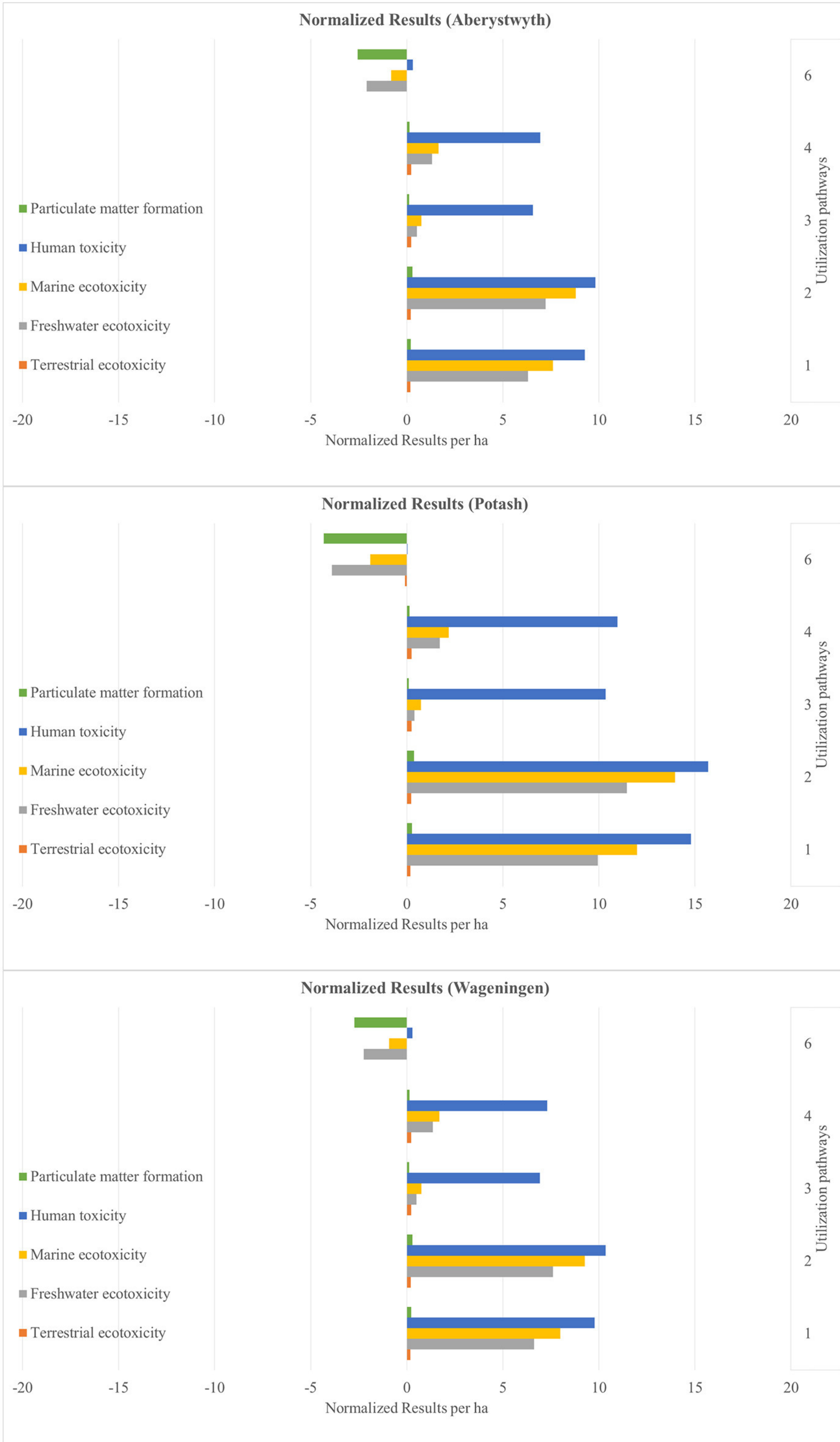

FIGURE 7 | Normalized results per ha for the sites Aberystwyth, Potash, and Wageningen—Part 2. Utilization pathways: 1. Small-scale combustion—chips; 2. Small-scale combustion—pellets; 3. Large-scale combustion—biomass baled for transport and storage; 4. Large-scale combustion—pellets; and 6. Large-scale production of insulation material—biomass baled for transport and storage.

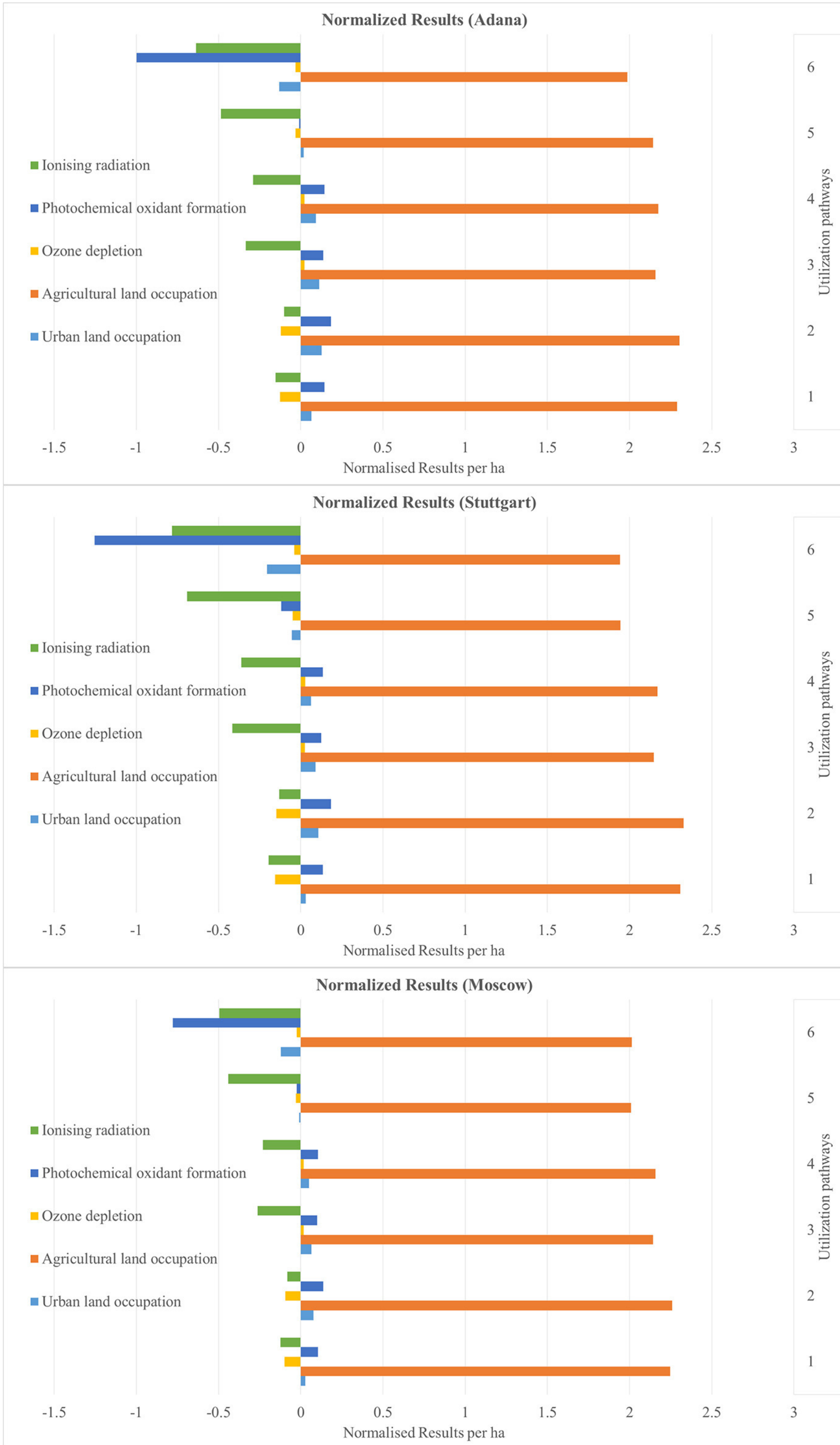

FIGURE 8 | Normalized results per ha for the sites Adana, Stuttgart, and Moscow—Part 3. Utilization pathways: 1. Small-scale combustion—chips; 2. Small-scale combustion—pellets; 3. Large-scale combustion—biomass baled for transport and storage; 4. Large-scale combustion—pellets; 5. Medium-scale biogas plant—biomass ensiled; and 6. Large-scale production of insulation material—biomass baled for transport and storage.

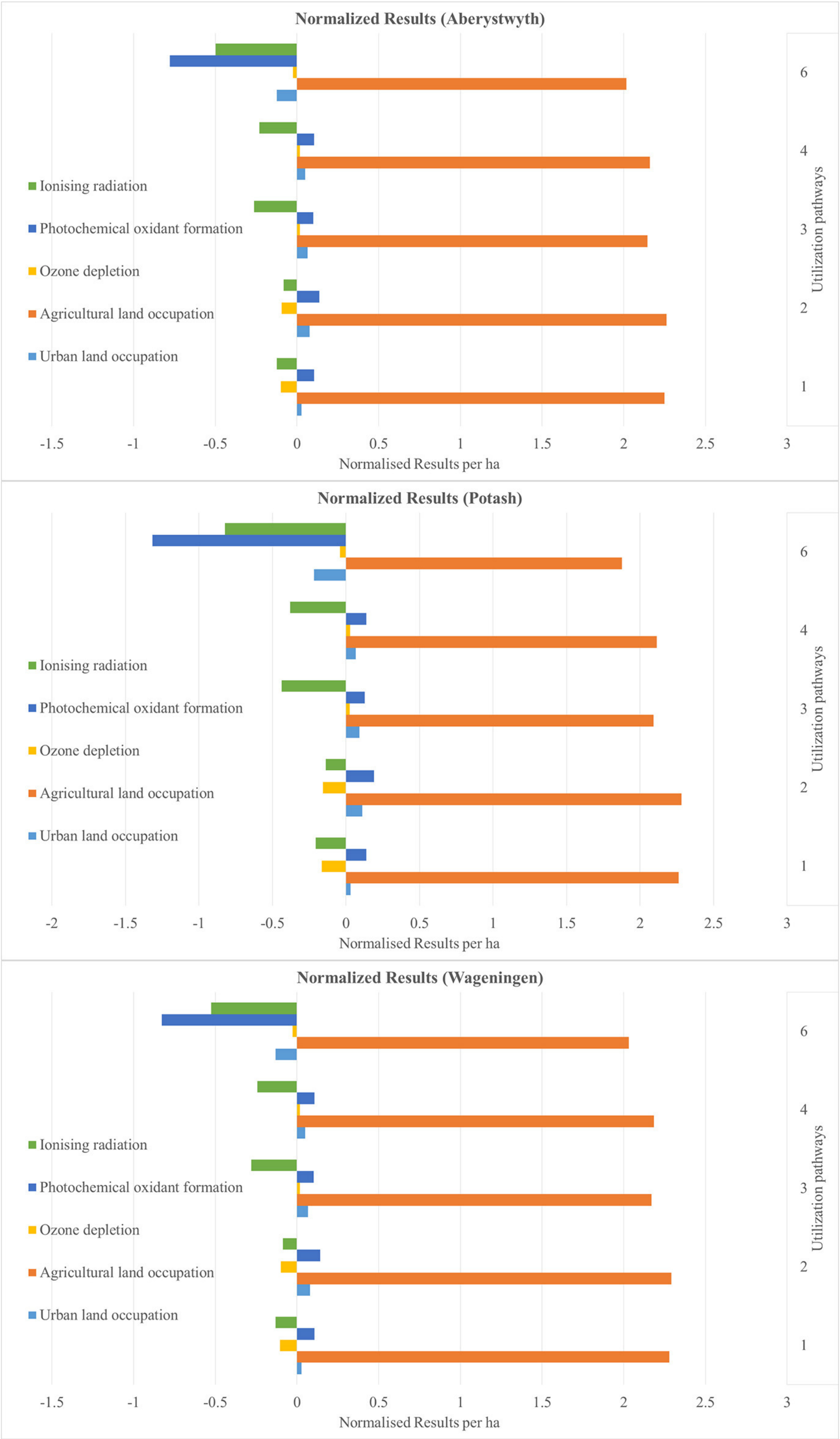

FIGURE 9 | Normalized results per ha for the sites Aberystwyth, Potash, and Wageningen—Part 3. Utilization pathways: 1. Small-scale combustion—chips; 2. Small-scale combustion—pellets; 3. Large-scale combustion—biomass baled for transport and storage; 4. Large-scale combustion—pellets; and 6. Large-scale production of insulation material—biomass baled for transport and storage.

and 5). However, they scored worse in most of the other impact categories. This also emphasizes the difficulty of determining the most sustainable utilization option from an environmental point of view. One way of resolving this issue is to combine the results of several impact categories into a single score for the total environmental sustainability (Rajagopalan et al., 2017). However, such an aggregation reduces the overall transparency of the results (Bare et al., 2000).

There is a large variation in the results between the six sites and between the six utilization pathways. The site differences are chiefly caused by variations in yield. The differences between the utilization pathways have several causes: the reference products have the largest impact, but the credits given for co-products and the effect of End-of-Life phase also play an important role. These four factors with a strong influence on the environmental benefits and impacts are discussed in the following sub-sections.

Influence of the Variability of the Biomass Yield

The average yields used in this assessment are based on the yield measured in the third year and are at the lower end of those of other studies (Christian et al., 2008; Iqbal et al., 2015). In this study, it was assumed that full yields are reached from the third year onwards. However, other studies analyzing long-term field trials suggest that full yields are only achieved from the fourth year onwards (Christian et al., 2008; Iqbal et al., 2015). That would mean that the yields used in this study are conservative assumptions and could be higher over the whole cultivation period.

The differences between the six sites for the same utilization pathways seen in **Figures 4–9** can be attributed to differences in yield. Sites on which significantly higher yields were achieved (e.g., Potash and Stuttgart) showed a better environmental performance. Other studies also emphasize the importance of yield for environmental performance (Meyer et al., 2016). However, it is worth mentioning that the influence of yield variation only changed an impact into a benefit, or vice versa, in very few impact categories, independent of utilization pathway (see **Figures 4–9**). Aberystwyth was a particularly interesting site; the values for the environmental benefits here were low compared to the other sites. The reason for that is that, in Aberystwyth, the yield was lower because the miscanthus was grown on marginal land. However, some utilization pathways, such as production of insulation material, still achieved comparatively low impacts on the environment even though the miscanthus was cultivated under marginal conditions.

Influence of the Selection of the Reference Product

The selection of an appropriate reference product is essential for the accuracy of the assessment, especially in the case of the heat-producing value chains 1–4 (Wolf et al., 2016). For the utilization pathways 1 and 2 (small-scale combustion), heat produced by combustion of light fuel oil was substituted. Changing the reference product to natural gas alters the results substantially. The net impact for the category MRD increases by 231%, for PMF by 220%, and for POF by 220%. In addition, the climate change saving potential is reduced by 77% and the benefit in the impact category fossil fuel depletion is reduced by 66%. This sensitivity analysis clearly shows the influence of the selection of the reference product on the result of the assessment. Furthermore, it emphasizes how crucial it is in practice to first phase out emission-intensive power plants based on coal and fuel oil, rather than those based on natural gas. However, the change of the reference product in utilization pathways 1 and 2 only turns a net benefit into an impact in the impact categories ionizing radiation and terrestrial acidification. The results of this sensitivity analysis are shown in the Supplementary Material (Table S8).

Heat generated by the combustion of natural gas was selected as reference product for the utilization pathways 3 and 4. Natural gas is a fossil energy carrier with comparatively low environmental impacts (May and Brennan, 2006), thus reducing the risk of overestimating the benefits of substitution by miscanthus-based heat. However, this also means that the environmental performance of the utilization pathways 3 and 4 can be improved considerably if heat generated by the combustion of fuel oil or coal is substituted.

The European electricity mix was used as reference product for the energetic utilization pathway 5. The choice of this reference is one reason for the low impacts on the environment of this utilization pathway. As electricity is an energy form with higher emissions per MJ than heat generation, the net benefits of its substitution are also higher. It should be noted that in this study an electricity mix was used as a reference product, which also includes electricity from renewable sources (Weidema et al., 2013). If only electricity generated by fossil sources is substituted, the environmental performance can be further improved.

Influence of Credits Given for Co-products

For those utilization pathways with more than one product, credits were given for the co-products. This was the case for the electricity produced as co-product in the CHP unit in the utilization pathways 3 and 4. The CHP produced 0.3 MJ of electricity for every MJ heat and it was assumed that this electricity substituted a European electricity mix. As already mentioned above, electricity has higher negative impacts on the environment than heat. That is why, in most impact categories, the credits given for the co-product were higher than the effect of substituting the reference product (see Table S9). The utilization pathway 5 produces heat as a co-product, which is partly utilized to heat nearby buildings, thus substituting fossil-based heat. In addition, the fermentation residues are rich in nutrients and can be used to substitute mineral fertilizers. These residues are a particularly valuable resource and the credits given for their utilization improve the environmental performance of this pathway considerably. The values used for these credits are displayed in Table S10.

