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Site-specific carbon footprints of Scandinavian wastewater treatment plants, using the life cycle assessment approach

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Graphical abstract

CARBON FOOTPRINT (LCA)

WASTEWATER TREATMENT PLANT

Highlights

- A systematic carbon footprint assessment of Scandinavian wastewater treatment plants.
- The carbon footprint was 0.15-0.66 kg CO₂ eq Mg⁻¹ plant material input.
- Direct greenhouse gas emissions accounted for 44-71% of the carbon footprint burden.
- Electricity consumption accounted for 2-28% of the carbon footprint burden.
- Direct greenhouse gas emissions were the most sensitive model parameters.

Abstract

The carbon footprints of seven wastewater treatment plants using different technologies in Denmark and Sweden were evaluated. The life cycle assessment approach was applied by using site-specific data including measured plant-integrated methane and nitrous oxide emissions. Four different functional units were adopted: 1 Mg of input material entering the wastewater treatment plant in 2015, and the removal of 1 kg of carbon, total nitrogen and phosphorus. The net carbon footprint values found in this study were between 0.15 and 0.66 kg CO₂ eq (Mg of input material)⁻¹ depending on the treatment facility. Direct greenhouse gas emissions were the main contributors to the carbon footprint, accounting for between 44 and 71% of the total burden. The remaining share of the total burden (66 and 29%) was mainly made by energy consumption, chemicals used, and emissions from effluent and land application of biosolids. Direct greenhouse gas emissions were very sensitive model parameters driving result uncertainties. When default values from emission reporting guidelines were applied instead of measured greenhouse gas emission rates, the net carbon footprint was up to four times smaller or seven times larger. The consumption of electricity from the energy grid for plant operation had a great impact on the carbon footprint, due to differences between the energy systems. The share of electricity consumption to the total carbon footprint burden for the Swedish plants was only 2%, whereas it was between 16 and 28% for the Danish plants. This difference was due to the smaller carbon footprint potential of the electricity mix in Sweden than in Denmark. Normalisation of the carbon footprint to the plant pollution load, and the calculation of model uncertainty, allowed for a reliable comparison of plants operating within the same energy system. Finally, suggestions were provided for performing a sound carbon footprint evaluation of wastewater treatment plants.

Keywords: methane emissions, nitrous oxide emissions, emission factors, environmental assessment, EASETECH, greenhouse gas reporting

1. Introduction

Wastewater treatment plants (WWTPs) are a source of anthropogenic greenhouse gas (GHG) emissions to the atmosphere that contribute to climate change (IPCC, 2014). These GHGs are emitted either directly as fugitive methane (CH_4) and nitrous oxide (N_2O) or indirectly as carbon dioxide (CO_2) of fossil origin, due to the energy and materials used in treatment processes (Yoshida et al., 2014a). In general, fugitive emissions of CO_2 from WWTPs are not included in direct GHG emission reporting, because they are considered of biogenic origin and therefore belong to the short carbon cycle (IPCC, 2006). The overall contribution of a WWTP to climate change is obtained by evaluating the facility's carbon footprint. This evaluation is useful at two different stages of implementation of mitigation actions: (1) in a preliminary stage, when a GHG mitigation strategy must be chosen, because it points out the factors that cause the highest environmental impacts, (2) and in a final stage, because it verifies the success of the mitigation actions carried out. Moreover, carbon footprint evaluations of WWTPs allow stakeholders to understand how much these facilities contribute to the global emission of anthropogenic GHGs. At the global level, the International Water Association has established a task group to minimise the carbon footprint of wastewater utilities (<http://www.iwataskgroupghg.com>). At the European level, the European Environment Agency promotes the carbon footprint assessment of water utilities using a life cycle perspective (EEA, 2014). At national level, the Danish and Swedish water and wastewater associations invite WWTP operators to annually report the carbon footprint of their WWTPs (DANVA, 2012; SVU, 2014).

