Energy, Sustainability and Society

ORIGINAL ARTICLE

Open Access

Valorization of organic waste fractions: a theoretical study on biomethane production potential and the recovery of N and P in Austria

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Abstract

Background: Due to climate change and the rising world population, sustainable energy and fertilizer production faces many challenges. The utilization of organic waste fractions is one possible solution for promoting sustainability. Organic waste fractions have a high potential for biomethane production, which could positively contribute to the current energy mix. Furthermore, organic waste fractions could be used for nutrient recovery (i.e., the recovery of N and P) concurrently to their use in biomethane production. This study examined the theoretical potential of organic waste fractions for valorization in Austria. Further, it provides a theoretical overview of biomethane production and nutrient-recovery potential.

Results: This analysis revealed a total substrate potential of 13 Mt per year in Austria, with the highest contribution from manure. Over 900 million Nm^3 of biomethane could potentially be produced from organic waste fractions. Furthermore, developing organic waste fractions as an energy source could improve the impact of the natural gas consuming sectors on climate, reducing 2.4 Mt of CO_2 emissions annually. Regarding nutrient recovery, more than 60 kt of N and 20 kt of P could potentially be recovered per year.

Conclusion: The study shows a high potential for producing biomethane from organic waste fractions in Austria. The overall production potential could substitute up to 11% of the Austrian natural gas demand, which could highly decrease the CO_2 emissions from fossil energy carriers. Furthermore, a high nutrient recovery potential was identified for an inclusive implementation of an efficient recovery.

Keywords: Biogas, Anaerobic digestion, P recovery, N recovery

Introduction

In 2011, the European Union published "a roadmap for moving to a competitive low-carbon economy in 2050", which aims to reduce greenhouse gases by 80 to 95% by 2050 compared to their levels in 1990 [1]. This should help to limit the global temperature rise to a maximum of 2 °C.

To reach this goal, the current fossil energy carriers must be replaced by renewable ones. There are several possible approaches to move towards a sustainable economy. One option is the electrification of the industrial and household sector and substitution of fossil energy carriers like natural gas, coal, and other fossil-fuel electricity production pathways with renewables including biomass, photovoltaic, and wind energy [2]. However, some industrial sectors rely on natural gas or carbon as part of their production pathway, such as the chemical industry [3]. Furthermore, renewable methane could serve as a bridge to the full electrification of private households. Therefore, renewable methane sources

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should be developed for these sectors. This could be accomplished through the production of synthetic natural gas via the thermochemical conversion of biomass to gas; alternatively, biogas could be produced via anaerobic digestion [4, 5].

Anaerobic digestion is an especially interesting option, since it could upgrade waste streams that are currently unused as energy carriers or burned in waste power plants [6]. After gas upgrading and cleaning to reach the required quality, the gathered biogas could be fed to the natural gas grid. This could increase the share of biogas within the total natural gas distribution.

Apart from its impact on climate change, valorization of organic waste fractions via anaerobic digestion has the potential to play a key role in the future food and feed supply. Currently, we are moving towards a global population of approximately 10 billion humans by the middle of the twenty-first century [7]. This could place high demands on the global food supply, especially if climate change is also considered [8]. Thus, increased food production is necessary, which will require significant quantities of N- and P-based fertilizers [9]. According to Mogollón et al. [10], P input in croplands will increase by 51 to 86% by 2050, i.e., from 14.5 Mt_P to 22–27 Mt_P annually. Regarding N, Mogollón et al. [11] stated that in a relatively optimistic and sustainable scenario, the annual demand could stabilize at 85 Mt_N in 2050; however, it could rise as high as 260 Mt_N.

Sufficient N_2 gas is accessible in the atmosphere to produce the needed amount of N-fertilizers. However, to produce fertilizers that can be used in agriculture, it is necessary to transform N_2 gas into NH_4^+, NO_2^-, or NO_3^- to extract N [12]. The most common way to manufacturer N-based fertilizers is by producing N via the Haber–Bosch process [13]. Therefore, a shortage of N fertilizer in the future appears unlikely. However, the Haber–Bosch process is highly energy intensive and therefore, to reduce climate change, NH_3 must be produced sustainably [14]. This could be accomplished by recovering N from waste streams, a process similar to the already employed removal which is done to prevent environmental contamination, though with an energy-intensive process. To reduce the environmental footprint of N fertilizers, the direct recovery and reuse of N from wastewater is a promising option.

P production is energy intensive, and P is currently primarily mined from igneous and sedimentary deposits located in Northern Africa, China, Kazakhstan, the Middle East, and Florida, USA [15]. Consequently, countries without sufficient P deposits are highly dependent on those countries with significant P resources. Furthermore, current P fertilizer usage is expected to lead to resource depletion by the end of the twenty-first century or the beginning of the twenty-second, according to Duley [16]. Another major disadvantage, that comes with the use of conventional fertilizers, are impurities in the fertilizers, like uranium, that follow from the mining process [17]. To reduce the demands on the global P resources, P recovery, like N recovery, is essential. Furthermore, several studies have shown that nutrient uptake of N and P, crop yield and soil quality is the same for conventional synthetic (fossil) and bio-based (recovered) fertilizers [18].

Thus, several current studies have focused on P and N recovery from different biogenic waste sources, especially from wastewater, sewage sludge, and manure [19–21]. Egle et al. [19] stated that P recovery from sewage sludge ash could consistently reach recovery rates of 60 to 90%. Further, this process entailed relatively low production costs, depending on the purity requirement of the products. For direct sewage sludge recovery, a P recovery rate of 40 to 50% was observed, and high rates of heavy metals were also removed. However, costs for this process were high compared to those of conventional fertilizer production. Vanotti et al. [20] reported even higher nutrient recovery rates from swine wastewater starting from 70%, going up to 100%, if P recovery via $MgCl_2$ was combined with NH_3 recovery through gas-permeable membranes and low-rate aeration. Van der Hoek et al. [21] states that a change of the wastewater treatment system would be necessary to use the full potential of N recovery from wastewater due to limited possibilities for improvement of sustainability in current systems. For instance, a separate urine collection has shown a significant improvement in the recovery rates of N. However, it was further observed that the recovery of nutrients should not negatively affect the biogas production process from wastewater and that the parallel usage of different recovery methods is highly recommended.

Some studies have examined anaerobic treatments for other biogenic wastes. Campos et al. [14] published a review on the treatment of agro-food wastes. The review listed several studies that focused on different feedstocks, such as biowaste from households and bush-, grass- and tree-cuts, and these studies found that the anaerobic pretreatment of such waste fostered high rates of nutrient recovery with values higher than 80%.