Influence of the Inclusion of the End-of-Life Phase

The inclusion of the End-of-Life of biobased products is also an important point with a strong influence on their environmental performance. The insulation material produced in pathway 6 is first used as a biobased construction material and after the use phase incinerated in a CHP. The positive influence of this multiple use is important for the relatively low impacts on the environment of miscanthus-based insulation material. For example, the production of this insulation material (including the cultivation phase on the Stuttgart site and the truck transport of the biomass) causes around 124 kg CO_2 eq. per m^3. Of this, around 117 kg CO_2 eq. can be recovered through its incineration, generating heat and power which substitute conventionally produced energy. In the impact category terrestrial acidification, 0.58 kg SO_2 eq. per m^3 are saved through this energy recovery, which is more than are emitted in the whole value-chain including the production process (0.42 kg SO_2 eq.). These advantages of multiple use in comparison to single use have also been shown in other studies (Höglmeier et al., 2014, 2015). Another advantage of material use is the temporal storage of carbon in the product (Sikkema et al., 2013). This storage function can help decelerate climate change.

Relevance of Different Impact Categories

The normalization step applied enables the assessment of the relevance of the different impact categories for the environmental performance of each utilization pathway (Wagner and Lewandowski, 2017). There are large variations in relevance within the utilization pathways and within the impact categories analyzed. Once the relevance of an impact category has been established, it becomes evident which need to be included in a holistic analysis of the environmental performance of miscanthus-based value chains. The relevance of the impact categories should not only be evaluated in general but also for each specific utilization pathway. This knowledge assists the selection of the impact categories that require further improvement in each pathway.

The following section classifies the impact categories according to their normalized values into three groups: impact categories of (1) low relevance; (2) average relevance and (3) high relevance.

Several impact categories have comparatively low normalized impacts or benefits on the environment in most pathways and are therefore deemed of low relevance. These include: terrestrial acidification (TA), mineral resource depletion (MRD), particulate matter formation (PMF), ionizing radiation (IR), ozone depletion (OD), urban land occupation (ULO), photochemical oxidant formation (POF), and terrestrial ecotoxicity (TET). In addition, as the model and the LCI data used contain some uncertainties, small differences of ± 2 in normalized values are not considered significantly different.

The impact categories marine eutrophication (ME) and fossil fuel depletion (FFD) are deemed of average relevance. They should be included in the assessment, if the utilization pathways analyzed are expected to have a substantial impact in these categories. This is the case for ME, when higher amounts of nitrogen fertilizer are applied. The ME then increases considerably because higher nitrogen fertilizer application leads to an increase in nitrate leaching, the main cause of ME. As the production process of mineral nitrogen fertilizer is quite energy-intensive, FFD should also be included, when higher amounts of nitrogen fertilizer are applied. The FFD should also be assessed if the production phase of the utilization pathways analyzed requires large amounts of energy.

On the basis of the comparatively high normalized results, the impact categories human toxicity (HT), marine (MET), and freshwater ecotoxicity (FET) and freshwater eutrophication (FE) are considered very relevant for the assessment of miscanthus-based value chains. These results usually represent a substantial net impact for the combustion chains and a considerable net benefit for utilization in a biogas plant and production of insulation material.

The impact categories climate change (CC) and agricultural land occupation (ALO) are both deemed of high relevance, even if they have comparably low normalized impacts or benefits. This is due to the related environmental and social problems, which are of high interest to society in general. Climate change, for example, is presently one of the most urgent environmental problems and, as a result, this impact category is included in virtually every study which assesses the environmental performance of miscanthus-based value chains (Godard et al., 2013; Parajuli et al., 2015; Roy et al., 2015). The ALO can be a problem if the utilization of land for biomass production leads to land-use competition and thus hinders the production of food crops.

Although the normalization of the results allows the evaluation of the relevance of different impact categories, this method has its limitations. For example, it does not consider social preferences. In addition, the preload of the environment is not taken into account. For this reason, the results of the relevance assessment always need to be adapted according to the goal and scope of the respective study.

How to Improve the Environmental Performance

The relevance of the different impact categories also helps to identify potential for improvement by starting the focus on the categories with the highest normalized scores. The high values of the combustion chains for HT are caused by the treatment of the ash, which is rich in heavy metals. In this study the entire ash was disposed of to sanitary landfill. A separation into fly ash and coarse ash could improve the environmental performance. In this case, only the fly ash, which contains most of the heavy metals, would be disposed of to landfill and the coarse ash, which is rich in phosphate and potassium, could be used as fertilizer (Pitman, 2006). Performance in MET and FET is also problematic, especially for the combustion chains. The combustion process of the miscanthus biomass is responsible for the largest share of the emissions in these impact categories. Improvements in the emission control systems of the combustion unit would be one possibility to decrease the impacts in these categories. Another could be adaption of the harvest date and selection of the genotype in order to utilize biomass that contains less elements which lead to harmful emissions in the combustion process (Iqbal and Lewandowski, 2014).

The impact category ALO chiefly describes the area of agricultural land needed to produce the amount of biomass required for each utilization pathway. If it is possible to obtain higher yields per hectare, less land would be needed to produce the same amount of biomass and thus the ALO would decrease. Another possibility would be to increase the use efficiency of the biomass utilization pathways, so that less biomass is needed to produce the same amount of products.

The ME is mostly caused by nitrate leaching through the use of nitrogen fertilizers. Nitrogen-fertilizer-induced emissions in form of N_2O are also a main hot spot in the impact category CC. Thus, a decrease in the amount of nitrogen fertilizer used would decrease the impact in these categories. Another possibility for improvement would be the use of nitrification inhibitors (Akiyama et al., 2010). In the impact category FFD, there is a clear distinction between the energetic (1, 2, 3, 4, 5) and the material (6) utilization pathways. The hot spots in the energetic pathways are the harvest, biomass transport to the conversion plant and pelleting process (where applicable). In utilization pathway 6 (insulation material), the production process is the main hot spot and has the largest potential for improvement, for example, through the use of renewable instead of fossil-based energy forms.

Outlook

The utilization pathways modeled in this assessment are all based on novel genotypes, except at the Adana site. These novel genotypes were more suitable than the standard genotype *Miscanthus* × *giganteus* for the utilization pathways analyzed, based on yield and quality parameters (Lewandowski et al., 2016). Thus, the environmental performance assessed in this study reflects the advances made in recent years in both agricultural management and miscanthus breeding. The results reveal substantial differences in environmental performance between the various utilization pathways. Furthermore, they emphasize the advantages of the multiple use of biomass (as in the case of insulation material) compared to single use as an energy carrier. In order to increase the environmental benefits of biomass-based value chains, in future the material use of biomass should be favored.

Another relevant outcome of this study was the demonstration of the positive environmental performance of marginal land for miscanthus biomass production and utilization. In a developing European bioeconomy with a steadily increasing demand for biomass, this is a promising opportunity to boost biomass production without competing with food crops.

AUTHOR CONTRIBUTIONS

MW was performing the LCA modeling and was leading the writing process. AK, AH, and YI provided data and contributed to the material and method parts and thus supporting the creation of the Life cycle inventory. Furthermore AH supported the modeling process and AK the discussion of the results. IL added valuable contribution to each chapter and in manifold discussions.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: http://journal.frontiersin.org/article/10.3389/fpls.2017.00990/full#supplementary-material

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Conflict of Interest Statement: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

3.2 Environmental performance of miscanthus, switchgrass and maize: Can C4 perennials increase the sustainability of biogas production?

In this thesis the research question was raised, if the cultivation and subsequent utilization of perennial crop biomass have lower environmental impacts than annual crop biomass. In order to address this issue, the environmental performance of the utilization of two perennials (miscanthus and switchgrass) and the annual crop maize for biogas production was assessed and compared in the impact categories climate change (CC), fossil fuel depletion (FFD), terrestrial acidification (TA), freshwater eutrophication (FE) and marine eutrophication (ME). The results of this assessment are presented in the following sub-chapter.

This sub-chapter is published in the journal *Sustainability*, as:

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Environmental Performance of Miscanthus, Switchgrass and Maize: Can C4 Perennials Increase the Sustainability of Biogas Production?

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Abstract: Biogas is considered a promising option for complementing the fluctuating energy supply from other renewable sources. Maize is currently the dominant biogas crop, but its environmental performance is questionable. Through its replacement with high-yielding and nutrient-efficient perennial C4 grasses, the environmental impact of biogas could be considerably improved. The objective of this paper is to assess and compare the environmental performance of the biogas production and utilization of perennial miscanthus and switchgrass and annual maize. An LCA was performed using data from field trials, assessing the impact in the five categories: climate change (CC), fossil fuel depletion (FFD), terrestrial acidification (TA), freshwater eutrophication (FE) and marine eutrophication (ME). A system expansion approach was adopted to include a fossil reference. All three crops showed significantly lower CC and FFD potentials than the fossil reference, but higher TA and FE potentials, with nitrogen fertilizer production and fertilizer-induced emissions identified as hot spots. Miscanthus performed best and changing the input substrate from maize to miscanthus led to average reductions of −66% CC; −74% FFD; −63% FE; −60% ME and −21% TA. These results show that perennial C4 grasses and miscanthus in particular have the potential to improve the sustainability of the biogas sector.

Keywords: anaerobic digestion; *Miscanthus x giganteus*; *Panicum virgatum*; *Zea mays*; LCA; GWP; carbon mitigation; fossil fuel depletion; acidification; eutrophication

1. Introduction

Biogas is a renewable energy carrier produced by anaerobic digestion of biomass. Various kinds of biomass can be utilized for biogas production, such as sewage sludge, agricultural residues (e.g., manure), biogenic waste and energy crops [1]. Power production based on biogas is more reliable than other renewable energy sources, e.g., wind and solar, and can be used to cover power demand peaks or fluctuations in production due to unfavorable weather conditions. Biogas can be utilized directly in combined heat and power units (CHP) or can be upgraded to biomethane and transported to large gas power stations via the gas grid.

The Renewable Energy Act (EEG) and its amendments have led to a rapid increase in biogas exploration in Germany [1]. Here, approximately 8075 biogas plants with a total installed capacity of 4.1 GW were in operation in 2016 [2]. The latest amendments promote the restructuring of biogas plants to flexible operation, and approximately 31% of the installed capacity [2] have already been modernized. This allows power production to be adapted more to demand. Currently, 182 biogas plants upgrade biogas to biomethane and inject it into the gas grid [2]. These numbers show that, in Germany, there is a significant biogas infrastructure in place and the process of adapting it to the needs of a future renewable power supply has already begun. However, to allow an economically and environmentally viable operation, this infrastructure needs a reliable, affordable and sustainable supply of biomass. In 2014, substrate input (based on mass) was composed of 52% energy crops (of which 73% was maize) and 43% manure [2]. However, the proportion of biogas produced from energy crops is considerably higher than their proportion by mass, because they have a higher specific biogas and methane yield than other biogas substrates, e.g., manure. In Germany, about 1.4 million ha energy crops are grown for biogas production, of which 0.9 million ha are biogas maize [2]. This reveals the great importance of energy crops—and in particular energy maize—in Germany. The high economic viability of maize [3] for biogas production is given by its high methane yield, easy digestibility, and well-established, optimized crop production and harvest logistics, including storage as silage.

However, the strong reliance of the biogas sector on maize as substrate crop can lead to environmental problems and a low acceptance in public opinion. The environmental profile of maize cultivation is characterized by a high nitrogen fertilizer input, high risk of erosion and leaching, and negative impact on biodiversity [4–6]. In particular, the regional concentration in areas with high biogas plant densities can lead to environmental problems, such as surface and groundwater pollution through erosion and leaching, and losses in biodiversity and soil organic matter due to the high proportion of maize in crop rotations [7]. Other aspects are also criticized, such as the high concentration of maize in the landscape and the use of good agricultural land for growing energy instead of food crops. For these reasons, the sustainability of the biogas sector is often questioned not only by environmentalists but also by the general public.

The replacement of maize (*Zea mays*) by crops with a more benign environmental profile is seen as one route towards more sustainable biogas production. These crops, however, should have an equally high yield and biomass supply potential as maize. The high-yielding and nutrient-efficient perennial C4 grasses miscanthus and switchgrass are considered promising options.

The miscanthus genotype, *Miscanthus x giganteus*, was introduced into Europe in 1935 and is today still the only commercial genotype available on the market [8]. However, promising breeding efforts have begun in recent years and latest results show the suitability of novel genotypes for marginal lands and the potential contribution of miscanthus to greenhouse gas (GHG) mitigation [9]. Progress in upscaling miscanthus cultivation and crop production has also raised interest in the industrial sector [10]. Miscanthus' beneficial environmental profile is mainly due to its perennial nature and because soil organic carbon tends to increase when arable land is converted to its cultivation [11]. It is a very resource- and land-use efficient crop with efficient nutrient-recycling mechanisms and high net energy yields per unit area [12,13]. For this reason, the global warming potential (GWP) and the resource depletion potential of miscanthus cultivation is low [14,15]. Miscanthus is suitable for biogas production and has a high methane yield potential per unit area [16–18]. For anaerobic digestion, the biomass is harvested before winter, which increases the yield and digestibility [18]. Whittaker et al. [19] proved storage of green miscanthus via ensilaging to be feasible with losses in a similar range as for maize. These losses were significantly reduced by the addition of silage additives [19]. Compared to the conventional harvest of dry biomass in early spring, a green harvest in late autumn prevents leaf fall over winter, which leads to a higher nutrient removal than at spring harvest [13,18]. However, the recycling of fermentation residues is assumed to at least partially compensate for this removal and contribute to the formation of soil organic carbon. Nevertheless, the effects of a green cut on the development of soil carbon and fertility needs to be further investigated and is for this reason not considered in this study.

The crop production and environmental profile of switchgrass (*Panicum virgatum*) is comparable to that of miscanthus, except establishment via seeds and not rhizomes. Switchgrass is native to the US and Canada, where it has been developed as a promising energy grass [20]. It is also suitable for biogas production as harvest of green biomass and even double-cutting is possible [21]. Although yields are generally lower than with *Miscanthus x giganteus* [22], switchgrass can perform equally well under abiotic stress, such as cold and drought [23]. Its major advantage over the miscanthus genotypes presently available (mainly propagated clonally via rhizomes) is its low-cost establishment via seeds. Currently, switchgrass is not commercially cultivated in Germany and miscanthus is grown on an estimated area of 4000 hectares, mainly for combustion purposes [9]. Extending the utilization to anaerobic digestion could contribute to the sustainability and crop diversity (important for biodiversity) of the biogas sector.

The objective of this paper is to assess and compare the environmental performance in biogas production of the perennial C4 grasses miscanthus and switchgrass and the annual C4 crop maize. This was done in a Life Cycle Assessment (LCA) according to ISO standards 14040 and 14044 [24,25], using data from a field trial and laboratory measurements. Wagner and Lewandowski [26] showed that, when analyzing the environmental performance of biobased value chains, it is crucial to consider more impact categories than just global warming potential (GWP). Therefore, the following impact categories were assessed to estimate the environmental performance of the crops and their subsequent utilization: climate change (CC)—which corresponds to the GWP, freshwater eutrophication (FE), marine eutrophication (ME), terrestrial acidification (TA) and fossil fuel depletion (FFD). The impact categories FE, ME and TA were chosen as eutrophication and acidification have been identified as important impact categories for agricultural systems. The category marine eutrophication represents the impact of nitrogen on biomass growth in aquatic ecosystems. Freshwater eutrophication represents the same impact, but caused by phosphorus [27,28].

The data for the LCA were collected from a randomized split-block field trial, where miscanthus, switchgrass and maize were grown under *ceteris paribus* conditions. The field trial was started in 2002 and allows a comparison of annual and perennial crops. Samples and yield measurements for this LCA were taken in 2012 and 2013 and laboratory analyses were performed to estimate biogas and methane yield and biomass quality.

2. Material and Methods

2.1. Scope and Boundaries

The scope of the present study is an assessment of the environmental performance of the cultivation of three dedicated energy crops ((i) miscanthus (*Miscanthus x giganteus*); (ii) switchgrass (*Panicum virgatum* L.) "Kanlow"; and (iii) silage maize (*Zea mays*) "Mikado") and their subsequent fermentation in a biogas plant. The biogas produced is utilized in a CHP unit (Combined Heat and Power) to produce electricity and heat. The cultivation as well as the utilization of the biomass takes place in Germany. One kilowatt hour of electricity ($kWh_{el.}$) was chosen as the functional unit (FU). The environmental impacts of these biobased value chains were compared with the German electricity mix as a fossil reference. In order to do this, a system expansion approach was applied which enables the inclusion of fossil reference system hot spots.

The systems are described in Figure 1. On the right side the maize cultivation is shown, on the left side the cultivation of the perennial crops miscanthus and switchgrass. The system boundaries include the production of the mineral fertilizers and the herbicides used, the production of the propagation material (miscanthus rhizomes as well as switchgrass and maize seeds), and the agricultural management (soil preparation, planting, mulching, fertilizing, spraying of herbicides, harvesting, recultivation resp. stubble cultivation) over the whole cultivation period which is for maize 1 year, for switchgrass 15 years and for miscanthus 20 years. Miscanthus and switchgrass are mulched in the first year and harvested from the second year onwards. All crops are harvested with a self-propelled forage harvester. The biomass is then transported to the biomass plant where it is fermented to biogas which is combusted in a CHP unit to produce electricity and heat. The fermentation residues are rich in nutrients and are used as fertilizer.