To the authors' best knowledge, only two studies previously accounted for the annual contribution of a WWTP to climate change (Gustavsson and Tumlin, 2013; Yoshida et al., 2014a), meaning that they evaluated the carbon footprint for accounting purposes. Yoshida et al. (2014a) used life cycle assessment (LCA) as a consistent accounting method, when evaluating the carbon footprint of only one Danish WWTP. Yoshida et al. (2014a) used plant-specific influent volume (m^3) of wastewater as functional unit, considered the Danish energy mix in the inventory year, used plant-specific data, and performed an uncertainty propagation analysis to investigate the reliability of the results. Conversely, Gustavsson and Tumlin (2013) investigated 16 WWTPs, located in Denmark, Sweden, Norway and Finland, using an inconsistent

assessment method, inspired by the LCA approach, and applied a number of assumptions resulting in important shortcomings of the study. They used the Swedish population equivalent load as functional unit, disregarding the true pollution load to the different WWTPs. Gustavsson and Tumlin (2013) also assumed the four Scandinavian countries to operate in the same energy system, thus neglecting the large differences among them, used mainly literature data and did not perform an uncertainty propagation analysis.

LCA is a holistic standardised method (ISO, 2006a; 2006b), in which resource use and environmental emissions from all involved processes are included, quantified and converted into environmental impacts. Most often, an LCA focuses on many impact categories (e.g. acidification, ecotoxicity, etc.), but it can also analyse one impact category only. When the evaluation focuses on climate change only, the assessment is called “carbon footprint” (JRC, 2011). Depending on the modelling framework used, LCA can be utilized for two primary aims. The attributional modelling framework is utilized for accounting purposes, meaning that the environmental impacts are accounted within the inventory time period, e.g. one year (JRC, 2011). Differently, the consequential modelling framework is utilized for supporting decisions about alternative options fulfilling the same service, e.g. the treatment of a defined quantity of wastewater (JRC, 2011). Comparing the two modelling frameworks is not possible, though, because they involve different inventory data collections, different calculation of impact assessment results and different LCA result interpretations (JRC, 2011). A literature review performed by Zang et al. (2015) showed that investigations of wastewater treatments were mainly performed using the consequential modelling framework.

Although direct GHG emissions from WWTPs have been recognized as being important, they are most often either estimated with emission factors referring to technologies used at older WWTPs or obtained by using models (Gustavsson and Tumlin, 2013; Parravicini et al., 2016; Yoshida et al., 2014a). Only for a few process units, Gustavsson and Tumlin (2013) used plant-specific direct GHG emissions rates measured at a limited number of the investigated WWTPs instead most of the direct GHG emissions were estimated using literature emission factors. For evaluating the

carbon footprint of two theoretical WWTPs, Parravicini et al. (2016) estimated the direct CH₄ emissions using literature emission factors, and estimated the direct N₂O emissions using a model based on measurements performed at activated sludge tanks. In addition to using plant-specific emission measurements, Yoshida et al. (2014a) estimated direct GHG emissions using literature emission factors and models to investigate the effect of different data collection schemes when assessing the environmental impact of a WWTP. The use of default emission factors for estimating direct GHG emissions could result in incorrect carbon footprint evaluations. Delre et al. (2017) quantified direct GHG emissions from WWTPs using a tracer gas dispersion method and showed that fugitive emissions are plant-specific and a result of the combinations of different wastewater and sewage sludge treatment technologies applied at individual plants. Additionally, the use of site-specific emission factors, measured with on-site measurement methods targeting individual emission sources and leakages, could lead to an underestimation of actual emissions, due to the potential risk of missing out important on-site GHG emission sources (Reinelt et al., 2017). A study investigating the actual contribution of direct GHG emissions to the carbon footprint of WWTPs employing different treatment technologies using site-specific and plant-integrated emission measurements is thus still lacking. Other LCA studies report the relationship between high energy consumption and a high carbon footprint value, but these studies did not include direct GHG emissions (Niero et al., 2014; Rodriguez-Garcia et al, 2011). However, LCAs or carbon footprint evaluations of WWTPs can produce misleading results if plant-specific data are lacking or if some factors and processes are excluded from the assessment (Corominas et al., 2013; Lorenzo-Toja et al. 2016; Yoshida et al., 2014a). Carbon footprint evaluations of WWTPs reported in the literature are often inconsistent and non-transparent, meaning that all relevant factors and processes are not included in the assessment, they are not in line with the European guidelines (JRC, 2011), and all choices related to methodology and data sources are not clarified. This observation is in line with a review study performed by Sabeen et al. (2018) that underlined the need to define more stringent guidelines when evaluating the impacts of the domestic wastewater treatment in all impact categories, including global warming. Therefore, the literature could benefit from some clear guidelines on how to include all relevant processes and factors involved in wastewater treatment, by counting all stages

of the material treatment, from its arrival at the WWTP, to its final release into the environment.