However, most recent studies have focused either on nutrient (P and/or N) recovery or on biomethane production potential. Furthermore, several studies focused only on one feedstock and did not address a broad range of biogenic waste streams. Therefore, this study examined both the biomethane production and nutrient recovery potential of major biogenic waste streams in Austria. Additionally, we investigated the possibility of replacing nonrenewable or energy-intensive products with the recovered pendant. Thus, this study aimed to identify the theoretical upper limit of the realizable provision of renewable biomethane and fertilizers using Austria as a national case study.

Methods

The potential calculations were carried out using reference values for nutrient recovery and biomethane production from literature and applying them to the reported annual waste generated in Austria. The relevant methodologies applied, definitions and values for the present study are described below.

Description of the analyzed system

This study focuses on the theoretical valorization potential of biogenic waste streams in Austria according to their methane, P and N recovery potential. In order to perform this analysis, data concerning anaerobic digestion potential of residual and waste materials was extracted from existing literature [22–26].

The selected streams were manure, straw, waste from food production, biowaste from households, bush-, grass- and tree-cuts, and sewage sludge (see Table 1). Since municipal garden and park wastes, cemetery wastes, roadside vegetation, and kitchen and foods wastes exist in comparatively smaller quantities in Austria [26], they were classified as "other" biogenic wastes. Manure included cattle, poultry, pig, and horse manure without straw in the analysis. Furthermore, only manure produced at agricultural facilities with more than 50 livestock animals was considered in this study. This was due to the difficulty of collecting manure from sites with 50 animals or less. For straw, only cereal, maize, rape straw, and sugar beet leaves were considered, since other straws do not exist in sufficient quantities.

As shown in Fig. 1, statistical data on the selected waste substrates were collected on a regional basis in Austria, based on the dry matter (DM). Further, biomethane production and nutrient recovery factors were used to evaluate the potentials for the specific fractions. The analyzed reference year was 2017. However, if no current material for this reference year was available, older data were used. For example, 2017 data were not available for food-production waste, and hence data from prior to 2017 were used for the analysis.

Methane production

Biogas is produced during the anaerobic digestion process of the considered substrates. Depending on the feedstock, the methane content in the biogas from the analyzed waste fractions varies between 50% [27] and 63% [28]. Depending on the category, either data on the biomethane content or on the biogas-production potential (including the potential biogas composition) were

Table 1 Methane and biogas production rates

Substrate	Methane production rate	Biogas production rate	Methane share	Sources
Manure				
Cattle dung		60 Nm^3/t_{DM}	60 vol%	[27]
Pig dung		60 Nm^3/t_{DM}	60 vol%	[27]
Pig manure		20 Nm^3/t_{DM}	60 vol%	[27]
Poultry manure		80 Nm^3/t_{DM}	60 vol%	[27]
Horse manure w/o straw		60 Nm^3/t_{DM}	60 vol%	[27]
Straw				
Cereal straw		331 Nm^3/t_{DM}	51 vol%	[27]
Maize straw		331 Nm^3/t_{DM}	51 vol%	[27]
Rape straw		187 Nm^3/t_{DM}	52 vol%	[27]
Beet leaves	105 Nm^3/t_{DM}			[29]
Waste from food production	145 Nm^3/t_{DM}			[22, 23]
Other biogenic wastes				
Municipal garden and park waste	105 Nm^3/t_{DM}			[29]
Cemetery wastes	105 Nm^3/t_{DM}			[29]
Roadside vegetation	105 Nm^3/t_{DM}			[29]
Kitchen and food wastes	164 Nm^3/t_{DM}			[30]
Biowaste from households	185 Nm^3/t_{DM}			[31]
Bush-, grass- and tree-cuts	105 Nm^3/t_{DM}			[29]
Sewage sludge	312 Nm^3/t_{DM}			[28]

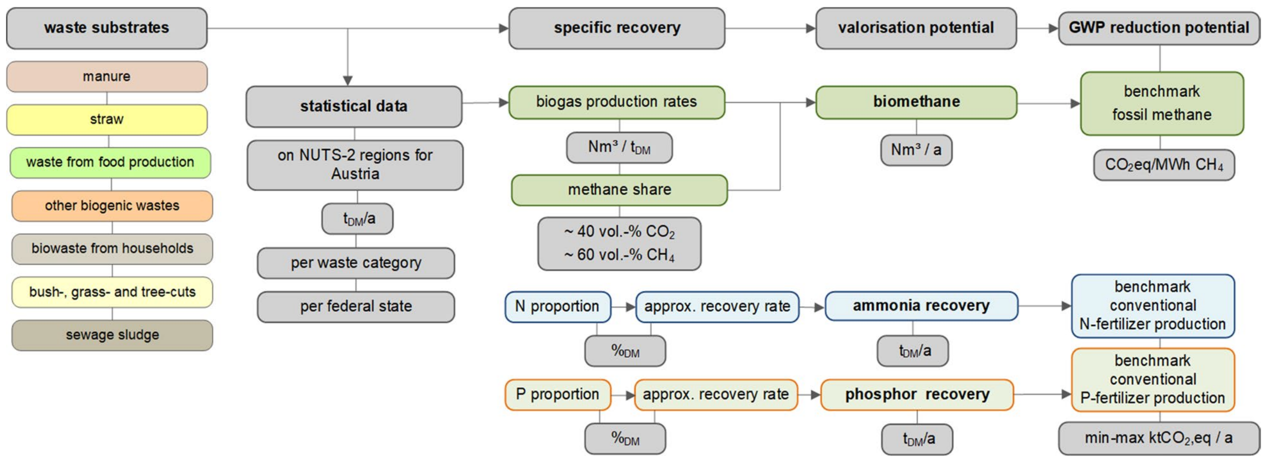

Fig. 1 Overview of the analyzed system and the used methods

available to calculate the methane production potential. If only a range was given, the median was used for calculations. The production rates used for the analysis are shown in Table 1.

P and N content

To calculate the theoretical recovery potential from the different substrate streams, it was first necessary to calculate the P and N content of the substrates. This was obtained by applying mass proportions obtained from the literature to the different waste streams. The values obtained for the analysis are shown in Table 2.

As shown in Table 2, sewage sludge has by far the highest share of P and N based on its dry matter. Apart from sewage sludge, green waste (bush-, grass-, tree-cuts, etc.) provides the second highest N and P proportions. All other categories have proportions $\leq 1\%_{DM}$.