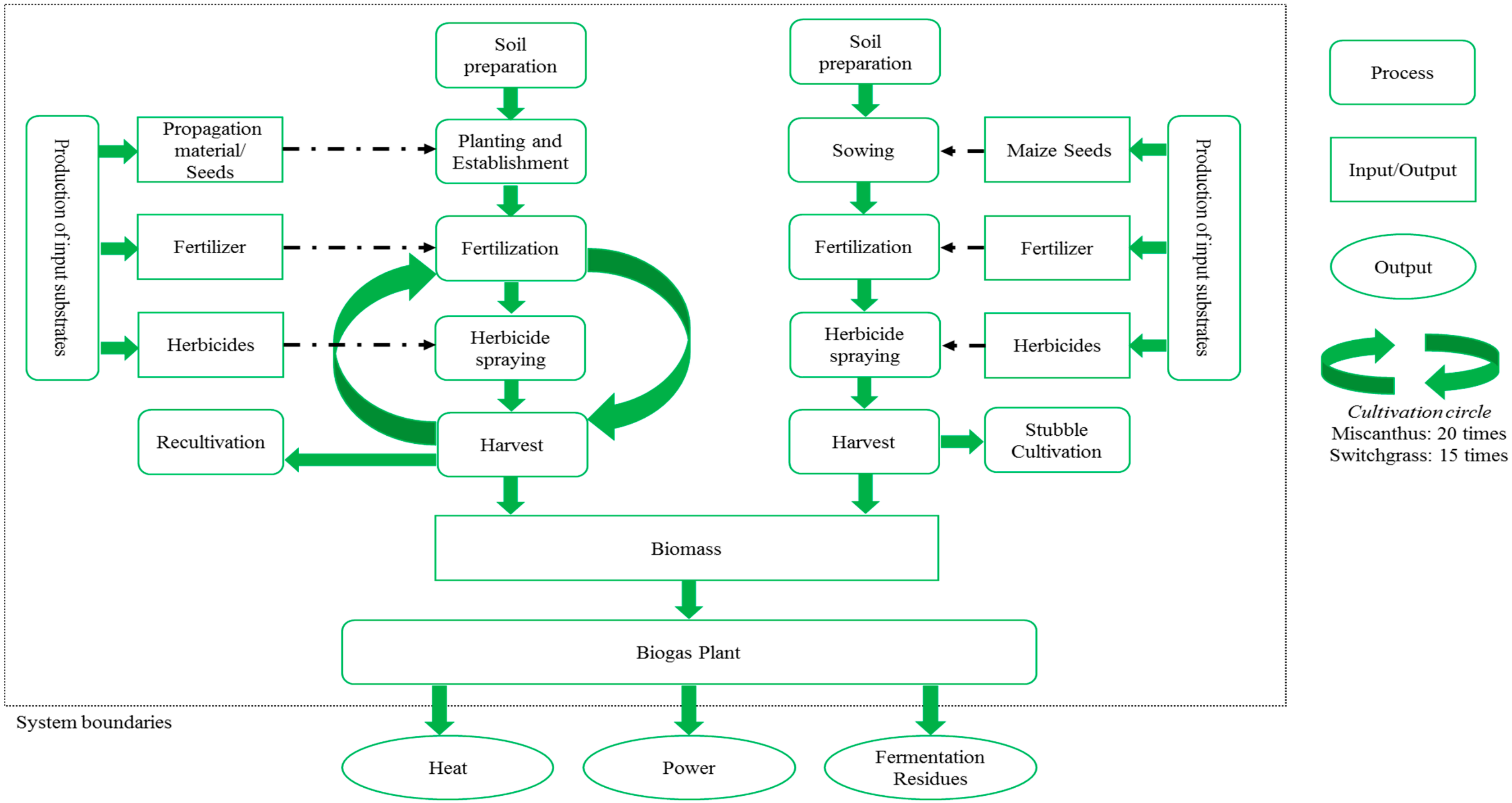

Figure 1. System description and boundaries for miscanthus, switchgrass (**left**) and maize (**right**) biomass cultivation, the fermentation to biogas and the subsequent utilization in a CHP unit.

2.2. Life Cycle Inventory

The data for the cultivation process used in this LCA were obtained from a multiannual field trial at Ihinger Hof. The Ihinger Hof is a research station of the University of Hohenheim and is located in southwest Germany (48.75°N and 8.92°E). The soil belongs to the soil class Haplic Luvisol. The long-term average annual air temperature and precipitation at the research station are 8.3 °C and 689 mm, respectively. The experimental design of the trial is described in Boehmel et al. [29].

Data on cultivation practices such as fertilizer and herbicide inputs were available for an 11-year period from 2002 to 2013. Miscanthus and switchgrass were established in spring 2002 by rhizome planting and sowing, respectively. Maize was sown on 27 April 2012 and 21 May 2013 at a density of 9.5 seeds m^{-2}. Nitrogen was applied as calcium ammonium nitrate (CAN), K_2O as potassium chloride and P_2O_5 as triplesuperphosphat (TSP). The use of herbicides during the miscanthus and switchgrass cultivation is described in Iqbal et al. [30]. For maize cultivation chemical weeding was performed using two conventional herbicides mixtures following good agricultural practice. The first application was a mixture of three herbicides (2.0 $L{\cdot}ha^{-1}$ Stomp Aqua, BASF SE, active ingredient 455 $g{\cdot}L^{-1}$ Pendimethalin; 1.0 $L{\cdot}ha^{-1}$ Spektrum, BASF SE, active ingredient 720 g L^{-1} Dimethenamid-P; and 1.0 $L{\cdot}ha^{-1}$ MaisTer power, Bayer, active ingredient 31.5 $g{\cdot}L^{-1}$ Foramsulfuron + 1.0 $g{\cdot}L^{-1}$ Iodosulfuron + 10.0 $g{\cdot}L^{-1}$ Thiencarbazone + 15.0 $g{\cdot}L^{-1}$ Cyprosulfamide). The second application was a mixture of two herbicides (1.7 $L{\cdot}ha^{-1}$ Laudis, Bayer, active ingredient 44 $g{\cdot}L^{-1}$ Tembotrione + 22 $g{\cdot}L^{-1}$ Isoxadifen-ethyl; and 0.35 $L{\cdot}ha^{-1}$ Buctril, Bayer, active ingredient 225 $g{\cdot}L^{-1}$ Bromoxynil).

The principle data for the cultivation of miscanthus, switchgrass and silage maize used in this analysis are summarized in Table 1. The data are shown for the years 2012 and 2013. In the year 2013 the weather conditions were not ideal for silage maize cultivation in Germany which is an important reason for the significantly lower yield of silage maize in the year 2013 compared to 2012. After a serious frost period in February 2012, the weather conditions in 2012 where quite usual, spring was rather dry, but followed by plenty of rain in June (Figure 2). Weather conditions in 2013 were completely contrary and very challenging for agriculture. The spring and especially May was unusually cool and wet. Due to this challenging weather conditions, maize sowing was delayed to late May. In July, the temperatures were unusually high and the crops faced a serious drought followed by few days of rain from 24 to 29 July. In this period, 168.5 mm of rainfall occurred in 4 major events, which represents 97% of the rain of the complete month.

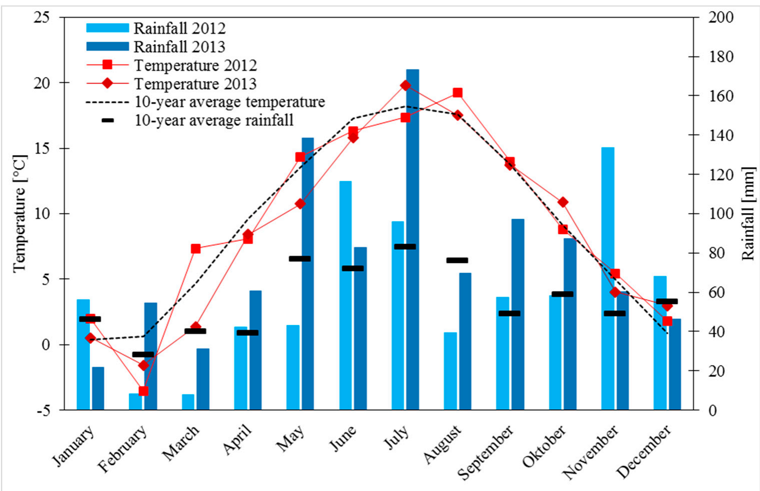

Figure 2. Temperature and rainfall in 2012 and 2013 at the field site on the research station "Ihinger Hof". For comparison the 10-year average temperature and rainfall from 2003 to 2012 is shown.

Maize was harvested at milk-ripe stage (end of September in 2012; late October in 2013) and miscanthus and switchgrass in late October in both years. The years 2012 and 2013 were selected to compare the environmental performance of perennial crops as an alternative to maize under different conditions for silage maize cultivation. The yield of maize, miscanthus and switchgrass is shown for the favorable year 2012 and non-favorable year 2013 in Table 1. However, the yield of the two perennial crops is the average yield over the whole cultivation period (20 years for miscanthus, and 15 for switchgrass) including the establishment phase based on the measured yield of the respective year. In the first year, miscanthus and switchgrass are mulched and not harvested. Full yields are only reached from the third year on. This calculation is exemplarily shown for the yield in 2012 for miscanthus in Equation (1) and for switchgrass in Equation (2) and was performed in the same way for the lower yields in 2013. The variable *yield_year2* describes the yield in the second cultivation year, which, for both crops, is slightly lower than the mean yield achieved in the following years.

$$Mean\ yield\ miscanthus\ [t\ DM\ ha^{-1}{\cdot}yr^{-1}] = \frac{yield\_year2\ +\ \mathrm{yield\_}year\mathrm{\_2012} \times 18}{20} \tag{1}$$

$$Mean\ yield\ switchgrass\ [t\ DM\ ha^{-1}{\cdot}yr^{-1}] = \frac{yield\_\ year2\ +\ \mathrm{yield\_}year\mathrm{\_2012} \times 13}{15} \tag{2}$$

The methane yield was measured as described in Kiesel and Lewandowski [18]. A biogas batch test was performed for 35 days at mesophilic conditions (39 °C) according to VDI guideline 4630. The approach of the biogas batch test was certified by the KTBL and VDLUFA interlaboratory comparison test 2014 and 2015. Each sample was assessed in four technical replicates.

Table 1. Summary of the in- and outputs of the three energy crops.

Input/Output	Unit	Maize		Switchgrass		Miscanthus	
		2012	2013	2012	2013	2012	2013
N	$Kg{\cdot}yr^{-1}{\cdot}ha^{-1}$	240	240	80	80	80	80
K_2O	$Kg{\cdot}yr^{-1}{\cdot}ha^{-1}$	304	204	137	137	128	128
P_2O_5	$Kg{\cdot}yr^{-1}{\cdot}ha^{-1}$	100	100	37	37	32	32
Herbicides	$Kg{\cdot}yr^{-1}{\cdot}ha^{-1}$	6.05	6.05	1.32	1.32	1.375	1.375
Dry matter yield	$Kg{\cdot}yr^{-1}{\cdot}ha^{-1}$	18915	12616	14227	8369	22760	18929
Dry matter content	%	25.4	21.1	38.9	36.2	43.4	41.2
Methane yield	$m^3\ CH_4\ yr^{-1}{\cdot}ha^{-1}$	5594	3635	3328	2095	5006	4542
Agricultural land required for biogas plant	$ha{\cdot}yr^{-1}$	173	266	291	461	194	213

The background data for the environmental impacts associated with the production of the input substrates (seeds, propagation material, herbicides and fertilizers) and the cultivation processes were taken from the GaBi database [31]. Direct N_2O and NO emissions from the mineral fertilizers used were calculated according to Bouwman et al. [32]. The estimations of indirect N_2O emissions from mineral fertilizers and N_2O emissions from harvest residues were done in accordance to IPCC [33]. Nitrate leaching to groundwater was calculated according to the SQCB—NO_3 model [34]. Ammonia emissions were calculated using emission factors from the Joint EMEP/CORINAIR Atmospheric Emission Inventory Guidebook [35]. Phosphate emissions were estimated according to van der Werf et al. [36].

In this study a transport distance of 100 km by truck for the input material such as herbicides or fertilizer and of 5 km by tractor for the biomass from the field to the biogas plant was assumed. This assumption is align with literature [37–39] and was done, since no data for the transport distance of the input substrates to the farmer and the biomass to the biogas plant were available. The emission stage for the truck used was assumed to be EUR5. The data for the transportation processes of the input material and the biomass were taken from the GaBi database [31].

After the harvest, the biomass of the different crops is ensiled. During the ensilage process dry matter losses of 12% were assumed [40]. The silage is subsequently fermented in a biogas plant. The methane hectare yield of the different crops is shown in Table 1. In the biogas plant methane losses of 1% were assumed [41]. The biogas is then combusted in a CHP with an electrical capacity of 500 kW to produce heat and power. The technical characteristics of the CHP used in this analysis are shown in Table 2. The inherent power consumption for miscanthus and switchgrass was assumed to be 12% and thus significantly higher than for maize. This is due to the more energy intensive pre-treatment of lignocellulosic biomass before the fermentation process. The emissions associated with the combustions of the biogas were taken from the ecoinvent database [42]. The electricity generated is fed into the grid. Twenty percent of the heat produced is used internally for the heating of the fermenter. In practice the remaining heat is partially used for heating nearby buildings thereby substituting heat produced by fossil sources. In this study, it was assumed that of the remaining heat 50% is used for this purpose.

Table 2. CHP unit—technical characteristics.

Technical Characteristics		Unit
Full load hours	7800	h
Plant output electrical	500	$kWh_{el.}$
Plant output total	1219	kWh
Electrical efficiency	41	% of plant total output
Thermal efficiency	41	% of plant total output
Inherent heat demand	20	% of total heat production
Inherent power consumption—perennial crops	12	% of total power production
Inherent power consumption—silage maize	6.6	% of total power production

The residues of the fermentation process are rich in nutrients. Table 3 shows the plant available nutrients, which can be recycled through the use of fermentation residues as fertilizers (related to the generation of the functional unit of 1 $kWh_{el.}$). The nutrient content is the average of the measured values of year 2012 and 2013. The phosphorus and the potassium content of the biomass fermented remains fully in the fermentation residues. Only 70% of the nitrogen compounds in the fermentation residues are available for the plants. That is why the nitrogen content can therefore not be taken fully into account. The nitrogen (N) content was analyzed according to the DUMAS principle (method EN ISO 16634/1 and VDLUFA Method Book III, method 4.1.2) using a Vario Macro Cube (Elementar Analysensysteme GmbH, Hanau, Germany) element analyzer. The phosphor (P) and potassium (K) contents were analyzed according to DIN EN ISO 15510 and VDLUFA Method Book III, method 10.8.2 [43] using ICP-OES and a ETHOS.lab microwave (MLS GmbH, Leutkirch, Germany).

Table 3. Nutrients in the biomass of the analyzed energy crops and plant available nutrients which can be recycled through the use of fermentation residues per FU.

Nutrient	Miscanthus		Switchgrass		Maize	
	in % of Biomass (d.b.)	in kg/FU	in % of Biomass (d.b.)	in kg/FU	in % of Biomass (d.b.)	in kg/FU
N	0.47	0.0036	0.50	0.0035	1.29	0.0058
P	0.09	0.0010	0.10	0.0010	0.18	0.0011
K	1.11	0.0119	1.03	0.0105	1.29	0.0083

2.3. Choice of Impact Categories

In this LCA study the life cycle impact assessment method ReCiPe was used [44]. The following impact categories were considered: climate change (CC), which corresponds to global warming potential (GWP); terrestrial acidification (TA); freshwater eutrophication (FE); marine eutrophication (ME); and fossil fuel depletion (FFD). Characterization factors were taken from Goedkoop et al. [44]. These impact categories were chosen according to their relevance for perennial biomass production, which was analyzed in the study by Wagner and Lewandowski [26].

3. Results

For each impact category analyzed, data are shown for the two climatically different production years 2012 and 2013 (2012 favorable and 2013 non-favorable for silage maize cultivation) and for two scenarios, one with and one without heat utilization. These are presented both in figures and in tables, depicting the results with (figures) and without (tables) a system expansion approach. The results are presented per functional unit (FU), which is kWh electricity. In the supplementary material (S1–S5), the same results are presented per kg dry biomass.

The value in each impact category shows the net impact or benefit of the substitution of the fossil reference through a biobased alternative. In this study, the German electricity mix was substituted by power generated through the fermentation of dedicated energy crops and the subsequent combustion of the biogas in a CHP unit. A negative value in this case is thus a net benefit while a positive value is a negative impact on the environment.

In contrast, the table shows the environmental impact of the generation of 1 $kWh_{el.}$ in each impact category without this substitution, separated into the main emission sources. In this context, the *recycling of nutrients* represents the emission savings associated with the reduction in fertilizer in other crops through the use of the fermentation residues. The *agricultural management* summarizes all operation steps from soil preparation, planting, mulching, fertilizing, and spraying of herbicides to recultivation. The *fertilizer-induced emissions* are emissions associated with the use of fertilizers, such as N_2O emissions, which occur after the application of nitrogen fertilizer. *Credits heat utilization* are credits given for the substitution of heat produced via a fossil reference (in the present study natural gas) by heat generated via the combustion of biogas in the CHP unit. In the heat utilization scenario, 20% of the heat produced is used internally in the biogas plant. Of the remaining 80%, one half (40% of total heat produced) is used to heat nearby buildings, thus substituting heat from conventional sources.