This study investigates the contribution of direct CH₄ and N₂O emissions to the annual carbon footprint of seven Scandinavian WWTPs employing different technologies for wastewater and sewage sludge treatment. The LCA approach for accounting purposes was used as well as site-specific data including measured plant-integrated GHG emission rates. Additionally, this study investigates the contribution of energy when WWTPs operate in different energy systems, i.e. Denmark and Sweden. Finally, suggestions for a sound carbon footprint assessment of WWTPs are provided.

2. Method

Four Danish and three Swedish WWTPs, representative of the regions in which they operate, were studied for the inventory year 2015. All WWTPs performed advanced nutrient removal and differed from each other in terms of treatment capacity (reported as population equivalent), processed material besides wastewater, treatment technologies and type of energy sold to the grid (Table 1). An important difference between the WWTPs was the fate of generated biosolids, namely either on-site incineration or application on agricultural land. The study followed the iterative four steps of the LCA defined in ISO standards 14040 and 14044 (ISO, 2006a; 2006b), i.e. a goal and scope definition, a life cycle inventory, a life cycle impact assessment and an interpretation of the results. Additionally, detailed guidelines issued by the European Commission were followed (JRC, 2010; 2011).

2.1. Life cycle assessment steps

The goal of this study was to evaluate the carbon footprint of seven specific WWTPs in 2015. The study was carried out considering four functional units, which allowed the analysis from different perspectives. The first adopted functional unit was 1 Mg of input material with a plant-specific composition. Yearly inlet composition for each WWTP is available in Tables S1-S7 in the Supplementary Data (SD), while specific input models are described in the following section. This functional unit reflects the actual processed material composition, because this assessment aims to study the importance of direct GHG emissions when accounting the actual impacts produced by

each facility in 2015. Since the main function of a WWTP is the removal of pollution from the wastewater, a useful functional unit could be the removal of organic biodegradable load, expressed as biochemical oxygen demand (BOD) or chemical oxygen demand (COD). However, such functional units could not be used in this study because measurements of BOD and COD were only available for wastewater and not for the additional processed material in the WWTPs, i.e. external sewage sludge, landfill leachate and food waste. To investigate the pollution removal capabilities of the WWTPs, the functional units were expressed in terms of 1 kg of carbon removed, 1 kg of total nitrogen removed and 1 kg of phosphorus removed. In this way the pollution removal performances are split according to the pollutant of interest. The removed mass of the specific pollutant was calculated subtracting the mass of the pollutant in the effluent from the mass of the pollutant contained in the inputs (wastewater plus other materials) (Section S2.4. in the SD).

The system boundaries of the study included all stages of the material treatment, from its arrival at the WWTP, to its final release into the environment (Fig. 1). The construction, maintenance and demolition of the WWTPs were excluded from the assessment, due to its focus on the operational phase in 2015. The attributional modelling framework was adopted, and multifunctionality was addressed using system expansion, including substitution, because the study focused on monitoring and reporting the environmental impacts of single facilities (JRC, 2011). The time horizon for the impact assessment was set at 100 years, while the timing of emissions was not modelled, because all emissions were considered to occur instantaneously.

In the life cycle inventory stage, primary and secondary data were collected. Primary data included the composition of the input and output materials, direct GHG emissions, consumption of fuels and chemicals, consumption of energy from the grid and energy sold to the grid. Furthermore, these primary data were plant-specific and based on WWTP environmental reports, plant operator communications and plant-integrated measurements (Delre et al., 2017; Samuelsson et al., 2018; Yoshida et al., 2014b). Site-specific direct GHG emissions were based on plant-integrated measurements. Secondary data, i.e. the production of fuels, chemicals and energy, were chosen in line with geographical, temporal and technological relevance from the

Ecoinvent database v3.3, sourced from the EASETECH (Environmental Assessment System for Environmental TEChnologies) database v2.3.6 (Clavreul et al; 2014) and the European Life Cycle Database (ELCD). For more information about the primary and secondary data herein, refer to Section S1 in the SD. The life cycle impact assessment focused on the climate change impact category, including CH₄, N₂O and fossil CO₂ emissions, according to the latest IPCC physical science basis report (IPCC, 2013). In the life cycle impact assessment, any changes in atmospheric carbon storage as a result of climate change were not included.