P and N recovery technologies

According to a review by [14], P-precipitation from liquid is a promising method for P recovery from agro-food-waste. P can be recovered in the form of struvite ($MgNH_4PO_4$), calcium phosphate ($Ca_3(PO_4)_2$*nH_2O), K-struvite ($KMgPO_4$*$6H_2O$) or newberite ($MgHPO_4$*$3H_2O$). All four products can be recovered with efficiencies ranging from 80 to 100% according to literature. The recovery method of P-precipitation from liquid could be applied to manure and sewage sludge and yield all four mentioned product types. However, [36] stated that P-precipitation from liquid to produce K-struvite is also a viable method for P recovery (with an efficiency rate of 80 to 90%) from organic biological wastes and could therefore be applied to other substrate flows as well. Similar high efficiencies (72 to 97%) for P-precipitation were also indicated in a review by Cieslik and Konieczka [37].

In case of P recovery by the production of struvite, NH_3 is recovered at the same time. Since the chemical composition of struvite is $MgNH_4PO_4$, the production consumes N as well as P with a N:P ratio of 2.2:1 according to their molar masses.

Table 2 N and P proportion for different substrates

Substrate	N proportion	P proportion	Sources
Manure			
Cattle dung	$0.4\%_{DM}$	$0.1\%_{DM}$	[32, 33]
Pig dung	$0.5\%_{DM}$	$0.3\%_{DM}$	[32, 33]
Pig manure	$0.4\%_{DM}$	$0.2\%_{DM}$	[32, 33]
Poultry manure	$0.5\%_{DM}$	$0.2\%_{DM}$	[32, 33]
Horse manure w/o straw	$0.5\%_{DM}$	$0.1\%_{DM}$	[32, 33]
Straw			
Cereal straw	$0.5\%_{DM}$	$0.3\%_{DM}$	[34]
Maize straw	$0.9\%_{DM}$	$0.2\%_{DM}$	[34]
Rape straw	$1.1\%_{DM}$	$0.6\%_{DM}$	[34]
Beet foliage	$0.3\%_{DM}$	$0.1\%_{DM}$	[34]
Waste from food production	$0.7\%_{DM}$	$0.1\%_{DM}$	[30]
Other biogenic wastes			
Municipal garden and park waste	$2.5\%_{DM}$	$1.7\%_{DM}$	[29]
Cemetery wastes	$2.5\%_{DM}$	$1.7\%_{DM}$	[29]
Roadside vegetation	$2.5\%_{DM}$	$1.7\%_{DM}$	[29]
Kitchen and food wastes	$0.9\%_{DM}$	$0.1\%_{DM}$	[30]
Biowaste from households	$0.9\%_{DM}$	$0.1\%_{DM}$	[30]
Bush-, grass- and tree-cuts	$2.5\%_{DM}$	$1.7\%_{DM}$	[29]
Sewage sludge	$7.5\%_{DM}$	$5.8\%_{DM}$	[21] [35]

Another method for P recovery could be membrane filtration. According to Bolzonella et al. [38], the observed P-recovery efficiency rate from anaerobic digested agro-food wastes was between 43 and 75%. However, unlike P-precipitation from liquid, membrane filtration can be used for N recovery as well. The study found a 47 to 65% recovery rate for N. A similar result was observed by De Vrieze et al. [39], with an average P recovery efficiency of 55% and an average N recovery efficiency of 70%. Unlike previously mentioned membrane filtration studies, Gerardo et al. [40] found P recovery efficiencies ranging from 80 to 100%.

Other methods mentioned by Campos et al. [14] are P recovery from sludge ash, with an average efficiency of 97%, and ion-exchange precipitation, with an average efficiency of 90%.

In a review by Gherghel et al. [41], four industrial processes were evaluated (AirPrex, PHOSPAQ, Seaborne, and AshDec). All these processes have shown recovery potentials for P of about 90%, applicable especially to sewage sludge. However, according to Kataki et al. [42] PHOSPAQ could also be applied to municipal, farm, and industrial wastes with P recovery efficiencies of up to 95%.

A median was calculated for each substrate stream according to the information obtained from the literature reviewed (see Table 3).

A detailed overview about the different recovery rate ranges and the calculations of the medians, as well as the allocation of the different processes to the waste streams is documented in Table 1 of the supplementary material. The selection of considered recovery processes and the allocation to the related waste category, is based on up to date literature, especially reviews, to the topic of fertilizer recovery.

From a cost perspective a study by Egle et al. [19] has shown a potential cost decrease of −4% in comparison to conventional P removal technologies (reference value of 11 € per population equivalent per year) for most applied technologies, if P is recovered from the liquid phase of a waste water treatment plant. If recovery is done in the sewage sludge phase the cost change varies between −9% (MEPHIREC®) and 40% (Stuttgart (MAP)) and if P recovery is applied to the sewage sludge phase cost increase varies between 10% (fertilizer industry) and 40% (RecoPhos® (mineral fert.)). However, the values for especially the values for recovery from sewage sludge or sewage sludge ash have shown high deviations in terms of cost change in comparison to the reference value of P removal. To sum up, especially a liquid phase recovery can have a cost decreasing effect if applied to waste water treatment.

Table 3 Approximated P and N recovery rates for different substrates

Substrate	P recovery rate	N recovery rate
Manure	90%	70%
Sewage sludge	90%	70%
Straw	*83%*	77%
Bush-, grass- and tree-cuts	83%	71%
Biowaste from households	*71%*	70%
Waste from food production	71%	70%
Other biogenic wastes	71%	70%

Reduction in global warming potential

According to Fehrenbach et al. [43], conventional N-fertilizer production leads to emissions of 6.5 $kg_{CO2,eq}/kg_N$ and conventional P-fertilizer production to 0.52 $kg_{CO2,eq}/kg_P$. This results in high global-warming potential (GWP) of about 750 $kt_{CO2,eq}/a$ for Austrian fertilizer production, which corresponds to about 10% of Austrians agricultural GHG emissions [44]. Therefore, it is necessary to investigate the impact of recovery processes on GWP. However, recovery processes can significantly vary in terms of chemical demands, primary energy demands, and global warming potential. Furthermore, the recovery processes employed also impact on these categories for the treatment plants overall, and changes in fertilizer transport volumes should be considered as well. For example, a reduction of N leads to lower oxygen demand and possibly to a lower GWP, as it produces less N_2O emissions. According to several studies [45–48], the GWP of P recovery from waste varies little between the different recovery processes. This could have a positive influence on the recovery system, and in fact, negative emissions were reported for some recovery-process integrations. Furthermore, these emissions are considered "negative" due to the reduction in N_2O emissions obtained through anaerobic digestion and the avoidance of fossil fuel usage.