3.1. Climate Change and Fossil Fuel Depletion

The production and use of the analyzed C4 crops, both perennial and annual, leads to a net GHG emission reduction up to 0.66 kg·CO_2-eqv. $(kWh_{el.})^{-1}$ through the substitution of a fossil reference (Figure 3). Furthermore, all scenarios show a net decrease of the fossil fuel depletion of up to 0.18 kg·oil-eqv. $(kWh_{el.})^{-1}$ (Figure 4). As expected, the scenarios with heat utilization lead to both higher GHG emission and fossil fuel saving (Figures 3 and 4). On average, miscanthus shows the highest GHG emission and fossil fuels saving potentials. Both perennial grasses perform better than maize (Figures 3 and 4). The advantage of miscanthus over switchgrass is larger than the advantage of switchgrass over maize.

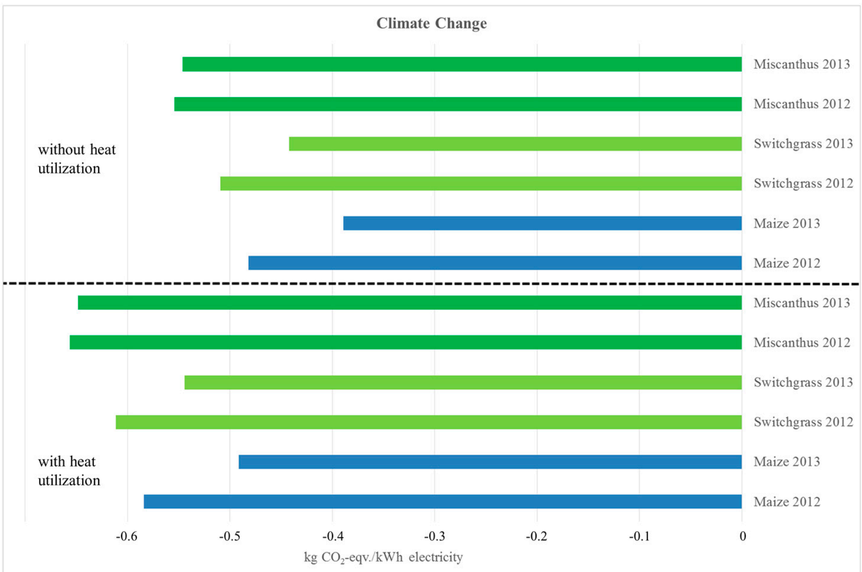

Figure 3. Assessment of the net benefits in kg·CO_2-eqv. of substituting 1 kWh_{el}. of the German electricity mix by power generated via combustion of the biogas in a CHP.

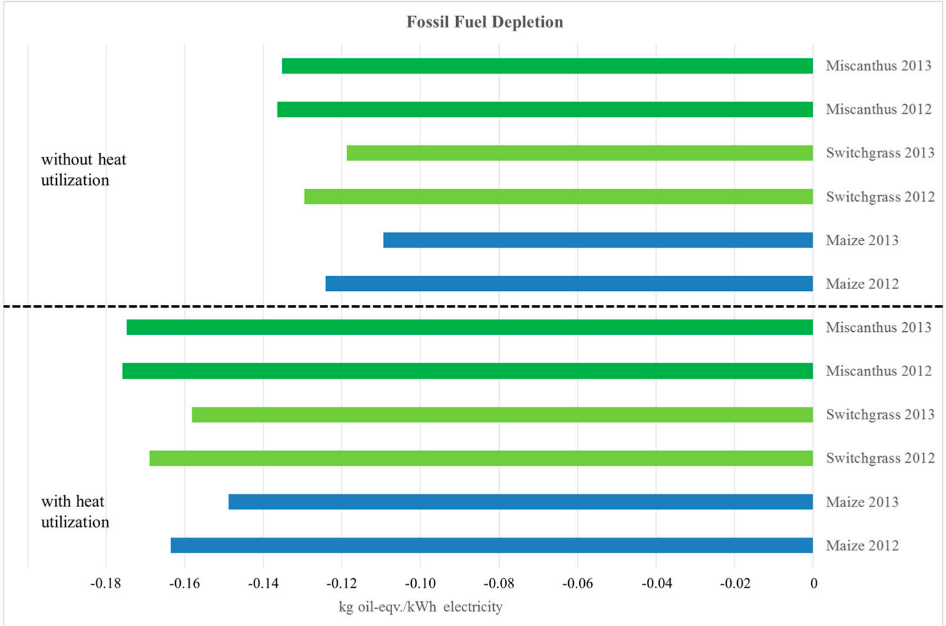

Figure 4. Assessment of the net benefits in kg·oil-eqv. of substituting 1 kWh_{el}. of the German electricity mix by power generated via combustion of the biogas in a CHP.

Table 4 shows the contribution of different processes to the GHG emissions and Table 5 the use of fossil fuels in these processes. The production of nitrogen fertilizer is responsible for the largest impact in both impact categories and for all crops. This is also the reason for the high credit—in terms of fossil energy savings—given for the recycling of nutrients from the fermentation residues (Table 5). Other processes with high impacts on GHG emissions and fossil energy consumption are harvest operation and biomass transport to the biogas plant (Tables 4 and 5).

Table 4. Assessment of the climate change in kg·CO_2-eqv. of 1 kWh_{el}. generated via the production and fermentation of dedicated energy crops and combustion of the biogas in CHP.

	Processes/Flows	**Maize per FU**		**Switchgrass per FU**		**Miscanthus per FU**		**Unit**
		2012	**2013**	**2012**	**2013**	**2012**	**2013**	
Input substrates	Production of nitrogen fertilizer	0.077	0.1185	0.0504	0.0800	0.0335	0.0369	kg·CO_2-eqv.
	Production of potassium fertilizer	0.0048	0.0075	0.0043	0.0068	0.0027	0.0029	kg·CO_2-eqv.
	Production of phosphate fertilizer	0.0064	0.0099	0.0047	0.0074	0.0027	0.0030	kg·CO_2-eqv.
	Recycling of nutrients	−0.0415	−0.0415	−0.0279	−0.0279	−0.0288	−0.0288	kg·CO_2-eqv.
	Herbicides	0.0028	0.0044	0.0012	0.0019	0.0008	0.0009	kg·CO_2-eqv.
	Seeds/Rhizomes	0.0002	0.0003	0.0001	0.0002	0.0003	0.0003	kg·CO_2-eqv.
Agricultural operations	Agricultural management	0.0075	0.0115	0.002	0.0032	0.0012	0.0013	kg·CO_2-eqv.
	Harvest	0.0038	0.0058	0.007	0.0111	0.0045	0.0049	kg·CO_2-eqv.
	Transport input substrates	0.0012	0.0018	0.0008	0.0013	0.0006	0.0006	kg·CO_2-eqv.
	Transport biomass	0.0049	0.0061	0.0047	0.0047	0.0045	0.0044	kg·CO_2-eqv.
	Ensilage	0.0003	0.0004	0.0005	0.0009	0.0004	0.0004	kg·CO_2-eqv.
	Fertilizer-induced emissions	0.0549	0.0906	0.0472	0.0725	0.0281	0.0311	kg·CO_2-eqv.
CHP	Biomass production system	0.1223	0.2154	0.0950	0.1622	0.0504	0.0580	kg·CO_2-eqv.
	CHP—Direct emissions	0	0	0	0	0	0	kg·CO_2-eqv.
	Credits heat utilization	−0.1021	−0.1021	−0.1021	−0.1021	−0.1021	−0.1021	kg·CO_2-eqv.
Total	**Total with credits**	**0.0202**	**0.1132**	**−0.0071**	**0.0600**	**−0.0518**	**−0.0441**	**kg·CO_2-eqv.**
	Total without credits	**0.1223**	**0.2154**	**0.0950**	**0.1622**	**0.0504**	**0.0580**	**kg·CO_2-eqv.**

Table 5. Assessment of the fossil fuel depletion in kg·oil-eqv. of 1 kWh$_{el}$. generated via the production and fermentation of dedicated energy crops and combustion of the biogas in CHP.

	Processes/Flows	Maize per FU		Switchgrass per FU		Miscanthus per FU		Unit
		2012	2013	2012	2013	2012	2013	
Input substrates	Production of nitrogen fertilizer	0.01598	0.02460	0.01046	0.01661	0.00695	0.00766	kg·oil-eqv.
	Production of potassium fertilizer	0.00206	0.00317	0.00182	0.00289	0.00113	0.00125	kg·oil-eqv.
	Production of phosphate fertilizer	0.00323	0.00497	0.00234	0.00372	0.00135	0.00148	kg·oil-eqv.
	Recycling of nutrients	−0.01020	−0.01020	−0.00742	−0.00742	−0.00774	−0.00774	kg·oil-eqv.
	Herbicides	0.00128	0.00196	0.00054	0.00087	0.00038	0.00042	kg·oil-eqv.
	Seeds/Rhizomes	0.00004	0.00005	0.00002	0.00003	0.00007	0.00008	kg·oil-eqv.
Agricultural operations	Agricultural management	0.00238	0.00367	0.00064	0.00101	0.00038	0.00042	kg·oil-eqv.
	Harvest	0.00121	0.00187	0.00222	0.00353	0.00143	0.00157	kg·oil-eqv.
	Transport input substrates	0.00037	0.00057	0.00027	0.00043	0.00019	0.00021	kg·oil-eqv.
	Transport biomass	0.00157	0.00194	0.00151	0.00152	0.00144	0.00139	kg·oil-eqv.
	Ensilage	0.00009	0.00014	0.00017	0.00028	0.00012	0.00013	kg·oil-eqv.
	Fertilizer-induced emissions	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	kg·oil-eqv.
CHP	Biomass production system	0.01801	0.03274	0.01258	0.02346	0.00569	0.00687	kg·oil-eqv.
	CHP—Direct emissions	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	kg·oil-eqv.
	Credits heat utilization	−0.03948	−0.03948	−0.03948	−0.03948	−0.03948	−0.03948	kg·oil-eqv.
Total	**Total with credits**	**−0.02147**	**−0.00674**	**−0.02691**	**−0.01602**	**−0.03379**	**−0.03262**	**kg·oil-eqv.**
	Total without credits	**0.01801**	**0.03274**	**0.01258**	**0.02346**	**0.00569**	**0.00687**	**kg·oil-eqv.**

3.2. Freshwater Eutrophication and Marine Eutrophication

The substitution of the fossil reference lead to a net increase in freshwater eutrophication of up to 3.5×10^{-5} kg·P-eqv. (kWh$_{el.}$)$^{-1}$ in all scenarios (Figure 5). On average, the freshwater eutrophication potentials are lowest for miscanthus, followed by switchgrass and then maize (Figure 5).

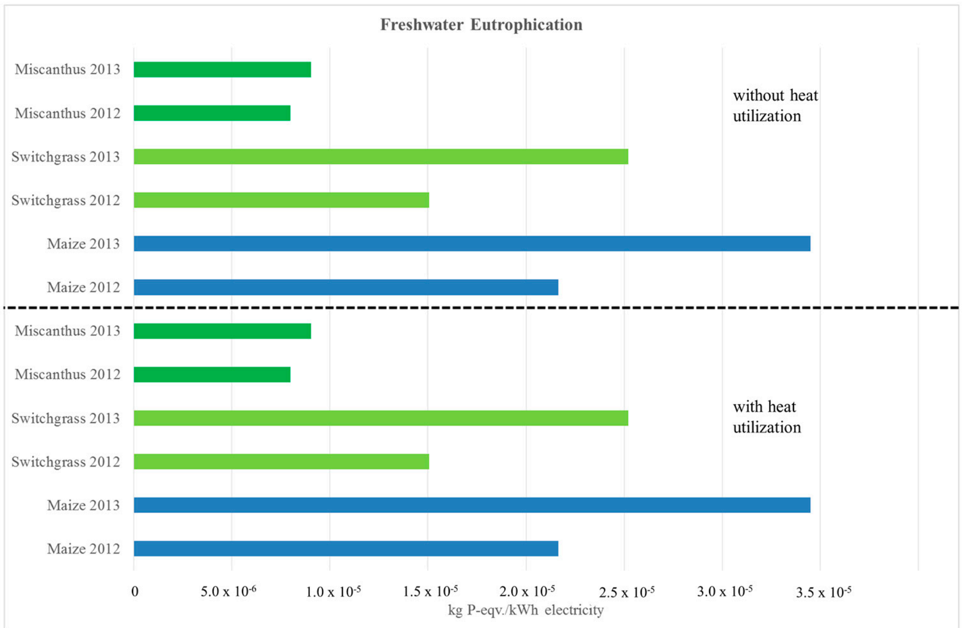

Figure 5. Assessment of the net impacts in kg·P-eqv. of substituting 1 kWh$_{el.}$ of the German electricity mix by power generated via combustion of the biogas in a CHP.

The recycling of nutrients leads to a high credit, which has a positive impact on the freshwater eutrophication (Table 6). In all scenarios, fertilizer-induced emissions account for the largest share of freshwater eutrophication. These are phosphate emissions associated with the use of phosphorus fertilizer, which are highest in maize and lowest in miscanthus (Table 6). The second-largest share comes from nitrogen fertilizer production, followed by the production of phosphate fertilizers (Table 6).

Table 6. Assessment of the freshwater eutrophication in kg·P-eqv. of 1 kWh$_{el.}$ generated via the production and fermentation of dedicated energy crops and combustion of the biogas in CHP.

	Processes/Flows	Maize per FU		Switchgrass per FU		Miscanthus per FU		Unit
		2012	2013	2012	2013	2012	2013	
Input substrates	Production of nitrogen fertilizer	1.18×10^{-7}	1.82×10^{-7}	7.74×10^{-8}	1.23×10^{-7}	5.14×10^{-8}	5.67×10^{-8}	kg·P-eqv.
	Production of potassium fertilizer	7.21×10^{-9}	1.11×10^{-8}	6.38×10^{-9}	1.01×10^{-8}	3.96×10^{-9}	4.36×10^{-9}	kg·P-eqv.
	Production of phosphate fertilizer	7.56×10^{-8}	1.16×10^{-7}	5.49×10^{-8}	8.72×10^{-8}	3.16×10^{-8}	3.48×10^{-8}	kg·P-eqv.
	Recycling of nutrients	-9.63×10^{-8}	-9.63×10^{-8}	-7.09×10^{-8}	-7.09×10^{-8}	-7.30×10^{-8}	-7.30×10^{-8}	kg·P-eqv.
	Herbicides	1.47×10^{-8}	2.26×10^{-8}	6.28×10^{-9}	9.97×10^{-9}	4.36×10^{-9}	4.80×10^{-9}	kg·P-eqv.
	Seeds/Rhizomes	1.34×10^{-7}	2.07×10^{-7}	2.88×10^{-8}	4.58×10^{-8}	2.76×10^{-7}	3.04×10^{-7}	kg·P-eqv.
Agricultural operations	Agricultural management	4.91×10^{-8}	7.56×10^{-8}	1.31×10^{-8}	2.09×10^{-8}	7.91×10^{-9}	8.72×10^{-9}	kg·P-eqv.
	Harvest	2.50×10^{-8}	3.85×10^{-8}	4.58×10^{-8}	7.28×10^{-8}	2.94×10^{-8}	3.24×10^{-8}	kg·P-eqv.
	Transport input substrates	7.63×10^{-9}	1.17×10^{-8}	5.55×10^{-9}	8.82×10^{-9}	3.87×10^{-9}	4.27×10^{-9}	kg·P-eqv.
	Transport biomass	3.23×10^{-8}	4.00×10^{-8}	3.12×10^{-8}	3.13×10^{-8}	2.97×10^{-8}	2.87×10^{-8}	kg·P-eqv.
	Ensilage	2.80×10^{-9}	2.80×10^{-9}	5.67×10^{-9}	5.67×10^{-9}	2.62×10^{-9}	2.62×10^{-9}	kg·P-eqv.
	Fertilizer-induced emissions	2.34×10^{-5}	3.60×10^{-5}	1.70×10^{-5}	2.70×10^{-5}	9.78×10^{-6}	1.08×10^{-5}	kg·P-eqv.
CHP	Biomass production system	2.38×10^{-5}	3.67×10^{-5}	1.72×10^{-5}	2.74×10^{-5}	1.01×10^{-5}	1.12×10^{-5}	kg·P-eqv.
	CHP—Direct emissions	0	0	0	0	0	0	kg·P-eqv.
	Credits heat utilization	-4.46×10^{-9}	-4.46×10^{-9}	-4.46×10^{-9}	-4.46×10^{-9}	-4.46×10^{-9}	-4.46×10^{-9}	kg·P-eqv.
Total	**Total with credits**	**2.38×10^{-5}**	**3.66×10^{-5}**	**1.72×10^{-5}**	**2.73×10^{-5}**	**1.01×10^{-5}**	**1.12×10^{-5}**	**kg·P-eqv.**
	Total without credits	**2.38×10^{-5}**	**3.67×10^{-5}**	**1.72×10^{-5}**	**2.74×10^{-5}**	**1.01×10^{-5}**	**1.12×10^{-5}**	**kg·P-eqv.**

A net benefit in the impact category marine eutrophication was achieved for the utilization of switchgrass and maize only in the year 2012—where the yield was significantly higher than in 2013—and when the heat utilization was accounted for (Figure 6). Miscanthus was the only crop that led to a reduction of marine eutrophication in comparison to the fossil reference in all years and scenarios. The maximum reduction was—4.6×10^{-5} kg·N-eqv. (kWh$_{el.}$)$^{-1}$ (Figure 6).