The results obtained in the environmental impact assessment were investigated by performing contribution analysis, sensitivity analysis (i.e. perturbation and scenario analysis) and uncertainty analysis (i.e. uncertainty propagation and uncertainty contribution analysis) (Clavreul et al., 2012). The contribution analysis allowed for the identification of the main contributors to the carbon footprint, while the perturbation analysis pointed out the most sensitive parameters in the model, i.e. those for which a small change would result in a large variation in the carbon footprint result. For each WWTP, sensitivity ratios (SRs) were calculated for all independent parameters, by considering a parameter increment of 10%. SR was defined as follows:

$$SR = \frac{\text{result variation}}{\text{parameter variation}} \cdot \frac{\text{initial parameter}}{\text{initial result}} \quad (\text{eq. 1})$$

The higher SR is the more sensitive is the model to a change of the specific parameter considered. This means that if SR is high, a 10% change in the parameter results in a large change in the carbon footprint. For example, if a parameter has an SR of 3, an increase of 10% of its value results in a carbon footprint that is 30% higher. A scenario analysis, which was performed to assess the effects of adopting different approaches for estimating direct GHG emissions, involved site-specific measurements, Danish national guidelines (DNGs), and Intergovernmental Panel on Climate Change (IPCC) guidelines being compared (IPCC, 2006; Thomsen, 2016). Additionally, a scenario analysis for the two WWTPs Avedøre and Våxjö was performed to investigate the effect of incidental CH₄ emissions caused by digester malfunction. For these two WWTPs only, measurements were performed when there were digester problems, in addition to normal operational conditions (Delre et al., 2017; Yoshida et al., 2014b). The uncertainty of the carbon footprint was calculated by uncertainty propagation of

model parameters. For each WWTP, parameters with SRs higher than 0.02 were implemented with their probability distributions in 100,000 Monte Carlo simulations. Normal, triangular and uniform distributions were adopted for each parameter according to the data source (Section S6 in SD). The distributions of parameter values accounted for measurement inaccuracies, as known as inherent uncertainty, and possible imbalance between the use of data and their representativeness, as known as unrepresentativeness uncertainty (Henriksson et al., 2014). In the Monte Carlo simulations, all parameters were considered independent, meaning that any parameter correlation was not accounted. In general, data correlations can be identified in two categories: correlations within a process record, and correlations between process chains of product systems (Bojaca and Schrevens, 2010; Henriksson et al., 2014). The former considers the correlation among parameter inputs, for example the correlation among different material fractions making the total material processed by each plant (Nhu et al., 2016). The latter considers correlations between two different systems having common unit processes, for example the shared background system of WWTPs using the same energy network and chemicals (Nhu et al., 2016). To the authors' best knowledge, the literature reports only two studies that identically randomised the Monte Carlo simulations for all the compared options, including the correlations between process chains of product systems in 1,000 Monte Carlo simulations (Longo et al., 2017; Nhu et al., 2016). Due to lack of software capable of supporting the huge computational effort with correlated parameters, the uncertainty propagation in this study did not identically randomise the Monte Carlo simulations for all WWTPs. However, this limitation was overcome by running 99,000 Monte Carlo simulations more than those run in the studies of Longo et al. (2017) and Nhu et al. (2016). Finally, an uncertainty contribution analysis was performed by using the law of the total variance reported by Bisinella et al. (2016) to provide an approximate contribution of the direct GHG emissions to the total carbon footprint uncertainty. The law of the total variance was applied to each WWTP models. In addition to the first run of 100,000 Monte Carlo simulations, a second run of 100,000 Monte Carlo simulations was performed considering only the uncertainty of the parameters describing direct CH₄ and N₂O emissions.