According to Amann et al. [48], P recovery could vary between 12 and 30 $kg_{CO2,eq}/kg_P$ from the liquid phase, 2 and 22 $kg_{CO2,eq}/kg_P$ from sewage sludge, −9 and 11 $kg_{CO2,eq}/kg_P$ from sewage sludge ash. Bradford-Harke et al. [47] analyzed two processes producing struvite resulting in specific GWPs of −5 $kg_{CO2,eq}/kg_P$ and 20 $kg_{CO2,eq}/kg_P$. Since this study does not focus on one specific recovery process, a GWP range was chosen. This was done according to the values considered in the study by Amann et al. [48]. Values between −9 and 2 $kg_{CO2,eq}/kg_P$ were stated for processes with recovery values similar to the ones used in this study. Both studies include factors like substituted fertilizer and energy as well as reduction of substrate incineration and transport demand as negative GHG emissions. Resource demand and direct emissions are counted as positive GHG emissions. All factors are compared to conventional P removal in waste water treatment systems, since removal is mandatory.

In terms of N recovery, emission values vary between 4.1 and 6.2 $kg_{CO2,eq}/kg_N$ according to Deviatkin et al. [49]. Therefore, GWP savings due to N recovery could range between 0.3 and 2.4 $kg_{CO2,eq}/kg_N$, if compared to conventional N fertilizer production, since N removal is not mandatory for waste water treatment.

Biomethane is considered to be CO_2 neutral, since it is common that biotic carbon emissions are not counted as GHG emissions [50], and therefore, its emissions are calculated as 0 $kg_{CO2,eq}/MWh_{CH4}$, compared to the fossil pendant emissions during the use phase of 239 $kg_{CO2,eq}/MWh_{CH4}$ [51]. This corresponds to a simplified estimate, as biomethane production also causes emissions.

Results and discussion

The results show the theoretical methane production and P and N recovery rates for different waste streams existing in Austria. Furthermore, the possible savings in CO_2,eq achievable through the use of biomethane and biomethane's application potential are discussed. Also a short discussion on challenges regarding potential realization approaches were incorporated in this section.

Substrate potential

As shown in Table 4, the Austrian Federal Institute for Agriculture [25] documents the manure amount with more than 7.5 Mt_{DM}/a. Therefore, this sector alone provides a higher quantity compared to the other sectors combined. In total, the relevant waste streams sum up to an annual quantity of 13 Mt_{DM}. However, this value displays the theoretical potential, without any technical, political or societal restrictions. Currently, a technical potential that includes these constraints would be on average about 23% lower according to Strümer et al. [52]. However, it can be expected that in the near future this value will be reduced by the increase of collection system efficiency and a more favorable political and legislative framework due to the need of a sustainable circular economy.

A detailed overview of the substrate quantities present in different Austrian states can be found in Additional file 1: Table S2.

There is still potential to increase the numbers shown in Table 4. As mentioned in "Methods", the manure waste only includes agriculture with livestock units higher than 50. By including also smaller farms via small-scale manure collection technologies, the full potential could be up to 2.5 times higher [25]. A similar increasing effect could occur by gathering and including waste fractions from other straw fractions that are currently not included due to missing harvest possibilities. The same effect must also be considered for sewage sludge from plants smaller than 2000 person equivalents that are currently not included in the study. Furthermore, the category of biowaste in residual waste that is not gathered separately due to a lack of recycling behavior is currently completely excluded. This could result in a significantly higher biowaste from households' yield. However, currently only rural areas in Austria have high biowaste recycling rates, while urban areas have lower recycling rates [53].

However, the extra collection that could increase substrate potential is always linked to new investment costs for existing plants and increased investment costs for new ones which have long depreciation periods. This could be overcome by changes in legislation or subventions on a national or EU basis. Since such actions could increase substrate potentials significantly, this would have a direct impact on increasing biomethane and nutrient production through anaerobic digestion plants, which could positively impact the economy as well. Furthermore, the maximum potential obtained according to current framework of 13 Mt_{DM} faces further problems beyond investments for existing plants or for new ones. These problems are linked to several limitations in the collection processes.

The limitations will reduce the amount of available biowaste resources considerably compared to the theoretically quantified ones in this analysis. For agricultural residues, farmers may believe that straw and manure and its nutrients are needed in the fertilization cycle to sustain long-term productivity. Potential substrates (e.g., biowaste from households, bush-, grass- and tree-cuts) are distributed over large areas, often far away from the locations of their potential utilization, and some are already integrated in established disposal pathways, such as thermal recycling. The economic viability of the valorization sector, with or without subsidies or environmental taxes on alternative nonrenewable sources,

Table 4 Substrate potential in Austria

Substrate	Quantities in t_{DM}/a	Sources
Manure	7,501,996	[25]
Straw	2,431,191	[25]
Waste from food production	1,235,441	[22–24]
Other biogenic wastes	614,600	[26]
Biowaste from households	525,751	[26]
Bush-, grass- and tree-cuts	476,552	[26]
Sewage sludge	236,200	[26]
Total	13,021,731	

will determine the final limit on reasonably available waste streams for biomethane production and nutrient recovery.

To enforce both the use of biowaste fractions for energy and resource recovery, local cooperation between waste collectors, sewage (biogas) plant and gas grid operators, and others is necessary to use existing infrastructure, logistics, and expertise to promote the recovery and upgrade of material and energetic resources from biowaste at attractive locations in Austria and throughout Europe. Furthermore, in comparison to conventional synthetic (fossil) fertilizers, bio-based/recovered fertilizers often faces legislative challenges that have to be overcome on a political level to implement a sustainable circular economy [54].

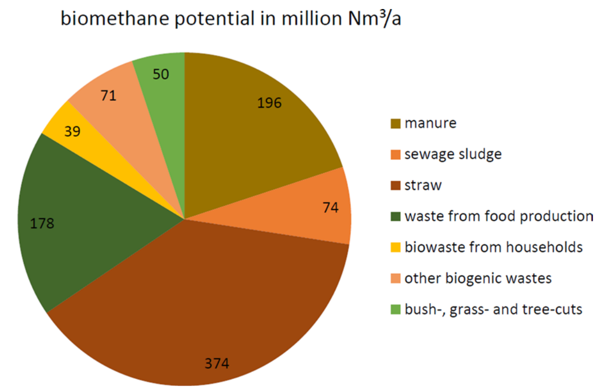

Fig. 2 Biomethane production potential

Biomethane production potential

Currently in Austria there are about 15 biomethane production plants with a production capacity of about 3000 Nm^3/h [55]. At present, biogas is still mainly a fuel for power generators, which use it to generate green electricity and heat. In the future, the aim is to process biomethane and feed it into the natural gas grid as well as to sell biogas at the filling station. The advantages of biomethane or virtual biogas are not only the location-independent use, but also the ability to store it. Renewable methane as produced via the biogas route is an efficiently storable renewable energy source besides storage options like hydropower. This means that its use is time-independent and biomethane can provide valuable balancing energy and close gaps in other renewable energies such as wind and photovoltaics. To increase the above-stated value, the full amount of organic wastes in Austria should be used for biomethane production.