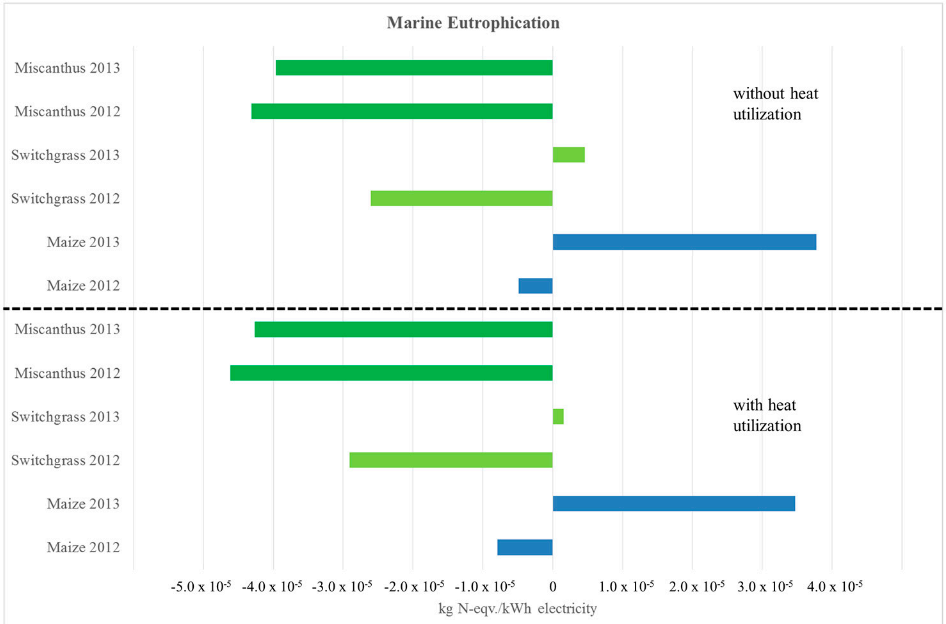

Figure 6. Assessment of the net benefits and impacts in kg·N-eqv. of substituting 1 kWh$_{el}$. of the German electricity mix by power generated via combustion of the biogas in a CHP.

The production of nitrogen fertilizer had the strongest impact on marine eutrophication for all crops, followed by fertilizer-induced emissions. Ammonia emissions and nitrate leaching due to the use of nitrogen fertilizer play a particularly important role here. Both impacts were highest for maize and lowest for miscanthus (Table 7). The recycling of nutrients results in a significant credit (Table 7).

Table 7. Assessment of the marine eutrophication in kg·N-eqv. of 1 kWh_{el}. generated via the production and fermentation of dedicated energy crops and combustion of the biogas in CHP.

	Processes/Flows	Maize per FU		Switchgrass per FU		Miscanthus per FU		Unit
		2012	2013	2012	2013	2012	2013	
Input substrates	Production of nitrogen fertilizer	2.60×10^{-5}	4.01×10^{-5}	1.70×10^{-5}	2.70×10^{-5}	1.13×10^{-5}	1.25×10^{-5}	kg·N-eqv.
	Production of potassium fertilizer	5.80×10^{-7}	8.92×10^{-7}	5.13×10^{-7}	8.14×10^{-7}	3.18×10^{-7}	3.51×10^{-7}	kg·N-eqv.
	Production of phosphate fertilizer	1.22×10^{-6}	1.87×10^{-6}	8.83×10^{-7}	1.40×10^{-6}	5.08×10^{-7}	5.60×10^{-7}	kg·N-eqv.
	Recycling of nutrients	-1.29×10^{-5}	-1.29×10^{-5}	-8.18×10^{-6}	-8.18×10^{-6}	-8.36×10^{-6}	-8.36×10^{-6}	kg·N-eqv.
	Herbicides	3.75×10^{-7}	5.76×10^{-7}	1.60×10^{-7}	2.54×10^{-7}	1.11×10^{-7}	1.22×10^{-7}	kg·N-eqv.
	Seeds/Rhizomes	1.89×10^{-6}	2.91×10^{-6}	1.06×10^{-6}	1.69×10^{-6}	1.64×10^{-6}	1.80×10^{-6}	kg·N-eqv.
Agricultural operations	Agricultural management	4.20×10^{-6}	6.46×10^{-6}	1.18×10^{-6}	1.87×10^{-6}	7.05×10^{-7}	7.77×10^{-7}	kg·N-eqv.
	Harvest	2.10×10^{-6}	3.23×10^{-6}	3.85×10^{-6}	6.11×10^{-6}	2.47×10^{-6}	2.72×10^{-6}	kg·N-eqv.
	Transport input substrates	2.97×10^{-7}	4.56×10^{-7}	2.16×10^{-7}	3.43×10^{-7}	1.50×10^{-7}	1.66×10^{-7}	kg·N-eqv.
	Transport biomass	2.97×10^{-6}	3.67×10^{-6}	2.86×10^{-6}	2.87×10^{-6}	2.72×10^{-6}	2.64×10^{-6}	kg·N-eqv.
	Ensilage	1.94×10^{-7}	2.98×10^{-7}	3.80×10^{-7}	6.03×10^{-7}	2.52×10^{-7}	2.78×10^{-7}	kg·N-eqv.
	Fertilizer-induced emissions	4.09×10^{-5}	6.29×10^{-5}	2.67×10^{-5}	4.25×10^{-5}	1.78×10^{-5}	1.96×10^{-5}	kg·N-eqv.
CHP	Biomass production system	6.78×10^{-5}	1.10×10^{-4}	4.67×10^{-5}	7.73×10^{-5}	2.96×10^{-5}	3.31×10^{-5}	kg·N-eqv.
	CHP-Direct emissions	4.58×10^{-6}	4.58×10^{-6}	4.58×10^{-6}	4.58×10^{-6}	4.58×10^{-6}	4.58×10^{-6}	kg·N-eqv.
	Credits heat utilization	-3.04×10^{-6}	-3.04×10^{-6}	-3.04×10^{-6}	-3.04×10^{-6}	-3.04×10^{-6}	-3.04×10^{-6}	kg·N-eqv.
Total	**Total with credits**	$\mathbf{6.94 \times 10^{-5}}$	$\mathbf{11.2 \times 10^{-5}}$	$\mathbf{4.82 \times 10^{-5}}$	$\mathbf{7.88 \times 10^{-5}}$	$\mathbf{3.11 \times 10^{-5}}$	$\mathbf{3.47 \times 10^{-5}}$	**kg·N-eqv.**
	Total without credits	$\mathbf{7.24 \times 10^{-5}}$	$\mathbf{11.5 \times 10^{-5}}$	$\mathbf{5.13 \times 10^{-5}}$	$\mathbf{8.19 \times 10^{-5}}$	$\mathbf{3.42 \times 10^{-5}}$	$\mathbf{3.77 \times 10^{-5}}$	**kg·N-eqv.**

3.3. Terrestrial Acidification

All scenarios led to higher terrestrial acidification than the fossil references. Maize without heat utilization performed worst and led to emissions of 3.5×10^{-3} kg·SO_2-eqv. $(kWh_{el.})^{-1}$ (Figure 7). Miscanthus performed best with the lowest terrestrial acidification potential (Figure 7).

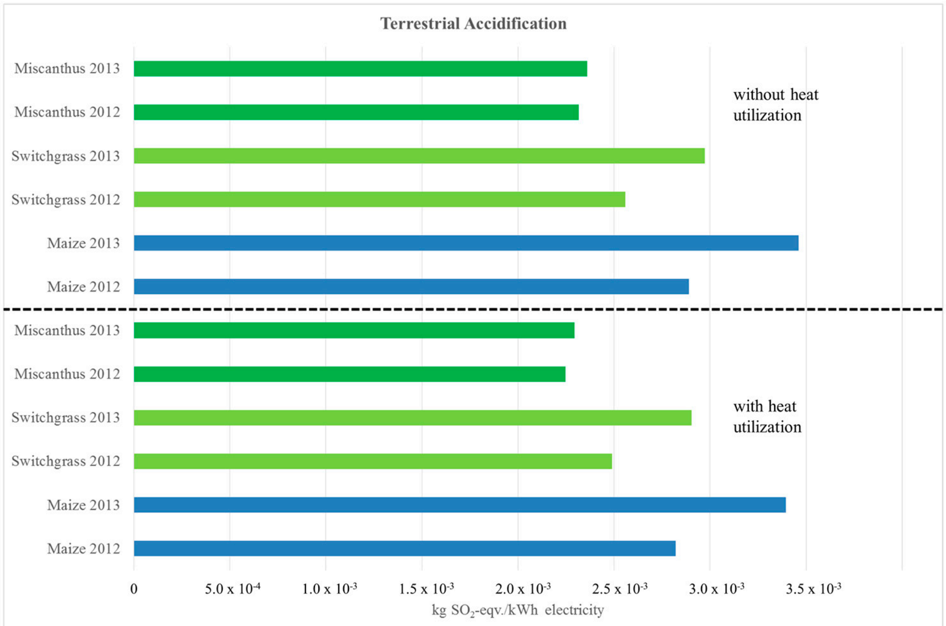

Figure 7. Assessment of the net benefits and impacts in kg·SO_2-eqv. of substituting 1 $kWh_{el.}$ of the German electricity mix by power generated via combustion of the biogas in a CHP.

Fertilizer-induced emissions—especially ammonia—had the highest impact on terrestrial acidification for all crops and accounted on an average for around 20% of total emissions (Table 8). The second largest source of emissions responsible for terrestrial acidification was production of nitrogen fertilizer, followed by transport of the biomass (Table 8).

Table 8. Assessment of the terrestrial acidification in kg·SO_2-eqv. of 1 kWh_{el}. generated via the production and fermentation of dedicated energy crops and combustion of the biogas in CHP.

	Processes/Flows	Maize per FU 2012	Maize per FU 2013	Switchgrass per FU 2012	Switchgrass per FU 2013	Miscanthus per FU 2012	Miscanthus per FU 2013	Unit
Input substrates	Production of nitrogen fertilizer	7.34×10^{-5}	1.13×10^{-4}	4.80×10^{-5}	7.63×10^{-5}	3.19×10^{-5}	3.52×10^{-5}	kg·SO_2-eqv.
	Production of potassium fertilizer	8.25×10^{-6}	1.27×10^{-5}	7.30×10^{-6}	1.16×10^{-5}	4.53×10^{-6}	5.00×10^{-6}	kg·SO_2-eqv.
	Production of phosphate fertilizer	4.73×10^{-5}	7.28×10^{-5}	3.43×10^{-5}	5.45×10^{-5}	1.97×10^{-5}	2.18×10^{-5}	kg·SO_2-eqv.
	Recycling of nutrients	-6.22×10^{-5}	-6.22×10^{-5}	-4.71×10^{-5}	-4.71×10^{-5}	-4.88×10^{-5}	-4.88×10^{-5}	kg·SO_2-eqv.
	Herbicides	6.35×10^{-6}	9.77×10^{-6}	2.71×10^{-6}	4.31×10^{-6}	1.88×10^{-6}	2.08×10^{-6}	kg·SO_2-eqv.
	Seeds/Rhizomes	2.19×10^{-6}	3.36×10^{-6}	8.04×10^{-7}	1.28×10^{-6}	1.98×10^{-6}	2.18×10^{-6}	kg·SO_2-eqv.
Agricultural operations	Agricultural management	5.16×10^{-5}	7.95×10^{-5}	1.46×10^{-5}	2.32×10^{-5}	8.73×10^{-6}	9.62×10^{-6}	kg·SO_2-eqv.
	Harvest	2.58×10^{-5}	3.97×10^{-5}	4.72×10^{-5}	7.50×10^{-5}	3.03×10^{-5}	3.34×10^{-5}	kg·SO_2-eqv.
	Transport input substrates	8.90×10^{-7}	1.37×10^{-6}	6.47×10^{-7}	1.03×10^{-6}	4.51×10^{-7}	4.97×10^{-7}	kg·SO_2-eqv.
	Transport biomass	3.69×10^{-5}	4.57×10^{-5}	3.56×10^{-5}	3.57×10^{-5}	3.39×10^{-5}	3.28×10^{-5}	kg·SO_2-eqv.
	Ensilage	2.46×10^{-6}	3.78×10^{-6}	4.83×10^{-6}	7.67×10^{-6}	3.21×10^{-6}	3.54×10^{-6}	kg·SO2-eqv.
	Fertilizer-induced emissions	8.29×10^{-4}	1.28×10^{-3}	5.42×10^{-4}	8.61×10^{-4}	3.61×10^{-4}	3.97×10^{-4}	kg·SO_2-eqv.
CHP	Biomass production system	1.02×10^{-3}	1.60×10^{-3}	6.91×10^{-4}	1.10×10^{-3}	4.48×10^{-4}	4.95×10^{-4}	kg·SO_2-eqv.
	CHP - Direct emissions	2.61×10^{-3}	2.61×10^{-3}	2.61×10^{-3}	2.61×10^{-3}	2.61×10^{-3}	2.61×10^{-3}	kg SO_2-eqv.
	Credits heat utilization	-6.82×10^{-5}	-6.82×10^{-5}	-6.82×10^{-5}	-6.82×10^{-5}	-6.82×10^{-5}	-6.82×10^{-5}	kg·SO_2-eqv.
Total	**Total with credits**	$\mathbf{3.57 \times 10^{-3}}$	$\mathbf{4.14 \times 10^{-3}}$	$\mathbf{3.24 \times 10^{-3}}$	$\mathbf{3.65 \times 10^{-3}}$	$\mathbf{2.99 \times 10^{-3}}$	$\mathbf{3.04 \times 10^{-3}}$	**kg·SO_2-eqv.**
	Total without credits	$\mathbf{3.64 \times 10^{-3}}$	$\mathbf{4.21 \times 10^{-3}}$	$\mathbf{3.31 \times 10^{-3}}$	$\mathbf{3.72 \times 10^{-3}}$	$\mathbf{3.06 \times 10^{-3}}$	$\mathbf{3.11 \times 10^{-3}}$	**kg·SO_2-eqv.**

4. Discussion

Here the results of this study are considered in a broader context, also including other environmental aspects not modeled in the LCA. The discussion concludes with opportunities and challenges of the introduction of novel perennial C4 crops in the biogas sector.

4.1. Environmental Performance in Impact Categories Modelled in the LCA

The results of this study show that, as soon as more impact categories are assessed than climate change and fossil fuel depletion, the environmental performance of the bioenergy conversion route "biogas" is not so clear-cut. All three energy crops have a significantly better environmental profile than the fossil reference (German electricity mix) in the impact categories climate change (CC) and fossil fuel depletion (FFD). Similar findings have been reported in the literature [45,46]. However, all three energy crops showed significantly higher impacts than the fossil reference in the impact categories freshwater eutrophication (FE) and terrestrial acidification (TA). The results for marine eutrophication (ME) were more variable. Here, miscanthus (both years) and switchgrass (2012 only) had a significantly lower impact than the fossil reference, whereas maize had a significantly higher impact in 2013 due to the low yield. High biomass yields have been shown to be a crucial factor for favorable environmental performance [47]. Again, these results correspond to findings of other studies, which mainly also found a higher impact of energy-crop-derived biogas than the fossil reference in acidification and eutrophication potential [48–50].

4.1.1. Overall Impact of Process Steps in Impact Categories

The production of nitrogen fertilizer was identified as the most relevant process step in the impact categories FFD and CC and the second most relevant in ME. Fertilizer-induced emissions were identified as the most important flow in the categories FE and ME and second most important in CC and TA. Similar results have been reported in the literature and numerous studies have already described the strong impact of nitrogen fertilizer production and related direct and indirect emissions on FFD and CC (e.g., [39,46,50,51]). The present study also showed a strong impact of mineral nitrogen fertilizer application on eutrophication (FE and ME) and acidification potential of crop production. This seems logical, since nitrate is one of the major contributors to eutrophication and the nitrification process a major contributor to soil acidification [27].

In TA, direct CHP emissions were the most important flow. Rehl et al. [49] identified sulfur dioxide from the CHP as one of the most important contributors to the acidification potential. One possibility to reduce these emissions could be the upgrading of biogas to biomethane, because sulfur dioxide is almost completely removed during this process. In addition, new techniques for biomethane production (e.g., pressurized anaerobic digestion) could help reduce the carbon footprint of biomethane production in the near future, because the demand for energy-intensive compression is reduced in such approaches [52]. Lijó et al. [53] reported production of nitrogen fertilizer, fertilizer-induced emissions and emissions of agricultural management as important factors for the environmental performance of energy crops. In this study, emissions from agricultural management were found to be the third most relevant process in CC, FFD and ME for maize cultivation, but considerably less important for miscanthus and switchgrass.