2.2. Life cycle assessment modelling

The studied WWTPs were modelled using EASETECH v2.3.6 – a mass flow-based life cycle assessment tool, which enables relationships between flow concentrations and emissions. Additionally, the software allows the user to parameterise systems and perform sensitivity and uncertainty analyses. Each WWTP was modelled according to plant-specific characteristics in terms of input material, emissions into the atmosphere and outputs. The key environmental data describing the reference flow of the study were water, carbon, nitrogen and phosphorus. Carbon was considered of biogenic origin (IPCC, 2006), and all WWTPs were modelled as “one process” with specific energy inputs and outputs. Detailed input composition is reported in Tables S15-S16 in SD. The consumption of electricity and district heating taken from the energy grid were considered as an input and therefore as a carbon footprint burden, due to the emission of fossil CO₂ following the production of the energy. Conversely, produced electricity, heat and methane, all of which were sold and used outside the WWTPs, were considered as outputs and therefore as carbon footprint savings, due to avoided production of energy from fossil or renewable sources. Any energy produced by the WWTP that was used on-site was not a WWTP output, and therefore it was not included in the carbon footprint evaluation. Fuel consumptions were considered as energy inputs into the WWTP. Besides energy balances, mass balances were also performed. All carbon footprint impacts related to the generation of waste treated at the WWTPs were excluded from the assessment (Fig. 1) as a general rule for the LCA modelling of waste management systems (Christensen and Birgisdottir, 2011). Those output materials which avoided the production of new materials generated carbon footprint savings (Fig. 1). The following output materials resulted in the avoided production of comparable products: sand and ash from biosolids incineration avoided the quarrying of construction filling material, and the use on land of biosolids avoided the production of mineral fertilisers (Fig.1).

All WWTPs had specific input material compositions (Table S15 and S16 in SD) and four different material outputs. The first material output was effluent sent to the natural water basin, where eventually parts of the leftover carbon and nitrogen were converted to CH₄ and N₂O (Foley et al., 2010). The second output was sand collected in the sand trap, which was sent outside the WWTP either for recycling as a construction

filling material or for landfilling. The third output represented screenings, i.e. coarse materials separated mechanically from the wastewater and sent to an external facility for incineration. The incineration of screenings was modelled using a built-in EASETECH process representing a generic Danish waste-to-energy plant (Møller et al., 2013). The downstream processes involved in incinerating screenings were cut off from the system boundaries. The last output of the WWTP model represented either produced biosolids or ash from on-site biosolids incineration, which was recycled as a construction filling material or landfilled. Fresh biosolids were stored off-site for 26 weeks. During storage, carbon losses into the atmosphere as CH_4 , and nitrogen losses as N_2O , were modelled according to Samuelsson et al. (2018). For Ryaverket WWTP, the storage time was set to 23 weeks, because biosolids were stored on-site for three weeks prior to off-site transportation (Table 1). Due to a lack of data, any other loss of carbon, nitrogen, phosphorus or water during biosolids storage was assumed negligible. In all WWTP models including the land application of biosolids, emission factors for land application were taken from Bruun et al. (2016) (Section S2.3 in SD) and calculated via agro-system modelling, based on sludge-amended soil incubations. The long-term consequences of the land application of sewage sludge, up to 100 years, were considered. The advantage of using these emission factors is that long-term dynamics are included for carbon and nitrogen, something rarely used in an LCA (Yoshida et al., 2018). To replace mineral fertiliser, substitution ratios in accordance with the Danish ministry guidelines were applied: nitrogen and phosphorous were substituted with a rate of 0.45 and 1, respectively (Ministry of Environment and Food of Denmark, 2015). This means that 1 kg of nitrogen in the biosolids replaces 450 g of nitrogen mineral fertiliser, while 1 kg of phosphorus in the biosolids replaces 1 kg of phosphorus mineral fertiliser. Carbon was not substituted in the model because both in Denmark and Sweden mineral fertilisers do not include carbon as soil improver. Assuming that both countries have the same regulation about biosolids land application is reasonable because the substitution rates hereby used are based on research investigating the performance of nitrogen contained in biosolids compared to nitrogen mineral fertiliser. Carbon footprint burdens from land application were caused by emissions of CH_4 and N_2O as well as the consumption of fuel used in spreading biosolids on land. Conversely, environmental savings were the result of avoiding the production of mineral nitrogen and phosphorus

fertilisers and from carbon sequestration in the soil. In EASETECH, the use on land of biosolids was modelled with two sub-processes, namely the land application sub-process, accounting for GHG emissions and carbon sequestration in the soil, and the sub-process regarding mineral fertiliser substitution. Finally, material transportation was assumed to take place with a truck with a gross weight of 10 Mg with a built-in EASETECH external process (Table S11 in SD). Clarification about the mass balance of the key environmental data and the EASETECH model description for all WWTPs is available in Section S2 in SD.