Using the substrate yield shown in Table 4 and the methane production rates shown in Table 1, the biomethane production potential was calculated. As shown in Fig. 2, the overall methane production potential is 981 million Nm^3 CH_4 per year. This corresponds to a heating value of approximately 10,088 GWh per year. According to the BP Statistical Review of World Energy [56], the Austrian natural gas consumption was 8.7 billion Nm^3 in 2018. Therefore, the biomethane generated from waste could substitute approximately 11% of the total Austrian natural gas demand. According to the emission factor mentioned in Sect. 2.5, 239 t_{CO2},eq per GWh_{CH4} could be saved. This results in savings of up to 2.4 Mt_{CO2}, eq per year by substituting 981 million Nm^3 of fossil methane with a renewable sources.

The highest share on the total biomethane production originates from the anaerobic digestion of straw. To be more specific, straw has a production potential of 374 million Nm^3 per year. This results mainly from its comparatively high yield of 2.4 Mt per year and its high methane production rates with ratios between 100 and 165 Nm^3 per t depending on the feedstock (see Table 1). Even though manure has the highest substrate share with 7.5 Mt per year or 58%, it has only the second highest methane potential of 196 million Nm^3 CH_4 per year. This results from the comparatively low biomethane production rates of 12 to 36 Nm^3 per t manure (see Table 1) depending on the origin of the feedstock. With 178 million Nm^3 per year, waste from food production has the third highest share. The other sectors have a combined production potential of only up to 234 million Nm^3 per year due to their small quantities.

N and P recovery and GWP reduction potential

As described in the method section, the N and P content of the different substrate classes was first calculated. This was calculated based on the factor proportions from Table 2. As expected from the values of Table 2, the N content is much higher compared to the P content, in the substrate (see Fig. 3).

As shown in Fig. 3, the total N content is 88 kt per year and the total P content is 28 kt per year. The major portion of the N and P contents come from the classes manure (42% of total N content and 44% of total P content). Even though the specific N and P contents are comparatively low with values below $1\%_{DM}$, the high quantities of the substrate lead to a N content of 37 kt and P content of 12 kt in the annual nutrient from manure potential. The second highest N and P contents come from sewage sludge. This is due to its high N and P contents, consequently sewage sludge provides approximately 20% of the total N and P even though it only contributes 2% of the total substrate mass. The high N and P contents and the high environmental value of recovering it from waste water was already documented in several studies, as for example, the study about the wastewater

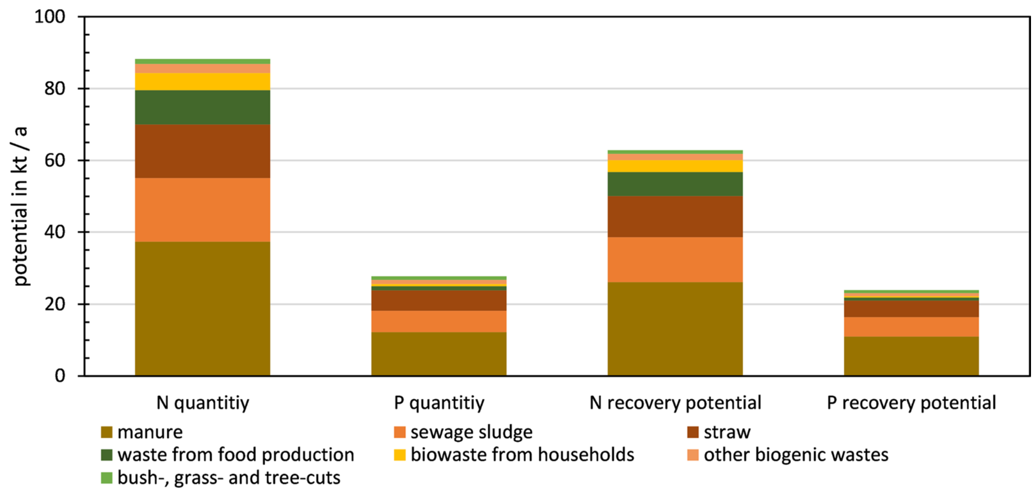

Fig. 3 N and P content and recovery potential in the different substrate fractions

treatment plant in Amsterdam West by van der Hoek [21]. This paper stated that a separate urine collection could significantly increase the recovery efficiency of N in wastewater treatment plants.

In total, a recovery rate of 71% for N and 86% for P was observed as shown in Fig. 3.

For N, a recovery potential of 62.8 kt per year was obtained. On an annual basis, 26.1 kt of this potential results from the N recovery of manure, 12.4 kt from the N recovery of sewage sludge, 11.5 kt from the N recovery of straw and 12.8 kt from the N recovery of the other wastes. For comparison, 62.8 kt recovery potential corresponds to 56% of the agricultural N input for the year 2018 with an N input value of 113 kt [57]. Since N is recovered as NH_3 or NH_4^+ compound, a recovery of N from wastes could significantly reduce the necessary amount of NH_3 production via the energy-intensive Haber–Bosch process. Therefore, it has the potential to significantly reduce the CO_2 footprint of the fertilizer production industries.

In case of the P recovery, the total recovery potential is 23.9 kt per year. The three major sectors for P recovery are again manure, sewage sludge and straw with an annual recovery potential of 10.9 kt, 5.4 kt and 4.7 kt. The other sectors contribute 2.9 kt per year to the P recovery potential. In total, Austrian agriculture had a demand of 28 kt of P as fertilizer in 2018 [57]. Further, a recovery rate of 23.9 kt per year could supply 85% of the 2018 demand. This could significantly reduce the environmental impact of P fertilizer production (e.g., due to mining) and could get Austria closer to a circular economy. Further, the Corporation for Agricultural Market Austria (AMA) states a declining demand for fertilizers. This could result in an overproduction of renewable P and N fertilizers and therefore reduce the possibility of selling them. A theoretical overproduction is also possible due to imports of organic products from other countries.

These findings are in good accordance to the N- and P-recovery potential from biogenic residues in Austria estimated by Pesendorfer et al. [58]. In their study, they considered a smaller range of potentially suited materials for N- and P-recovery, which resulted in a lower nitrogen potential (40.4 kt_N per year compared to 62.8 kt_N per year in this study). Nevertheless, the amounts of recoverable P are similar (22.7 kt_P per year vs. 23.9 kt_P per year), because the major P-sources (i.e., manure and sewage sludge) are considered in both estimations.

The global warming potential for recovery was calculated according to the emission values of Sect. 2.5 combined with the P and N recovery potentials shown in Fig. 3. Since there is no single value for emission factors for nutrient recovery, a range of potential GHG emission has to be displayed. Figure 4 shows the maximum and minimum GWP due to nutrient recovery for N and P recovery, as well as a benchmark according to Fehrenbach et al. [43] for comparison.