4.1.2. Impact of the Process Steps for Each Crop

Emissions and fossil fuel depletion from production of nitrogen fertilizer and agricultural management and fertilizer-induced emissions were highest for maize in each of the considered impact categories. This is because maize production consumes more energy for soil cultivation and requires higher nitrogen fertilizer levels for high yields than the C4 perennial grasses. For maize, data from the treatment with the highest nitrogen fertilization (240 $kg \cdot N \cdot ha^{-1}$) were used, which on long-term average yielded significantly higher than the medium fertilization rate (120 $kg \cdot N \cdot ha^{-1}$). However, the high nitrogen fertilization is probably above the marginal revenue and a lower fertilization rate could reduce the environmental impact of maize. Nevertheless, the nitrogen demand of miscanthus and switchgrass are still lower than that of maize. In addition, for miscanthus and switchgrass, data from the treatment with the highest nitrogen fertilization rate (80 $kg \cdot N \cdot ha^{-1}$) were used, in order to consider the higher nutrient removal by the green harvested biomass. Although green harvest increases the withdrawal of nitrogen compared to a spring harvest, the biomass of miscanthus and switchgrass contained approximately 60% less nitrogen than maize biomass (Table 3).

The annual cultivation of maize led also to significantly higher emissions and fossil fuel depletion for agricultural management in CC, ME and FFD. For this reasons, changing the crop production system from annual crops with a high nitrogen demand to perennial C4 crops with improved nutrient efficiency seems to be a very promising option for increasing the environmental sustainability of the biogas sector and the bioeconomy, as already described by Lewandowski [54]. Compared to maize, miscanthus and switchgrass showed in the scenarios without heat utilization 59%–73% and 25%–28% lower CC potential, 68%–79% and 28%–30% lower FFD potential, 57%–69% and 25%–28% lower FE potential, 53%–67% and 29% lower ME potential and 16%–26% and 9%–12% lower TA potential, respectively.

Considering all impact categories, miscanthus performed best amongst the three assessed crops. Especially in 2013, the yield and thereby the environmental performance of miscanthus was much more stable compared to maize and switchgrass. Both crops reacted more sensitively to the unfavorable weather conditions in 2013. This resulted in lower yields and is also reflected by the performance in the environmental impact categories. The higher stress tolerance and yield stability of miscanthus is therefore not only favorable for the farmer, but also from an environmental point of view.

The nutrient recycling via fermentation residues led to a significant credit for all crops, especially in the impact categories CC, FFD and ME. However, fermentation residue application on the perennial grasses miscanthus and switchgrass and resulting emissions need to be further investigated. Since the fermentation residues cannot be incorporated into the soil in such perennials, higher ammonia emissions could occur, which could lead to higher eutrophication and acidification potentials [48]. This needs to be further investigated to allow consideration of such an effect in future assessments of the environmental performance.

4.2. Other Environmental Aspects

In the section above, the environmental performance was analyzed in five impact categories and it was shown that the perennial grasses, especially miscanthus, performed better than the annual crop maize. However, the five considered impact categories are not sufficient for a holistic assessment of the environmental performance. Therefore, other aspects relevant to environmental performance are discussed in the following section.

Intensive soil cultivation in annual maize is accompanied by an increased risk of soil erosion, due to the slow youth development of the crop [6]. For annual maize, there is also a low to medium risk for soil compaction [55]. However, for green-harvested miscanthus and switchgrass the risk of soil compaction may be lower due to its perennial nature, but needs to be assessed to allow comparison. The combination of intensive soil cultivation and low amount of crop residues in silage maize has a negative impact on content of soil organic carbon. Both environmental aspects could be improved by changing substrate supply of biogas plants from maize to perennial C4 grasses, since miscanthus and switchgrass generally lead to an increase in soil organic carbon compared to annual cropping systems [11,56,57]. Under miscanthus, the largest proportion of the soil organic carbon is found in the topsoil, which can be explained by the high proportion of roots in the top 0.35 m [58]. The sequestration of carbon in the soil can increase the GHG mitigation potential significantly, especially if the cropping system is changed from annual to perennial [56,59]. In this study, the sequestration effect was not considered, because the effect of the green harvest on the root and rhizome development and on the soil carbon sequestration potential is not yet known. Therefore, the development of the soil organic carbon under green harvested miscanthus and switchgrass needs to be further investigated to determine the sequestration potential of this harvest regime.

Agricultural land occupation is another important environmental aspect, due to limited expansion potential for agricultural land and negative impacts from the transformation of natural land. In this paper, agricultural land occupation was not directly assessed, but the data in Table 1 show that maize required the smallest area (173 ha) of agricultural land in 2012 to supply the biogas plant with the required biomass. Changing the input substrate from maize to miscanthus or switchgrass increased the agricultural land demand in 2012 by 12% or 68%, respectively. Under unfavorable weather conditions in 2013, the agricultural land demand for miscanthus cropping was 20% lower and for switchgrass 73% higher than for maize cultivation. Agricultural land occupation for biogas production can lead to indirect land-use change (iLUC), which can significantly reduce the GHG mitigation potential and even lead to higher GWP than the fossil reference [14]. For this reason, the comparatively high agricultural land demand of switchgrass to deliver the required biomass substrate is a clear disadvantage compared to the other crops. In contrast, the area demand of miscanthus was only slightly higher and even lower when unfavorable weather conditions occurred for maize production. Again, the higher abiotic stress tolerance and yield stability of miscanthus can be seen as environmental advantage. However, both perennial C4 crops could be grown in future mainly on marginal or contaminated land [9,23]. This could reduce the pressure on agricultural land and expand the area available for biomass production.

Biodiversity is difficult to assess just by the crop itself, because it strongly depends on other factors, e.g., the distribution of fields in a landscape and structural elements such as hedges. However, modern agriculture is assumed to have a negative impact on the biodiversity by simplification of agricultural landscapes, e.g., large field sizes, and small amount of crop varieties which are grown in monoculture [4]. An increased number of crop species and a higher proportion of perennial cropping systems in modern agriculture is seen as one option to promote biodiversity [4]. For this reason, replacing biogas maize with miscanthus or switchgrass could positively affect the biodiversity by adding novel, perennial crops to the agricultural landscapes. However, it should be noted that the impact on soil biodiversity may be influenced by the choice of the perennial biomass crop [60]. Furthermore, both perennials can be characterized by their comparatively low-input crop management, after their successful establishment in year one. For miscanthus, a higher abundance of insects, spiders and earthworms than in arable land is reported, as well as additional niches for birds and, provided a spring harvest is performed, over winter cover for small mammals in intensive arable regions [11,61]. For switchgrass, similar positive effects can be expected, which leads to the assumption, that both could increase the biodiversity and structure-richness of agricultural landscapes. Again, the effect of the pre-winter harvest, which clearly removes the winter cover for small mammals and reduces the mulch layer, is not yet known and needs to be investigated. However, both crops also induce risks for biodiversity because they are not native to Europe and could potentially appear as invasive species. *Miscanthus x giganteus* has a very low invasiveness risk, because it does not produce fertile seeds and no escapes were observed over more than two decades of *M. x giganteus* production in Europe. Current miscanthus breeding efforts aim to produce fertile genotypes that can be propagated by seeds [10], but several mechanisms to avoid seed escape are incorporated, including preferring candidates which require a very long vegetation period for seed production to avoid viable seeds being produced in regions of biomass cultivation [9]. It is also necessary to mention that miscanthus as well as switchgrass seedlings have a very low competitiveness compared to weeds and a slow youth development. For this reason it is quite unlikely that they become invasive species in Europe. Nonetheless, the invasiveness potential of novel miscanthus genotypes and switchgrass needs to be investigated and monitored.

Finally, the socioeconomic aspects of landscape appearance need to be considered. Crops such as maize are often criticized in the public, due to their height and monotony. The same could appear for miscanthus, due to its height and density in well-established commercial fields. Smaller and nicely flowering miscanthus genotypes or switchgrass could be experienced more favorably and might influence the appearance of landscapes more positively. However, this could compromise the yield and lead to a trade-off between yield and public acceptance. Public acceptance could also be positively influenced by using smaller fields or strip cropping instead large monoculture fields.

4.3. Implementation—Chances and Challenges

In this study, it is shown that implementation of perennial C4 grasses for biogas production can have significant environmental benefits. From an environmental point of view, miscanthus in particular would be a desirable crop for biogas production. The main weak point of switchgrass is clearly its lower yield potential than miscanthus and related to that its higher area demand, fossil fuel consumption and emissions. For the farmer, the implementation of miscanthus and switchgrass as biogas crops is accompanied by opportunities and challenges, which are discussed in the following section but require further research.

This study is based on methane yields measured in a batch test using milled biomass. In order to transfer these values to a full-scale biogas plant, a pre-treatment of the biomass was considered for miscanthus and switchgrass, which leads to a higher electricity demand for plant operation. For this reason, the electricity demand for miscanthus and switchgrass was assumed to be almost twice as high as that for maize. Before implementation, the methane yield, the necessity of a pre-treatment and the energy consumption of such a pre-treatment should be verified under more realistic conditions. Ensiling of miscanthus biomass, and presumably also switchgrass, appears possible [19], but also needs to be demonstrated in practice.

The long-term performance of green-harvested miscanthus is one of the major uncertainties for its biogas utilization, because miscanthus reacts sensitively to very early mid-season harvest, but tolerates green harvest in late autumn [18]. However, it is not yet known if green-harvested miscanthus is productive for as long as a spring-harvested crop (more than 20 years) and if recycling of fermentation residues is sufficient to maintain its productivity. In addition, the farmer has to dedicate arable land to miscanthus for several years to achieve return on investment, due to the high establishment costs. However, current research focuses on reducing establishment costs by developing seed-based genotypes, which may allow direct sowing in future [10]. Further, most biogas plants are designed for a minimum of 20 years' operation, which would fit in very well with the expected productive lifetime of miscanthus. Cost-effective miscanthus establishment offers the chance of significantly reducing biomass costs. As shown in this paper, the yield of miscanthus is not as sensitive as annual maize to unfavorable weather conditions, which may become more common in future due to climate change. One of the main reasons for the low maize yield was the very late sowing date and the early summer drought stress. In miscanthus, planting is only required once in 20–30 years and the established crop benefits from winter soil moisture. Therefore, miscanthus seems very suitable for risk mitigation of such weather conditions.

In contrast to miscanthus, switchgrass can be established cheaply via direct sowing of seeds. However, the establishment of switchgrass is difficult due to an often low germination rate, low competitiveness of seedlings and limited availability of herbicides. Current research focuses on the optimization of the establishment method and herbicide testing [62]. Nevertheless, early green harvest of switchgrass seems less problematic than in miscanthus and even a double cut is possible [21]. The shorter productive life of approximately 15 years, lower investment costs and the ability of direct sowing may increase farmers' willingness to adopt this crop. However, the lower yield potential limits its implementation to very poor and shallow soils, where it is likely to perform better than miscanthus [23].

From an environmental point of view, miscanthus cultivation for biogas production is generally recommended if the biogas plant technology is suitable for the digestion of fibrous substrates or adequate pre-treatment options are available.

Supplementary Materials: The following are available online at www.mdpi.com/2071-1050/9/1/5/s1, Table S1: Climate change in kg CO_2-eqv. per kg DM biomass, Table S2: Freshwater eutrophication in kg P-eqv. per kg DM biomass, Table S3: Fossil fuel depletion potential in kg oil-eqv. per kg DM biomass, Table S4: Marine eutrophication potential in kg N-eqv. per kg DM biomass, Table S5: Terrestrial acidification potential in kg SO_2-eqv. per kg DM biomass.

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4. General Discussion

In the general discussion the results presented in the chapters 2 and 3 are critically scrutinized and discussed in the context of the research questions raised in the introduction of this thesis.

The first research question deals with the identification of the main parameters, which are most relevant for the environmental performance of perennial crop-based value chains. In this thesis the following three key parameters were determined as most relevant: yield (Meyer *et al.* 2017), fertilizer-induced emissions (Wagner & Lewandowski 2017) and the selection of the reference system (Wagner *et al.* 2017). However, the assumptions made and the input data used for these key parameters vary widely in present LCA studies, which assess the environmental performance of perennial crop-based value chains. In chapter 4.1 the influence of those choices on the results of these assessments are discussed and recommendations are elaborated.

In the second research question the issue was raised, which impact categories have to be included in order to holistically assess the environmental performance of perennial crop-based value chains. In chapters 2.2 and 3.1 the relevant impact categories were determined for different utilization pathways using a normalisation approach (Wagner *et al.* 2017; Wagner & Lewandowski 2017). The use of a normalisation step to determine the most relevant impact categories is frequently applied by LCA practitioners (Ahlroth *et al.* 2011; Pizzol *et al.* 2016). In general, it can be concluded that the normalisation approach is a suitable method to estimate the significance of different impact categories. The results of this approach show, that it is crucial to include more impact categories than just the global warming potential (GWP). However, there are some drawbacks associated with the use of a normalisation approach in regard to the uncertainty and the robustness of the results (Pizzol *et al.* 2016). These drawbacks are discussed in chapter 4.2, where the selection of the normalisation factors (NFs) applied in this thesis is critically reflected. In addition, the influence of the geographic scale of the NFs and of the preload of the environment is elaborated.

However, even as it was shown which of the available impact categories are relevant and have to be included, there are still some environmental impacts missing, which could be particularly relevant for the environmental evaluation of perennial crops. In chapter 4.3 a brief overview is given of environmental impacts of land-use which are missing, such as on soil quality and biodiversity. It is also elaborated, how these impact can be included in future assessments of bio-based value chains.

The third research question addresses the differences in the environmental performance of various miscanthus-based value chains. The results show that the cascade use of biomass, first as material and later as an energy carrier, such as in the case of miscanthus-based insulation material, has the best environmental performance of all analysed miscanthus utilization pathways (Wagner *et al.* 2017). The fifth research question additionally deals with the comparison of perennial with annual crop-based value chains. In the analysed utilization pathway biogas production, the assessed perennial crops have lower impacts on the environment compared to annual crops (Kiesel *et al.* 2017). However, the effect of carbon sequestration was not included in these assessments. The inclusion of this effect could further improve the environmental performance of perennial-crop based value chains, and in particular those value chains which use biomass to produce bio-based materials. In chapter 4.4 the role of carbon sequestration in carbon mitigation and its influence on the environmental performance is discussed. Here, carbon sequestration is understood in a broader context including sequestration in the soil as soil organic carbon (SOC) and in bio-based products.

The fourth research question raised the issue if it makes sense from an environmental point of view to cultivate perennial crops on marginal land and thereby reduce the land-use competition with food or feed crops. It was demonstrated that even on marginal sites, with low yield potentials, miscanthus-based value chains showed a relatively good environmental performance (Wagner *et al.* 2017). In chapter 4.5, a focus is on exploring the possible drawbacks of the utilisation of these, until now abandoned sites, and possible mitigation options are shown. In addition, the possibility to use contaminated besides economically marginal land is elaborated.

4.1. The influence of data used on the LCIA results and the importance of high data quality

As demonstrated by Meyer *et al.* (2017), Wagner *et al.* (2017) and Wagner & Lewandowski (2017) the biomass yield is strongly correlated with the environmental performance of bio-based value chains. When the yield increases, the negative impact on the environment decreases and in the same time the CO_2 emission mitigation potential increases. The influence of the yield is much more pronounced than the influence of the biomass composition (Meyer *et al.* 2017). However, most studies assessing the environmental performance of perennial crop-based value chains rather use average or estimated and not measured yields (Murphy *et al.* 2013; Jeswani *et al.* 2015; Styles *et al.* 2015). This increases the uncertainty associated with the results. This is particularly important in the assessment and comparison of the environmental performance of different perennial crops, or of annual and perennial crops. Therefore it is crucial to obtain the field data for the cultivation process in the LCA under *ceteris paribus* conditions. If the two crops are not grown under similar conditions there is the risk to compare crops cultivated on different types of land (good agricultural land/marginal sites), or under different agricultural management practices. This would lead to non-comparable yield potentials and thus results in misleading conclusions about their environmental performance.

An important factor affecting the environmental performance of bio-based value chains, and here in particular the impact category climate change, is the fertilizer use (Wagner & Lewandowski 2017). The environmental impacts associated with the fertilizers applied are caused by upstream and on-field emissions. The upstream emissions are due to the energy-intensive production of mineral fertilizer (Tzilivakis *et al.* 2005) and are thus strongly correlated with the amount of fertilizers applied. In addition to the upstream emissions the on-field fertilizer-induced emissions, such as N_2O, have a significant impact on the environmental performance. For example, it was shown for biodiesel that the direct and indirect N_2O emissions account for 20-40% of the total GWP (Dufossé *et al.* 2013). However, there is a considerable uncertainty concerning the amount of N_2O which is emitted, as the emissions are depending on several factors such as the soil type and the kind of fertilizer used (Bouwman *et al.* 2002a; Yan & Boies 2013). In LCA studies, the IPCC standard value of 1% N_2O-N losses per kg N applied (IPCC 2006) is often used (Godard *et al.* 2013; Murphy *et al.* 2013; Nguyen & Hermansen 2015) to analyse bio-based value chains. This value though was calculated from a global data set and therefore does not consider site-, crop- or fertilizer-specific information (IPCC 2006). In order to reduce the uncertainty associated with the N_2O emissions, it is strongly recommended to apply more specific emission factors. Bouwman *et al.* (2002b) for example described several fertilizer type-specific emission factors. The emission factor for ammonium sulphate is in line with the value proposed by the IPCC. However, for calcium ammonium nitrate (CAN) the emission factor of 0.7% N_2O-N losses per kg N differs considerably (Bouwman *et al.* 2002b). The use of nitrification inhibitors could further reduce the nitrogen fertilizer-induced N_2O emissions by around 38% (Akiyama *et al.* 2010). This point again emphasizes the necessity of using specific data in the assessment of the environmental performance of bio-based value chains as the type of fertilizer used can have a significant impact on the results.