3. Results and discussion

This section reports and discusses the results following the LCA steps described in section 2.1. Contribution, sensitivity and uncertainty analysis are presented in sections 3.1, 3.2 and 3.3, respectively, while suggestions for a sound carbon footprint assessment are presented in section 3.4.

3.1. Contributing factors and processes

Fig. 2a-d shows the carbon footprint of the seven WWTPs divided into contributing processes and factors using all four functional units (exact data are available in Tables S17-S20 in SD). The carbon footprint (kg CO₂ eq) was expressed per Mg of plant input material (a) as well as per kg carbon (b), total nitrogen (c) and phosphorus (d) removed during treatment. The use of different functional units did not change the results in terms of contributing factors and processes within the same WWTP (Fig. 2e-f). The net carbon footprint shown in Fig. 2a-d was calculated as the sum of the carbon footprint burden (positive value) and carbon footprint savings (negative value). For all seven WWTPs, net carbon footprint values were positive, thus none of the plants was found to be carbon-neutral, meaning that in 2015 the investigated WWTPs overall produced environmental burdens during their operations. Net carbon footprints were between 0.15 kg CO₂ eq (Mg of input material)⁻¹ for Ryaverket and 0.66 kg CO₂ eq (Mg of input material)⁻¹ for Holbæk, between 0.9 kg CO₂ eq (kg C removed)⁻¹ for Lynetten and 2.2 kg CO₂ eq (kg C removed)⁻¹ for Lundtofte, between 6.5 kg CO₂ eq (kg TN removed)⁻¹ for Lynetten and 12.6 kg CO₂ eq (kg TN removed)⁻¹ for Holbæk,

and between 31.3 kg CO₂ eq (kg P removed)⁻¹ for Våxjö and 94.8 kg CO₂ eq (kg P removed)⁻¹ for Holbæk. Carbon footprint burdens were between 2.1 and 23.2 times larger than carbon footprint savings (Ryaverket and Lundtofte, respectively). Direct GHG emissions based on site-specific measurements were the largest contributors to burdens with site-specific values between 44% for Källby and 71% for Avedøre (Fig. 2e). This result implies that the carbon footprint of a WWTP could be reduced significantly if GHG mitigation actions focused on reducing direct GHG emissions. Reduction of direct N₂O emissions would be important, because, with exception of Källby, the carbon footprint of the direct N₂O emissions was usually higher than for direct CH₄ emissions. Contributions to the burden of direct N₂O emissions were between 14% for Källby and 48% for Avedøre, whereas the contribution to the burden of direct CH₄ emissions was between 12% for Lundtofte and 30% for Källby and Våxjö. A comparison with previous LCA studies of WWTPs is limited, due to fundamental differences between studies, such as site-specific technologies and processes, methodological LCA modelling frameworks (consequential vs. attributional), system boundaries and cut-offs, data quality and approaches used for N₂O and CH₄ emission quantification. This limitation is also underlined by Sabeen et al. 2018 reviewing studies from 1990 to 2016. Although the above mentioned limitation, a comparison with previous literature is attempted below.

Gustavsson and Tumlin (2013) found that the contribution of the direct GHG emissions to the carbon footprint of the investigated WWTPs was between about 15 and 60% of the total burdens, and N₂O emissions contributed the most. However, besides the methodological inconsistencies underlined in section 1, Gustavsson and Tumlin (2013) considered for most of the plants emission factors which are not plant-specific, and in the case of N₂O, the study referred to an Australian investigation (Foley et al., 2010), infringing the geographical delimitation of the study (JRC, 2011). Also Lorenzo-Toja et al. (2016) found that direct GHG emissions were important in the evaluation of the carbon footprint of two Spanish WWTPs and that most of the contribution was given by N₂O emissions. However, specific percentages cannot be provided because Lorenzo-Toja et al. (2016) grouped processes and direct emissions in a different way than the current study. Direct N₂O emissions were found important by Schaubroeck et al. (2015) when studying an Austrian WWTP under different scenarios and using site-