As shown in Fig. 4, the emissions due to N-recovery varies between 258 and 390 $kt_{CO2,eq}$ per year. The specific value of 6.5 $kg_{CO2,eq}$ per kg_N for conventional N-fertilizer production stated by Fehrenbach et al. [43] would mean a reduction by 17 to 149 $kt_{CO2,eq}$ per year since the benchmark value is 407 $kt_{CO2,eq}$ per year. This mainly results from a less energy-intensive process for N-recovery compared to conventional N-fertilizer production.

In terms of GWP of P recovery, a different behavior was observed. As mentioned previously, the GHG emissions of the recovery process depends on the process of P recovery. As shown in Fig. 4, the GWP varies

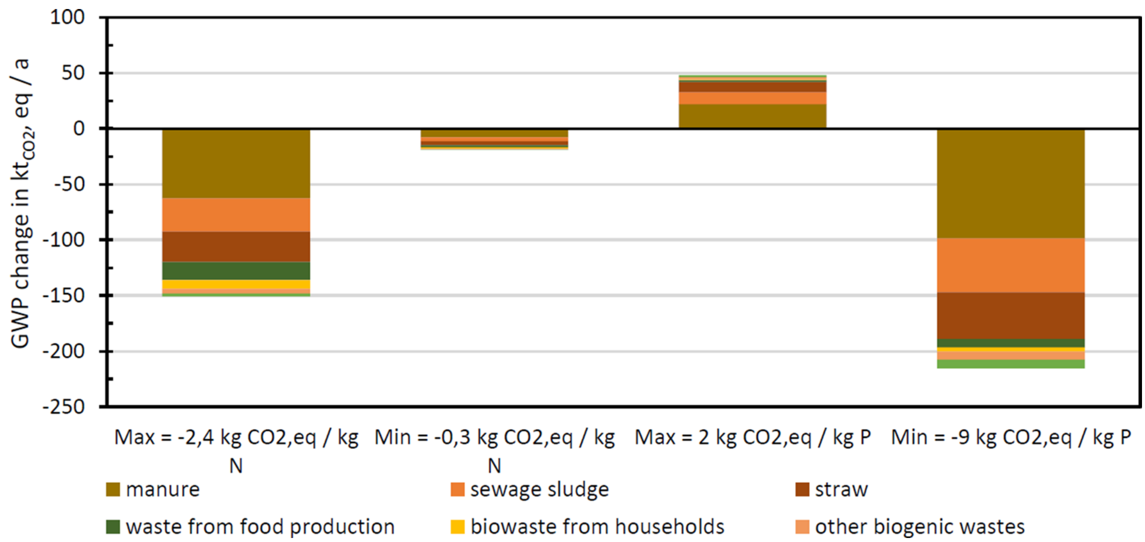

Fig. 4 GWP range of N and P recovery

between 48 and −215 $kt_{CO2,eq}$ per year. In contrast to that of the N recovery, the maximum emission factor for P recovery had shown a higher GWP compared to the benchmark. For the analyzed system, this would lead to an extra 36 $kt_{CO2,eq}$ per year due to the recovery process. However, P recovery also has the potential to positively influence the GWP of waste treatment. By using the minimum value shown in Sect. 2.5, a reduction of up to 227 $kt_{CO2,eq}$ per year by using recovered P-fertilizer instead of a new one would be possible as well.

In total, it has to be stated that the reality will be somewhere in between. However, even though P recovery could lead to an increase of GWP, recovery has to be recommended in view of dwindling P resources. GWP could then be reduced even for processes with positive emission factors by using renewable energy carriers instead of the current conventional ones. This could be applied to N recovery as well. According to Deviatkin et al. [49], the stated emission factors for N recovery have a large share of GWP due to chemical production and electricity generation. Both can be significantly reduced by using renewable electricity instead of the current mixes and chemicals from biorefineries or from recovered chemicals. Therefore, and because of potential improvements in recovery processes, a reduction of GWP for nutrient recovery can be expected in the future. Furthermore, in case of N, a GWP reduction for conventional N-fertilizers cannot be expected due to well-developed technologies.

Conclusions

This work analyzed the theoretical nutrient recovery and biomethane production potential from waste fractions for Austria as reference system. Furthermore, the effect on the GWP was analyzed as part of the study. The analysis showed a total potential for biogenic waste of approximately 13 Mt per year. Out of this, manure contributed the highest quantity with 7.5 Mt per year. Further, straw and waste from food production also contributes with over 1 Mt per year.

The methane potential analysis has shown a production potential of up to 981 million Nm^3, with straw waste, waste from food production and manure contributing the highest share in all categories. This corresponds to more than 10 TWh and therefore to about 11% of Austria's natural gas demand. By using waste as source for biomethane production, a national greenhouse gas emission reduction of ~ 2.4 Mt_{CO2}, eq per year could be achieved.

Despite this significant potential, it is necessary to point out its theoretical character. Realization approaches need to meet significant challenges regarding decentralized feedstock accessibility and non-uniform and low density character as well as volatile quality of carbon sources impacting the efficiency and cost for conversion to biomethane.

N and P recovery has been proved to have a positive effect. It was concluded that up to 63 kt N and 24 kt P could be recovered from biogenic waste on an annual basis. Owing to these high values, it was possible to obtain a substitution potential for Austria. This would correspond to 56% N-fertilizer and 85% P-fertilizer substitution by recovered nutrients. Especially in terms of P-fertilizer, such recovery systems could relieve the demand on P resources, if installed comprehensively.

Furthermore, nutrient recovery could also help to reduce the climate change, if the recovery route is processed in a sustainable way and therefore reduces the carbon footprint of recovered fertilizers in comparison to conventionally produced ones. N recovery has the potential to reduce greenhouse gas emissions due to N-fertilizer production by up to 149 $kt_{CO2,eq}$ per year. In terms of P recovery, the GWP reduction potential is even higher with up to 227 $kt_{CO2,eq}$ per year. However, since recovery methods vary a lot, P recovery has the potential to increase GWP compared to conventional P-fertilizer production of up to 36 kt_{CO2}. Combining biomethane production and nutrient recovery, a maximum GWP reduction of ~ 2.8 Mt_{CO2} was calculated.

Currently, the basic data concerning the environmental impact of nutrient recovery and biomethane production is quite weak on a national or European basis. Therefore, future work has to be conducted for a detailed LCA for the different recovery methods either on the European level or on a national level. This would allow more detailed knowledge on the global warming potential. Furthermore, such a study should include other relevant environmental factors such as primary energy demand, human toxicity, etc., to define optimal recovery strategies. Besides the impact on environment, influence on economy is an important factor as well. Therefore, a techno-economy assessment has to be carried out for an efficient design of the recovery systems.