In addition, to the uncertainty concerning the N_2O emissions, the data used for the amount of fertilizers applied varies widely in LCA studies, which analyse the environmental performance of miscanthus-based value chains. The fertilizer quantity dispensed range for example from 45 kg N / 20 kg P_2O_5 (Godard *et al.* 2013) to 90 kg N / 250 kg P_2O_5 (Monti *et al.* 2009) per hectare. Even when including the varying yield assumptions, the differences are still very large. This leads to a further uncertainty regarding the upstream but also the on-field emissions. In particular in case of phosphate fertilizer, which has a significant impact on the eutrophication potential through the run-off and leaching of phosphorus (Wagner & Lewandowski 2017). For LCA practitioners the recommendation for future studies would be to use the fertilizer amount which is realized in field trials or in practice as input data. If such data is not available, the fertilizer amount used in the assessments should be based on the nutrient withdrawal through the harvested biomass. Hereby fertilizer application rates of around 52 kg N ha^{-1} yr^{-1} and 27 kg P_2O_5 ha^{-1} yr^{-1} should be sufficient for a harvest in late winter. These values were calculated based on a dry matter biomass yield of around 20 t ha^{-1} yr^{-1} (Kiesel & Lewandowski 2017).

In order to determine the environmental impact mitigation potential of different perennial crop-based value chains, it has to be assessed which reference system will be substituted through these value chains. The choice of the reference system can have a profound impact on the results. There are for example substantial differences if miscanthus-based heat is substituting heat generated by the combustion of light fuel oil or heat generated by the combustion of natural gas (Wolf *et al.* 2016; Wagner *et al.* 2017). The generation of 1 MJ_{th} by the combustion of light fuel oil leads to 94 g CO_2 eq., while the generation of heat by the combustion of natural gas causes only to 33 g CO_2 eq. per MJ_{th}.

In addition, it was shown for several utilization pathways that in various impact categories the substitution process of the reference system had a far larger impact on the environmental impact mitigation potential than the production process of the bio-based product itself (Wagner *et al.* 2017). That is mostly due to the emission and energy-intensive production process of the reference system. An example for that is the energy-intensive production of glass wool mats, which are substituted by miscanthus-based insulation material (Wagner *et al.* 2017). The production of 1 m^3 of miscanthus-based insulation material leads to a freshwater eutrophication of 0.043 kg P eq. (excluding the End-of-Life phase). However to produce an amount of glass wool mats with a comparable function (that is with the same functional unit) 0.145 kg P eq. are emitted (Wagner *et al*. 2017). If the same amount of miscanthus biomass, which is necessary to produce 1 m^3 of insulation material (194.3 kg biomass dry matter based), is used to generate heat, 0.022 kg P eq. are emitted. In comparison, the same amount of heat generated through the combustion of light fuel oil only leads to emissions of 0.009 kg P eq. (Wagner *et al.* 2017).

As shown above the substituted reference system has a significant impact on the environmental performance of the assessed value chains. Hereby it is crucial for LCA users while comparing the impact on the environment of perennial crop-based value chains to select the references system, which is most likely to be replaced in practice by the bio-based alternative. The most probable substituted reference system is the marginal producer and thus the economically least competitive one (Schmidt 2008).

Government officials or public agencies which plan measurements for supporting the introduction of bio-based value chains should select the product to be substituted not only out of economic considerations, but also by taking into account environmental aspects. Thus selecting a product for which a substitution through a perennial crop-based alternative also yields a relatively high environmental impact mitigation potential. However it has to be considered, that the selection of the appropriate reference system and especially the most suitable value chain are always strongly depending on the local circumstances. These are besides the site-specific demand for the different bio-based products (including bio-based energy carriers), in particular the availability of biomass and utilization options.

4.2. Normalisation – An appropriate approach to assess the relevance of impact categories?

According to the ISO standard 14044 "*the aim of the normalization is to better understand the relative magnitude for each indicator result of the product system under study*" (ISO 2006). In the normalisation step, the result for the potential environmental effect in an impact category is divided by a reference value. The reference value or normalisation factor (NF) can either be a.) a baseline scenario (e.g. an alternative product system); b.) the total inputs and outputs for a certain area (global, regional, local level); or c.) the total inputs and outputs per capita basis for a certain area (ISO 2006). The last two approaches are the most accepted and used in the LCA community (Laurin *et al.* 2016). In the studies of Wagner *et al.* (2017) and Wagner & Lewandowski (2017), which are contained in this thesis, NFs are applied based on the emissions caused by an average European (25+3) citizen in the year 2000 (Goedkoop *et al.* 2009). As various bio-based value chains with different end products were assessed in this thesis, it was not possible to define a common baseline scenario. That is why NFs based on an alternative product system were inapplicable. NFs per capita basis were used to enable a meaningful comparison with other regions independent of their area size. In addition it allows a more coherent presentation of the results.

An important factor influencing the normalised results, is the spatial resolution of the normalisation factors. Slapnik *et al.* (2015) showed that country specific NFs for a state in Europe can differ significantly from the European NFs. Furthermore, if NFs based on the global instead of the European population are used, the normalisation can lead to quite different results. In the current studies European NFs were used because the goal of the study was to assess and compare the analysed bio-based value chains in a European context. That is why averaged European normalisation factors were deemed most appropriate, so that the results of the environmental performance of the bio-based value chains are not country specific, but can be applied across Europe. However, if different utilization options for biomass are compared for a specific site, or if it is analysed which site is the most suitable location for a selected utilization pathway it makes sense for some impact categories to use more regionalized NFs. In Figure 4.1, various impact categories are classified according to their relevance for different geographic scales. The classification of the impact categories is based on Stranddorf *et al.* (2005). In case of impact categories relevant on a global scale, such as climate change, global NFs should be used. For the categories which impact the environment on a regional or local scale, national or when available even regional NFs are deemed most appropriate.

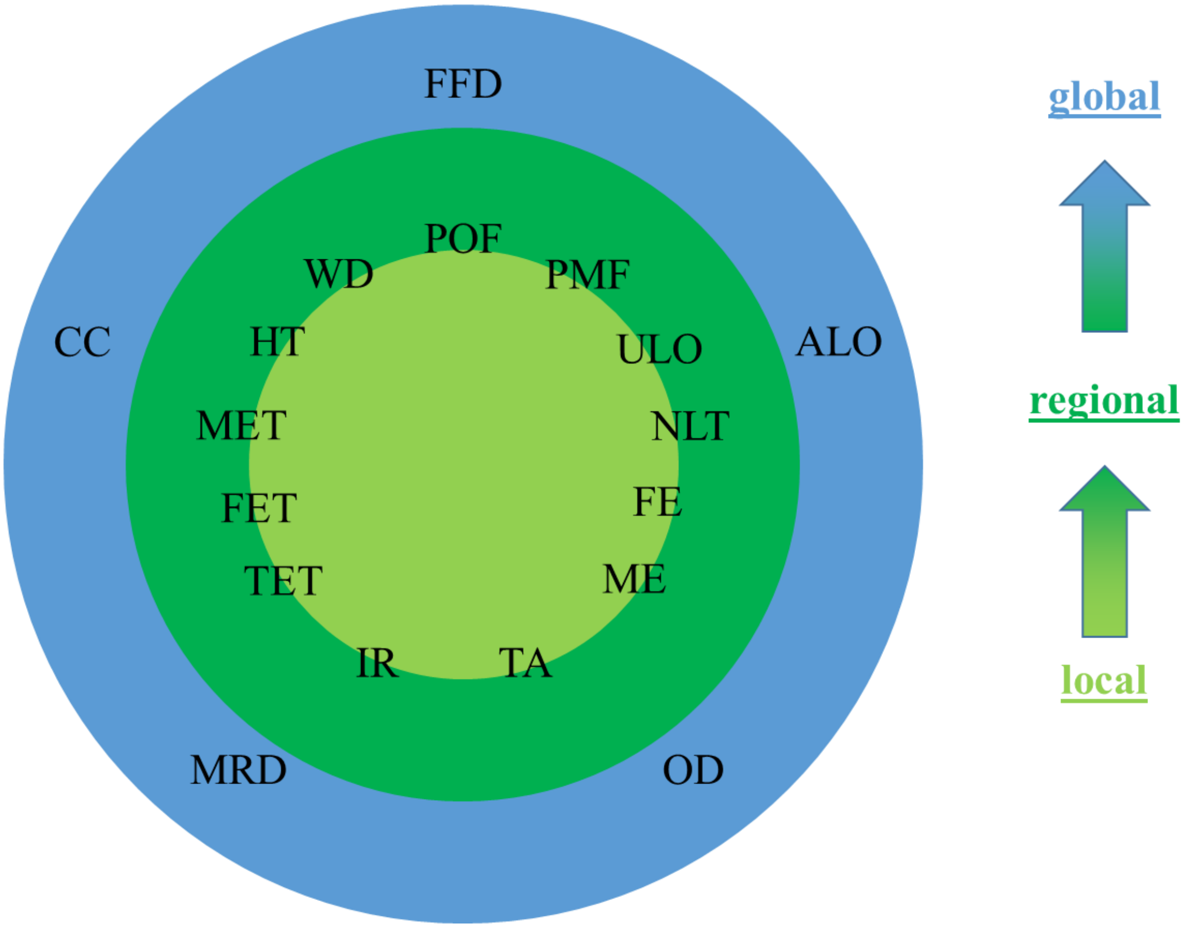

Figure 4.1: Geographic scale at which the selected impact categories should be applied (FFD: Fossil Fuel Depletion, ALO: Agricultural Land Occupation, OD: Ozone Depletion, MRD: Mineral Resource Depletion, CC: Climate Change; POF: Photochemical Oxidant Formation, PMF: Particulate Matter Formation, ULO: Urban Land Occupation, NLT: Natural Land Transformation, FE: Freshwater Eutrophication, ME: Marine Eutrophication, TA: Terrestrial Acidification, IR: Ionising Radiation, TET: Terrestrial Ecotoxicity, FET: Freshwater Ecotoxicity, MET: Marine Ecotoxicity, HT: Human Toxicity, WD: Water Depletion)

The preload of the environment is another parameter which influences the assessment of the relevance of impact categories, but is not taken into account in the normalisation approach. As explained above the normalisation factors are based on the average emissions of a European citizen in the year 2000 (Goedkoop *et al.* 2009). In the normalisation step, the emissions of the respective value chain are calculated in relation to this amount (ISO 2006). That means when the environmental impact in one category is already quite high, and thus perhaps already critical for the environment, a further increase could be deemed not relevant using a normalisation approach. However, this is based on the preload of the environment and thus misleading as it tells nothing about the relevance of the impact. In contrast to this, terrestrial acidification might not be the most pressing problem at a site, where the initial acidification is low and the buffer capacity of the soil is high, even when it is indicated as relevant impact category in the normalisation step.

A solution to include the preload of the environment could be the use of a distance-to-target (DTT) approach. This approach is normally classified as a weighting approach, but can also be used as an external normalisation method (Pizzol *et al.* 2016). Through this approach, the characterization results are related to target levels, which are based either on the carrying capacity or on policy targets. Castellani *et al.* (2016) developed an EU related DTT weighting set, which is based on the distance of the domestic EU impacts from the desired targets. These desired targets are either based on binding or non-binding EU policy goals. This DTT weighting set could be used to further strengthen the robustness of the determination of the relevance of various impact categories. However, the influence on the relevance of different impact categories is relatively small. The application of the DTT weighting only changed the normalised results set for example of the impact category freshwater eutrophication by the factor 1.01 and of the impact category marine eutrophication by the factor 1.13 (Castellani *et al.* 2016). This is independent of the value chain analysed. That means that the normalised results in these categories are slightly higher using the DTT approach but only by 1% and 13%, respectively. Such minor differences do not change the assessment of the relevance of the different impact categories. In addition, a further weighting step, such as the use of a distance-to-target (DTT) approach, increases the uncertainty associated with the results (Pizzol *et al.* 2016). That is why it is advised against the use of currently available DTT weighting sets when analysing the environmental performance of perennial crop-based value chains.

In summary, it is recommended to include, where possible, various impact categories (such as all recommended by the Joint Research Centre of the European Commission (Hiederer 2011)) and assess the relevance of the different impact categories for a specific value chain using a normalisation approach. The information about the relevance, should then be used to focus on the most relevant categories for further analysis. The inclusion of various impact categories is advised in order to not omit an impact category mistakenly for example based on the preload as explained above. As LCA is always an iterative process, it is recommended to calculate in a first step a rough approximation of the environmental performance using for example standard emission factors. Based on that assessment the relevance of the analysed impact categories is determined. In a second round the effort such as the data gathering is then concentrated on processes or emission sources important for these categories in order to improve the accuracy of the results. This can be demonstrated using as example the impact category marine eutrophication (ME), for which nitrate leaching due to the use of nitrogen fertilizers is an important emission source (Wagner & Lewandowski 2017). In value chains, where the normalisation step shows that ME could be a relevant impact category, more site-specific models should be used to assess the nitrate leaching such as the SQCB – NO_3 model described by Faist Emmenegger *et al.* (2009). It includes more specific information regarding soil parameters and crop type, compared to the standard emission factors of the IPCC (IPCC 2006).

4.3. Missing impact categories for the holistic assessment of bio-based value chains

It was shown in several studies that perennial crops, and miscanthus in particular, have a positive impact on the biodiversity (Semere & Slater 2007a, 2007b) and on the soil quality (Kahle *et al.* 2001; Das *et al.* 2016; McCalmont *et al.* 2017) when compared to annual crops. The cultivation of miscanthus for example leads to an increase in the soil organic carbon (SOC) (Chimento *et al.* 2016; Gauder *et al.* 2016; McCalmont *et al.* 2017) as well as in the soil organic matter (SOM) (Beuch *et al.* 2000; Kahle *et al.* 2001). In addition, it positively affects the composition of the SOM (Kahle *et al.* 2001). Further benefits are an improved microbial activity as well as soil porosity, and a reduced bulk density (Holland *et al.* 2015). But there are also negative effects associated with the cultivation and utilization of perennial crops. Especially when using marginal land, which often has a high degree of biodiversity (Dauber & Miyake 2016), the agricultural use has negative impacts on the species richness (Dauber *et al.* 2015).

Despite the importance of such land-use related impacts on the environment, they are to a great extent not yet included in current LCA studies, which assess the environmental performance of bio-based value chains (Milà i Canals *et al.* 2006). In order to holistically assess the environmental impacts, especially of perennial crops, the impact of land-use on the soil quality and biodiversity should be included.

There are several approaches which try to estimate the impact of land-use on soil quality in LCA (Garrigues *et al.* 2012). These range from one-indicator approaches, which use for example SOM (soil organic matter) as an indicator for soil quality (Milà i Canals *et al.* 2007; Morais *et al.* 2016), to multi-indicator models like the Swiss Agricultural Life Cycle Assessment for Soil Quality (SALCA-SQ) which includes nine indicators encompassing chemical, physical and biological aspects of the soil (Oberholzer *et al.* 2012). However, there is criticism that with the currently available models it is not yet possible to assess the impact of land-use and land-use change on soils in a robust and comprehensive way. Criteria of such a robust and comprehensive assessment would be according to Vidal Legaz *et al.* (2017) the completeness of the scope, the environmental relevance, scientific robustness and certainty, or the documentation, transparency and reproducibility. In case of the SALCA-SQ model, for example, not all modelled results were consistent with the observed impacts. In addition, the input data requirements are very high and thus hindering the practical implementation (Vidal Legaz *et al.* 2017). Nevertheless, because of the importance of the soil as the foundation of all agricultural production processes and as a non-renewable resource it is recommended to integrate the impact of land-use on soil quality in assessments of perennial-crop based value chains. In order to do that, it is advised to apply from the currently available models the method developed by Milà i Canals *et al.* (2007), which is also recommended in the ILCD handbook (European Commission 2010). This method accounts for changes in SOM due to land-use transformation and occupation impacts. According to Brandão & Milà i Canals (2013) SOM "*is probably the most cited indicator of soil quality within soil science research*". SOM serves the soil biota as an energy as well as a food source (Brandão & Milà i Canals 2013) and thus functions as an indicator for the soil life activity. Furthermore, the SOM content is strongly related to other soil quality indicators, for example the cation exchange capacity. In addition, it is partly related to erosion protection through the higher aggregate stability and water infiltration associated with an increase in SOM, which reduces the vulnerability of the soil to erosion. However, it does not include the vegetation cover, which has an import influence on the soil erosion (Milà i Canals *et al.* 2007). According to Milà i Canals *et al.* (2007), the SOM content in the soil can be estimated from the SOC content. The SOC content can be determined by direct measurements or it can be calculated using models adjusted to the site-specific conditions. In addition, the authors provided literature values for potential yearly carbon sequestration rates in cropland for different agricultural management practices such as reduced and conservation tillage (Milà i Canals *et al.* 2007). The changes in the SOM respectively SOC content are assessed per hectare. However, through the biomass yield it is possible to relate the results to the bio-based product produced and thus to the functional unit. The indicator proposed by Milà i Canals *et al.* (2007) only assesses impacts of land-use on the soil quality and the related Life Support Functions (LSFs), such as the biotic production potential (Brandão & Milà i Canals 2013). It does not include the impact of land-use on the biodiversity (Milà i Canals *et al.* 2007).