specific on-site measurements. Direct N₂O emissions were found to contribute by 74 to 98% of the carbon footprint in the different scenarios. Such large contribution could be due to the synergy of the investigated low energy-demanding technology and the electricity mix based on renewable energies for 53%. Daelman et al. (2013) found that direct GHG emissions were the largest part of the carbon footprint burdens of a Dutch WWTP with enclosed process units. The site-specific N₂O and CH₄ emissions caused 78 and 14% of the total footprint burdens, respectively; while the remaining 8% was caused by the energy consumption. However, these percentages should be smaller because no use of chemicals was included by Daelman et al. (2013). Both direct CH₄ and N₂O emissions were found relevant by Longo et al. (2017) when investigating a WWTP with different scenarios for the reject water treatment. Direct GHG emissions accounted for 56% of the total carbon footprint when their emissions were estimated using literature emission factors, while the remaining share was mainly made by electricity consumption (35%) and treatment of biosolids through composting (9%). In their life cycle inventory, also Foley et al. (2010) reported that N₂O was the largest contributor to GHG emissions in wastewater treatment including direct GHG emissions, emissions from land application of biosolids and emissions occurring from WWTP effluent. However, Foley et al. (2010) did not use plant-specific GHG emission measurements, because they compared ten ideal WWTPs with different treatment characteristics. Finally, Rodriguez-Garcia et al. (2014) established that direct N₂O emissions were the largest contributor to the carbon footprint burdens of two pilot-scale WWTPs. Most of the literature studies evaluating the carbon footprint of a WWTP report a higher contribution of direct N₂O emissions compared to direct CH₄ emissions probably because the global warming potential of N₂O is almost 10 times higher than CH₄ (IPCC, 2013). However, the contribution to the total carbon footprint burdens is also influenced by the quantity of GHG emitted and the magnitude of other contributing processes and factors.

The use of energy (electricity and heat) from the grid, along with fuel consumption, had very different impacts on the carbon footprint of the WWTPs operating in the two different countries. For the Danish WWTPs, energy consumption accounted for between 16% of the burden in Holbæk and 29% of the burden in Lynetten, whereas for the Swedish WWTPs, energy consumption accounted for only 3-

4% of the burden (Fig. 2e), due to the different energy systems of the two countries. In Denmark, in the inventory year, electricity was produced mainly by wind, coal and natural gas, while district heating was produced mainly by coal, biofuels, natural gas and incinerated waste (Table S21 in SD). In Sweden, electricity was produced mainly by nuclear power and hydropower, while district heating was produced mainly by biofuel and incinerated waste (Table S21 in SD). These different energy mix compositions led to a smaller carbon footprint potential per kWh of energy from the grid consumed in Sweden than in Denmark (Table 2) illustrating the importance of the energy system when comparing the carbon footprint of WWTPs operating in different energy systems. Fig. 3 shows the carbon footprint of the use of energy from the grid (heat and electricity), fuel consumption (gasoline, diesel and heating oil) and energy generated and sold to the grid for the seven WWTPs (electricity, heat and methane gas). Exact data are available in Tables S23-S26 in SD). In Fig. 3 the carbon footprint (kg CO₂ eq) is expressed per Mg of plant input material (a) as well as per kg carbon (b), total nitrogen (c) and phosphorus (d) removed during treatment. The net carbon footprint of energy was calculated as sum of the carbon footprint burden (positive value) and carbon footprint savings (negative value). The net carbon footprint of energy depends on a combination of the energy mix composition in the grid, the type and quantity of consumed energy and the type and quantity of energy that the WWTP generated and sold to the grid. Considering these combinations, WWTPs in Denmark had positive net carbon footprints in terms of energy, while plants in Sweden had negative net carbon footprints in terms of energy (Fig. 3a-d). Additionally, the use of electricity from the grid was always the largest contributor to the carbon footprint burden of energy consumption – in Denmark between 97 and 100%, and in Sweden between 64 and 93% (Tables S23-S26 in SD). For the Swedish plants, electricity consumption from the grid accounted for 2% of the total carbon footprint burden, whereas for the Danish plants, electricity consumption from the grid accounted for between 16% of the total burden in Holbæk and 28% of the total burden in Lynetten (Table S27-S30 in SD). Concerning the type of energy sold to the grid, the largest carbon footprint savings were obtained for WWTPs selling methane, because it was assumed to replace the production of natural gas, an energy source with a larger carbon footprint potential in comparison to generating electricity and heat in both countries