Supplementary information

Supplementary information accompanies this paper at https://doi.org/10.1186/s13705-020-00272-3.

Additional file 1: Table S1. Recovery methods, products and efficiencies according to literature - (1) manure, (2) straw, (3) waste from food production, (4) other biogenic wastes, (5) biowaste from households, (6) bush-, grass- and tree cuts, (7) sewage sludge. **Table S2.** Substrate potential for the different Austrian states (Bundesanstalt für Agrarwirtschaft, 2016; Bundesministerium Nachhaltigkeit und Tourismus, 2019; FNR, 2013; Reisinger, 2012; Universität Rostock, 2007).

Abbrevations
DM: Dry matter; GHG: Green house gases; GWP: Global warming potential.

Units
GWh: Gigawatthours; $kg_{CO2,eq}$: Kilograms CO_2 equivalent; kt: Kilotonnes; Nm^3: Normal cubic meters; Mt: Megatonnes; MWh: Megawatthours.

Chemicals
$Ca_3(PO_4)_2*nH_2O$: Calcium phosphate; $KMgPO_4*6H_2O$: Potassium struvite; $MgCl_2$: Magnesium chloride; $MgHPO_4*3H_2O$: Newberite; $MgNH_4PO_4$: Struvite; N: Nitrogen; NH_4^+: Ammonium; NO_2^-: Nitrite; NO_3^-: Nitrate; P: Phosphorus.

Acknowledgements
This work was supported by the Austrian Research Promotion Agency [Grant Number 864876-ReNOx 2.0] and by the Association Energy Institute at Johannes Kepler University Linz.

Authors' contributions
DCR is the principal author of this paper. JL is coworker in the technical process assessment working group and has contributed with ideas and discussion to the paper, as well as editing. ME is project leader of the ReNOx 2.0 project that has led to the presented results and contributed with comments and discussion to the paper. All authors read and approved the final manuscript.

Funding
This work was funded by the Austrian Research Promotion Agency under the Grant Number 864876-ReNOx 2.0.

Availability of supporting data
Not applicable.

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Received: 27 January 2020 Accepted: 5 November 2020
Published online: 23 November 2020

References
1. European Commission (2011) A Roadmap for moving to a competitive low carbon economy in 2050. European Commission, Brussels
2. Schiffer ZJ, Manthiram K (2017) Electrification and decarbonization of the chemical industry (in English). Joule 1(1):10–14
3. Lechtenböhmer S, Nilsson LJ, Åhman M, Schneider C (2016) Decarbonising the energy intensive basic materials industry through electrification–implications for future EU electricity demand. Energy 115:1623–1631
4. Billig E, Thraen D (2017) Renewable methane–a technology evaluation by multi-criteria decision making from a European perspective. Energy 139:468–484
5. Steinlechner C, Junge H (2018) Renewable methane generation from carbon dioxide and sunlight. Angew Chem Int Ed Engl 57(1):44–45
6. Mao CL, Xi JC, Feng YZ, Wang XJ, Ren GX (2019) Biogas production and synergistic correlations of systematic parameters during batch anaerobic digestion of corn straw (in English). Renew Energy 132:1271–1279
7. Lutz W, Kc S (2011) Global human capital: integrating education and population. Science 333(6042):587–592
8. Tscharntke T et al (2012) Global food security, biodiversity conservation and the future of agricultural intensification (in English). Biol Cons 151(1):53–59
9. Zhou K, Barjenbruch M, Kabbe C, Inial G, Remy C (2017) Phosphorus recovery from municipal and fertilizer wastewater: China's potential and perspective. J Environ Sci (China) 52:151–159
10. Mogollón J, Beusen A, Van Grinsven H, Westhoek H, Bouwman A (2018) Future agricultural phosphorus demand according to the shared socioeconomic pathways. Glob Environ Change 50:149–163
11. Mogollón J, Lassaletta L, Beusen A, Van Grinsven H, Westhoek H, Bouwman A (2018) Assessing future reactive nitrogen inputs into global croplands based on the shared socioeconomic pathways. Environ Res Lett 13(4):044008