As already mentioned in the introduction, miscanthus has in comparison to annual plants a positive impact on the biodiversity such as on the abundance of invertebrate populations

(Semere & Slater 2007b) as well as weed vegetation (Semere & Slater 2007a) and on farmland bird populations (Bellamy et al. 2009). A study, which analysed the impact of miscanthus cultivation on small mammals and birds highlighted the positive impact due to the low intensity of the agricultural management such as the soil cultivation. The positive impact is more distinctive when a harvest in late winter is applied, because then there are no disturbance of the miscanthus stands in summer and fall (Semere & Slater 2007a). However, there are still some major obstacles to be overcome until the impact of land-use on biodiversity in LCA can be assessed in a robust and consistent way. One major obstacle for example are missing indicators in the biodiversity assessment methods such as the functional diversity (Souza *et al.* 2015; Teillard *et al.* 2016). Until more comprehensive assessment methods for the impact of land-use on the biodiversity are available, it is recommended to assess the impact on the biodiversity applying approaches which use the species richness as indicator (Teixeira *et al.* 2016) such as Baan *et al.* (2013) and Chaudhary *et al.* (2015). The method developed by Chaudhary *et al.* (2015) allows the spatially explicit assessment of the total impact of land-use (occupation and transformation) on the biodiversity related to a functional unit. Their methods enables the quantification of the regional species loss in 804 terrestrial ecoregions due to land occupation and transformation for six different land-use types and five taxa. However, it has the limitation that it only contains two land-use types for agriculture: *agriculture, arable* and *agriculture, permanent crops*. The land-use type *agriculture, permanent crops* for example encompasses all perennial crops such as perennial grasses and woody perennial. This impedes any comparison between different perennial agricultural systems or crops. The same also applies to annual crops. In addition, there are several agri-environment schemes which could enhance the farmland biodiversity such as wildflower strips (Dicks *et al.* 2014; Tschumi *et al.* 2016), hedgerow trees (Vickery *et al.* 2004) and no-tillage agriculture (Dicks *et al.* 2014). Until now it is not yet possible to integrate the effects of such specific measurements in the assessment of the impact of the land-use on the biodiversity. This clearly shows that these assessment methods have to be further developed to enable the inclusion of different land-use types and measurement-specific effects.

Based on the tremendous importance of the soil quality and the biodiversity it is recommended that the impact of land-use and land-use change should be integrated in future LCAs assessing the environmental performance of perennial crop-based value chains. In addition, it is recommended to assess and report the impact of land-use on biodiversity and soil quality separately and not aggregate the results in order to enable an in-depth analysis.

4.4. Carbon sequestration – A twofold mitigation strategy through storage of CO_2 in the soil and in the product

As explained above SOC can be used as an indicator to assess the impact of land-use on the soil quality. However, the increase of SOC is also associated with another positive effect on the environment. Through the increase of carbon in the soil, CO_2 is removed from the atmosphere and thus the climate change slowed down as the soil act as a temporary sink.

It was shown that the transition from arable to perennial crops such as miscanthus increased the soil organic carbon (SOC) content by 0.7 to 2.2 Mg C ha^{-1} yr^{-1} which corresponds to 2.6 to 8.1 Mg CO_2 ha^{-1} yr^{-1} (McCalmont *et al.* 2017). However this increase in SOC was only shown for the transition from annual to perennial crops. The effect of replacing grassland though perennial grasses is associated with a high degree of uncertainty and might lead to a decrease in SOC and thus to additional CO_2 emissions (Harris *et al.* 2015). Other studies found no significant changes in the SOC content while replacing C3 grassland with miscanthus (Zatta *et al.* 2014). In future LCAs, an increase in SOC should be accounted for when assessing the environmental performance of the cultivation of perennial crops instead of annual ones. When cultivating perennial crops, in particular perennial grasses, on former grassland there is not such a straightforward answer. Further research is necessary to obtain better estimates of the transition-caused SOC content changes.

However, the increase in SOC when replacing annual plants with perennials has also to be seen critically. The replacement of annual food crops can lead to indirect land-use change (iLUC) effects, which can have severe impacts on the environment (Overmars *et al.* 2011). Indirect land-use change occur when biomass production for industrial purposes displaces agricultural production and thus causes additional land-use change because then this agricultural crops have to be grown on other land (Melillo *et al.* 2009). If land on which food crops were cultivated is now used to produce biomass for industrial purposes the impact of the iLUC has to be included in the assessment of the environmental performance. However it is a different matter when annual bioenergy crops are substituted by perennials. An example for such a case is the utilization of miscanthus as a biogas substrate which is able to substitute silage maize (Kiesel *et al.* 2017; Kiesel & Lewandowski 2017). In this case, there is no iLUC when the energy yield of the perennial alternative is comparable or even higher.

In addition to the carbon sequestration in the soil, bio-based value chains could help to mitigate climate change trough the sequestration of carbon in the product. An example for such a bio-based product is insulation material produced from miscanthus biomass. The GWP of miscanthus-based insulation material was assessed by Wagner *et al.* (2017). This study included the cultivation of miscanthus, the production of the insulation material, the End-of-Life phase of the product and the substitution of a reference system. However, the carbon sequestration in the product was excluded in this assessment. Pawelzik *et al.* (2013) identified and evaluated seven approaches to include the biogenic carbon storage in products in LCA. For the application of a cradle-to-grave system they recommended the approach described within the International Reference Life Cycle Data System (ILCD) handbook (European Commission 2010). In this approach it is differentiated between biogenic carbon stored in the product for more or less than 100 years. The chosen time period is based on the 100 year timeframe, which is used by the IPCC (IPCC 2007). In case, where carbon is stored for more than 100 years, it is assumed to be stored permanently. If it is stored less than 100 years, which is probably the case for most bio-based products, then a weighting factor of 1% per year is applied. This means that the amount of biogenic carbon contained in the product is multiplied by the years it is assumed to be stored, divided by one hundred (European Commission 2010).

In the following section, this calculation is exemplarily shown for miscanthus-based insulation material. It is assumed that the insulation material has a use phase of 25 years, after which it is incinerated and the carbon is released back into the atmosphere. The substitution of a fossil reference system (in this case glass wool mats) by 1 m^3 miscanthus-based insulation material leads to a climate change mitigation potential of 295 kg CO_2 eq. (Wagner *et al.* 2017). In order to produce 1 m^3 of insulation material 194.3 kg dry miscanthus biomass is necessary which has a carbon content of 43.9% (Monti *et al.* 2008; Wagner *et al.* 2017). This is equal to approximately 313 kg CO_2 per 1 m^3 of insulation material. Using the ILCD approach this results in an additional CO_2 mitigation potential of 78 kg CO_2 eq. (European Commission 2010), which corresponds to a further increase of the climate change mitigation potential by around 26%. This equals 6.2 t CO_2 eq. ha^{-1} yr^{-1}.

Based on the magnitude of the influence on the carbon mitigation potential of bio-based value chains it is recommended, in accordance with Pawelzik *et al.* (2013), to integrate the carbon sequestration in the product in further studies analysing the environmental performance of the material use of biomass. Thereby, it is crucial to clearly indicate which approach was applied and on which underlying assumptions the calculation of this sequestration potential is based.

The ILCD approach could also be used to quantify the impact of CO_2 which is only temporarily stored in the soil. That could occur for example when miscanthus is integrated in a crop rotation and thus only cultivated for a 20 years period. The SOC which is sequestrated during the cultivation period would probably be re-emitted after the recultivation. Using the SOC values estimated by McCalmont *et al.* (2017) this corresponds to an additional climate change mitigation potential of 0.2-0.8 t CO_2 eq. ha^{-1} yr^{-1}.

4.5. Assessing the environmental performance of perennial crop cultivation on marginal land

In the last decade, there were growing concerns that the use of biomass for industrial purposes, in particular for energy generation, would lead to a rise in food insecurity (Tenenbaum 2008). In order to resolve this problem, the biomass cultivated for industrial purposes should neither compete with food nor feed production (Tilman *et al.* 2009). When additional biomass is needed, as in a developing bioeconomy, these biomass ideally should either stem from by-products of food and feed crop cultivation (Kim & Dale 2004; Chitawo & Chimphango 2017) or should be grown on sites where no food or feed crops are cultivated. One example of such sites is marginal land, which is no longer agriculturally used (Tilman *et al.* 2009). Cai *et al.* (2011) assessed the availability of marginal land, which in this context was defined as marginal mixed crop and vegetation land, marginal cropland and marginal grassland. In Europe, this resulted in 111 million ha, which are potentially available to grow perennial biomass crops. However, the amount of available marginal land predicted varies widely. This is also due to the uncertainty about the definition of "*marginal land*" which is frequently used as an umbrella term encompassing reclaimed, degraded or abandoned land (Dauber *et al.* 2012).

In the discussion on using marginal land for biomass production it has to be mentioned that perennials, such as miscanthus, which are grown on marginal sites, have lower yield in comparison to those grown on good agricultural land (Lewandowski *et al.* 2016). Though in this thesis it was shown, that the environmental performance of perennial crop-based value chains is still relatively good, in comparison with a mostly fossil-based reference system, even when grown on marginal land (Wagner *et al.* 2017). However, an intensive agricultural use of these marginal land should also be viewed critically. Marginal land is often associated with a higher degree of biodiversity, such as a higher species richness, in comparison to productive agricultural land (Verhulst *et al.* 2004; Dauber & Miyake 2016). If marginal sites are cultivated intensively to produce biomass, this has a negative effect on the biodiversity (Foley *et al.* 2005;

Dauber *et al.* 2015) due to the loss of habitats as well as changes in species abundance and richness (Immerzeel *et al.* 2014). In order to decrease the pressure of land-use on the biodiversity there are two approaches: *land sparing* and *land sharing* (wildlife-friendly farming). The *land sharing* approach propagates the extensive use of land in order to increase the amount of animal, wild plant and other species on existing farmland. The *land sparing* approach on the contrary proposes to increase the yield on agricultural sites in order to decrease the overall need for land and land-use change and thus protecting intact biodiversity habitats (Green *et al.* 2005). As it was shown by Meyer *et al.* (2017) the yield has a strong positive impact on the environmental performance of perennial crop-based value chains. Therefore, the use of the *land sharing* approach, with its low yields through extensive land use, seems in this context as not the most appropriate option. It is proposed to apply the *land sparing* approach and therefore divide the marginal land into species-rich marginal land (SRML) and marginal land which is more species-poor (SPML). This should be analysed *ex-ante* in a biodiversity assessment. While the SPML could be cultivated quite intensively, SRML should only be cultivated, when other environmental advantages of perennial crops prevail, such as the erosion protection through the long soil cover.

A further example for land, where no competition with food crops exists, is contaminated land such as brownfields and land contaminated by sludge and landfills due to municipal activities (Pidlisnyuk *et al.* 2014). The contamination is often caused by so called trace elements such as cadmium (Cd), copper (Cu), chromium (Cr) and lead (Pb). Several studies have shown that miscanthus can be cultivated on these sites. It improves the soil quality and has the ability to stabilize the trace elements in the soil. As most inorganic contaminants are sequestered into the root system, the above-ground miscanthus biomass can be safely used for industrial purposes despite the contamination (Li *et al.* 2014; Nsanganwimana *et al.* 2014; Pidlisnyuk *et al.* 2014). A recent study suggested that in the EU-18 the area of trace element-contaminated land is over 4 Mio ha (Evangelou *et al.* 2012), which emphasizes the potential of using such sites. Even though the uptake of trace elements, such as heavy metals, by miscanthus is rather slow (Pidlisnyuk *et al.* 2014), it nevertheless improves the environmental performance of the miscanthus biomass and thus the whole value chain as it reduces the terrestrial ecotoxicity due to the uptake of heavy metals from the soil. Therefore, it is crucial when assessing the biomass cultivation on these sites to include those impact categories which are strongly influenced through heavy metals, such as terrestrial ecotoxicity and human toxicity (Wagner & Lewandowski 2017).

4.6. Conclusion

Based on the outcomes of the current study it can be concluded that a holistic Life-Cycle Assessment for analysing the environmental performance of perennial crop-based value chains should:

- Include at least the impact categories marine ecotoxicity, human toxicity, agricultural land occupation, freshwater eutrophication and freshwater ecotoxicity besides the GWP.

  It was clearly demonstrated that evaluating the environmental performance solely based on the GWP can lead to incorrect conclusions drawn from the studies.

- Assess the impacts of land-use on soil quality and in particular on the biodiversity.

  It is recommended to assess the impacts of land-use on the biodiversity by adopting an approach which uses the species richness as indicator such as the one described in Chaudhary *et al*. (2015). In case of the impact of land-use on soil quality it is advised to follow the approach detailed by Milà i Canals *et al*. (2006), which uses the SOM content in the soil as an indicator.

- Apply site and crop-specific data for biomass yield and fertilizer-induced emissions.

  Yield and fertilizer-induced emissions are strongly dependent on site and management-specific parameters such as soil, climate and type of fertilizer used. The environmental performance of various bio-based value chains thus differ highly due to site-specific conditions.

- Only compare the performance of different crops when the data regarding the cultivation is obtained from field trials under *ceteris paribus* conditions.

  As explained above, the key parameters for the cultivation process are highly site-specific. This indicates, that a comparison is not possible, if the data used for the assessment is not obtained from field trials under equal conditions.

Many studies which analyse the environmental performance of perennial crop-based value chains include only the GWP and use generic data especially in the assessment of the biomass cultivation. However as demonstrated above, these results are often associated with a high degree of uncertainty. Based on the outcome of this study it is strongly doubted if it is possible to transfer and use the outcomes of such assessments in practice, for example as a decision support tool. It is highly recommended to use, whenever available, site and crop-specific data. A much closer cooperation between LCA users and agronomists could help to solve this issue and lead to a considerable improvement in the accuracy and reliability of assessments analysing the environmental performance of perennial crop-based value chains.

The case studies included in this thesis assessed only the environmental performance of perennial crop-based value chains. The key findings of this thesis though, such as the necessity to include more impact categories or the considerable influence of fertilizer-induced emissions, are also applicable for annual crop-based value chains. However, there is a major difference between annual and perennial crops. As a low input and resource efficient crop, miscanthus requires relatively small amounts of fertilizer and pesticides. In addition, soil cultivation is only necessary in the first year due to its perennial nature. This means that even with lower yields, miscanthus has a comparatively good environmental performance. Annual biomass crops on the contrary often require high amounts of fertilizers and pesticides, and as a result their environmental performance is more sensitive to a decrease in yield. Consequently from an environmental point of view they are less or even unsuitable for the cultivation of marginal sites, where often only lower yields are achievable. Therefore a utilization of perennial biomass crops is preferable on these sites. This is further reinforced by the ecosystem services provided by perennial plants, such as erosion protection or improvement of the soil quality.

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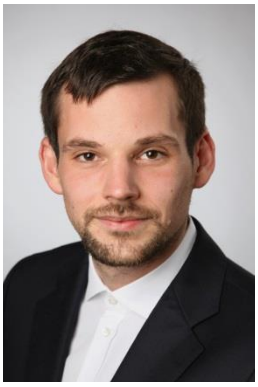

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Professional experience

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University of Hohenheim, Institute of Crop Science, Department of Biobased Products and Energy Crops
Doctoral Student

- PhD. Thesis: "Methodological approaches for assessing the environmental performance of perennial crop-based value chains"

08/2013 – 08/2014

University of Hohenheim, Institute of Crop Science, Department of Biobased Products and Energy Crops
Research Assistant

- Support of the planning and coordination of the new master course "bioeconomy"
- Supervision of the exercises to the lecture "Nachhaltigkeit und Bewertung von NawaRo-Pflanzen - Life Cycle Assessment"
- Upbringing of Miscanthus plants from seeds in the greenhouse and the subsequent planting in the field

11/2011 – 02/2012

PE International, Leinfelden – Echterdingen
Trainee

- Data research and processing
- Creation and editing of LCA reports and presentations
- Application of the LCA – Software GaBi
- Building-up of a master database

06/2006 – 07/2006	**Forestry Office Gaggenau** **Trainee** ▪ Practical work placement

Education

10/2010 – 03/2014	**University of Hohenheim, Stuttgart** **Master Programme Biobased Products and Bioenergy** ▪ Master's thesis: „Ist Durum aus (intensiver) heimischer Produktion nachhaltiger als importierter Durum aus extensiver Produktion?“ ▪ Master of Science: Grade “Very Good”
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10/2005 – 09/2006	**University of Karlsruhe, Karlsruhe** ▪ Study courses in the field of engineering
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Conference contributions and summer schools

10/2016	**Oral presentation** at the *“3rd German-Chinese Workshop on Biotechnology in a Bioeconomy”*: Wagner, M, Lewandowski, I (2016) Miscanthus as bioeconomy crop for marginal land
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Publications

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