12. Savci S (2012) An agricultural pollutant: chemical fertilizer. Int J Environ Sci Dev 3(1):73
13. Galloway JN et al (2008) Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. Science 320(5878):889–892
14. Campos JL et al (2018) Nitrogen and phosphorus recovery from anaerobically pretreated agro-food wastes: a review. Front Sustain Food Syst 2:91
15. Cichy B, Kuzdzal E, Krzton H (2019) Phosphorus recovery from acidic wastewater by hydroxyapatite precipitation. J Environ Manage 232:421–427
16. Duley B (2001) Recycling phosphorus by recovery from sewage. Rhodia Consumer Specialties UK Ltd.(for Centre Europeen d'Etudes des Polyphosphates), Oldbury
17. Schnug E, Lottermoser BG (2013) Fertilizer-derived uranium and its threat to human health. ACS Publications, New York
18. Vaneeckhaute C, Meers E, Michels E, Ghekiere G, Accoe F, Tack F (2013) Closing the nutrient cycle by using bio-digestion waste derivatives as synthetic fertilizer substitutes: a field experiment. Biomass Bioenerg 55:175–189
19. Egle L, Rechberger H, Krampe J, Zessner M (2016) Phosphorus recovery from municipal wastewater: An integrated comparative technological, environmental and economic assessment of P recovery technologies. Sci Total Environ 571:522–542
20. Vanotti MB, Dube PJ, Szogi AA, Garcia-Gonzalez MC (2017) Recovery of ammonia and phosphate minerals from swine wastewater using gas-permeable membranes (in English). Water Res 112:137–146
21. van der Hoek J, Duijff R, Reinstra O (2018) Nitrogen recovery from wastewater: possibilities, competition with other resources, and adaptation pathways. Sustainability-Basel 10(12):4605
22. Reisinger H (2012) Rückstände aus der Nahrungs-und Genussmittelproduktion. Materialien zur Abfallwirtschaft, Umweltbundesamt, Wien
23. Braunschweig FAL, Leipzig IE. Biogaserzeugung durch Trockenvergärung von organischen Rückständen, Nebenprodukten und Abfällen aus der Landwirtschaft; 2007.
24. Fachagentur Nachwachsende Rohstoffe eV (2013) Leitfaden Biogas: Von der Gewinnung zur Nutzung. Gülzow, Germany
25. Bundesanstalt für Agrarwirtschaft, "Grüner Bericht 2016," 2016, [Online]. https://www.awi.bmlfuw.gv.at/index.php?id=gruenerbericht, Accessed 03 Sept, 2019.
26. Bundesministerium Nachhaltigkeit und Tourismus, "Die Bestandsaufnahme der Abfallwirtschaft in Österreich - Statusbericht 2019," Vienna, 2019. https://www.bmnt.gv.at/dam/jcr:909b907d-e688-4572-afcd-6f6727573014/BAWP_Statusbericht_2019.pdf, Accessed 03 Sept 2019.
27. LFL. *Biogasausbeute verschiedener Substrate*. 2019. https://www.lfl.bayern.de/iba/energie/049711/?sel_list=20%2Cb&anker0=substratanker#substratanker. Accessed 04 Sept 2019.
28. Kuo J, Dow J (2017) Biogas production from anaerobic digestion of food waste and relevant air quality implications. J Air Waste Manage Assoc 67(9):1000–1011
29. FNR. Leitfaden Biogas: Von der Gewinnung zur Nutzung. Fachagentur Nachwachsende Rohstoffe e.V. (FNR); 2013.
30. Alibardi L, Cossu R (2015) Composition variability of the organic fraction of municipal solid waste and effects on hydrogen and methane production potentials. Waste Manage 36:147–155
31. KTBL. Betriebsplanung Landwirtschaft 2008/09: Daten zu Schlachtabfällen und Schweinemist aus Genesys-Merkblatt M101 - Biogasausbeute von Hofdüngern und Co-Substraten. Kuratorium für Technik u. Bauwesen in der Landwirtschaft (KTBL), 2008.
32. Pfundtner E (2007) Der sachgerechte Einsatz von Biogasgülle und Gärrückständen im Acker-und Grünland. Bundesmin. für Land-und Forstwirtschaft, Umwelt und Wasserwirtschaft
33. Baumgarten A et al (2017) Richtlinie für die sachgerechte Düngung im Ackerbau und Grünland, *Anleitung zur Interpretation von Bodenuntersuchungsergebnissen in der Landwirtschaft*. BMLFUW, Wien
34. LK OÖ. *LK-Strohrechner: Kostenloses Excel-Programm der Landwirtschaftskammern*. 2019. Accessed 05 Sept 2019.
35. Lampert C, Reisinger H, Zethner G (2014) Bioabfallstrategie. Umweltbundesamt, Wien
36. Vaneeckhaute C et al (2017) Nutrient recovery from digestate: systematic technology review and product classification (in English). Waste Biomass Valoriz 8(1):21–40
37. Cieślik B, Konieczka P (2017) A review of phosphorus recovery methods at various steps of wastewater treatment and sewage sludge management. The concept of "no solid waste generation" and analytical methods. J Clean Prod 142:1728–1740
38. Bolzonella D, Fatone F, Gottardo M, Frison N (2018) Nutrients recovery from anaerobic digestate of agro-waste: Techno-economic assessment of full scale applications. J Environ Manage 216:111–119
39. De Vrieze J et al (2019) Resource recovery from pig manure via an integrated approach: a technical and economic assessment for full-scale applications. Bioresour Technol 272:582–593
40. Gerardo ML, Aljohani NH, Oatley-Radcliffe DL, Lovitt RW (2015) Moving towards sustainable resources: Recovery and fractionation of nutrients from dairy manure digestate using membranes (in English). Water Res 80:80–89
41. Gherghel A, Teodosiu C, De Gisi S (2019) A review on wastewater sludge valorisation and its challenges in the context of circular economy (in English). J Clean Product 228:244–263
42. Kataki S, West H, Clarke M, Baruah DC (2016) Phosphorus recovery as struvite from farm, municipal and industrial waste: Feedstock suitability, methods and pre-treatments (in English). Waste Manage 49:437–454
43. Fehrenbach H et al (2008) Criteria for a sustainable use of bioenergy on a global scale. Federal Environment Agency Germany, prepared by the Institute for energy and Environmental research (IFEU), Heidelberg
44. Eurostat. *Greenhouse gas emissions by source sector (source: EEA)*.2020 https://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_air_gge&lang=en. Accessed 20 July 2020.
45. Prapaspongsa T, Poulsen TG, Hansen JA, Christensen P (2010) Energy production, nutrient recovery and greenhouse gas emission potentials from integrated pig manure management systems. Waste Manage Res 28(5):411–422
46. Nakakubo T, Tokai A, Ohno K (2012) Comparative assessment of technological systems for recycling sludge and food waste aimed at greenhouse gas emissions reduction and phosphorus recovery (in English). J Clean Product 32:157–172
47. Bradford-Hartke Z, Lane J, Lant P, Leslie G (2015) Environmental benefits and burdens of phosphorus recovery from municipal wastewater. Environ Sci Technol 49(14):8611–8622
48. Amann A, Zoboli O, Krampe J, Rechberger H, Zessner M, Egle L (2018) Environmental impacts of phosphorus recovery from municipal wastewater (in English). Resour Conserv Recy 130:127–139
49. Deviatkin I et al (2019) Technical implications and global warming potential of recovering nitrogen released during continuous thermal drying of sewage sludge. Waste Manage 90:132–140
50. Simon K-MR (2015) A critical assessment of microbiological biogas to biomethane upgrading systems. In: Gübitz G, Bauer A, Bochmann G, Gronauer A, Weiss S (eds) Biogas science and technology. New York, Springer, pp 117–135
51. Juhrich K (2016) CO_2 emission factors for fossil fuels. Clim Change 28:9–25
52. Stürmer B (2020) Greening the gas grid—evaluation of the biomethane injection potential from agricultural residues in Austria. Processes 8(5):630
53. Pladerer C, Hietler P (2019) Abfallvermeidung in der österreichischen Lebensmittelproduktion. Österreichische Wasser- und Abfallwirtschaft, journal article 71(5–6):238–245
54. Chojnacka K, Moustakas K, Witek-Krowiak A (2020) Bio-based fertilizers: a practical approach towards circular economy. Biores Technol 295:122223
55. Austrian Compost and Biogas Association. *Kompost Biogas | Anlagenstandorte*. 2020. https://www.kompost-biogas.info/biogas/anlagenstandorte/. Accessed 22 July 2020.
56. British Petroleum p.l.c. BP Statistical Review of World Energy; 2019.
57. AMA. *Düngemitteleinsatz rückläufig zum Vorjahr*. 2019. https://www.ama.at/Marktinformationen/Getreide-und-Olsaaten/Aktuelle-Informationen/2019/Duengemitteleinsatz-ruecklaeufig-zum-Vorjahr. Accessed 06 Sept 2019.
58. Pesendorfer S, Ellersdorfer M, Stocker K. Nutrient recovery and utilization potential from wastewater in Austria. In 15th Minisymposium Verfahrenstechnik and 6th Partikelforum, Leoben; 2019.

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