(Table 2). Natural gas had a larger carbon footprint potential compared to electricity and heat available in the Danish and Swedish energy grids, because the latter are currently produced ostensibly from sources with a relatively minimal impact on global warming, for example biofuel, waste, hydropower and nuclear power. From a carbon footprint perspective, in 2015, it was thus preferable for the WWTPs to deliver methane directly to the natural gas grid, rather than convert it to heat and electricity. From a GHG mitigation perspective, the energy system should be analysed and the WWTP should sell to the grid a type of energy that has a larger carbon footprint potential per kWh than the type of energy consumed. However, the carbon footprint potential of a specific type of energy could change over time, due to changes in the energy system. Also the literature underlines the influence of the energy source on the WWTP carbon footprint. Foteinis et al. (2018) reported that the use of electricity generated with renewable energy could reduce the carbon footprint of a Greek facility by up to 87.5%. Baskurt et al. (2017) found that using photovoltaic panels for satisfying 60% of the plant electricity demand could reduce the carbon footprint of a Turkish WWTP by 50%. Büyükkamaci and Karaca (2017) reported that using natural gas instead of electricity from the Turkish grid decreased the GHG emission by 26%. Finally, Polruang et al. (2018) showed that the carbon footprint of seven WWTPs in Thailand would decrease between 34 and 42%, if the current energy mix, based on fossil fuels, is substituted by a combination of renewable energies and nuclear power.

Fig. 4 shows the quantity of energy taken from the grid and fuel consumption in terms of kWh per different functional units for all studied WWTPs. Exact data are available in Tables S31-S34 in SD. As reported by Niero et al. (2015), larger Scandinavian WWTPs tended to consume less energy per functional unit than smaller WWTPs (Fig. 4). This result was confirmed by an energy benchmarking study on English facilities performed by Belloir et al. (2015). Additionally, when studying the environmental efficiency of WWTPs regarding the removal of nutrients, also D’Inverno et al. (2018) and Lorenzo-Toja et al. (2018) found that larger plants were more efficient than smaller plants in Italy and Spain, respectively. Therefore, considering only the carbon footprint of energy consumption, the operation of larger WWTPs should be preferred to the operation of smaller facilities, due to higher efficiency.

Fig. 5 shows the energy balance of all WWTPs, expressing the net energy (consumption – energy sold to the grid) and its composition in terms of kWh per different functional units. Four out of the seven WWTPs had a negative energy balance, meaning that energy sold to the grid was larger than fuel consumption and energy taken from the grid combined (Fig. 5). Växjö delivered the largest amount of methane per Mg of input material to the natural gas grid, most likely due to the treatment of substrate in the thermal hydrolysis process before digestion (Table 1), which did not occur at any of the other facilities. However, the highest methane production per kg of carbon input and per kg of nitrogen input amongst the seven plants was obtained at Ryaverket (Table S31 in SD). At Avedøre and Holbæk, produced methane was converted into electricity, before it was sold to the grid (Table 1). Ryaverket, Lynetten and Källby were the only WWTPs selling heat to the district heating grid, derived from different technologies: Lynetten sold extra heat produced from sewage sludge incineration, while Ryaverket and Källby sold heat produced from a heat pump, using the difference in temperature between wastewater and the outdoor temperature. Energy balance and carbon footprint are two different analyses, which are not necessary connected. The current study showed that WWTPs having a negative energy balance (kWh of energy consumed < kWh of energy sold to the grid) could still produce positive net carbon footprints, thus produce burdens to the climate change (Fig. 2 and Fig. 5).

At one WWTP, biosolids were stored on-site for 26 weeks, while at three WWTPs, biosolids were stored at an external facility before use on agricultural land (Table 1). The carbon footprint of external storage accounted for from 7% of the burden for Holbæk to 12% of the burden for Källby (Fig. 2e), and it consisted of CH₄ and N₂O emissions into the air. The use of chemicals contributed to a burden between 4% for Lynetten, Avedøre and Holbæk and 16% for Källby (Fig. 2e), the latter of which was the only facility studied using calcium nitrate for the prevention and removal of hydrogen sulphide. Calcium nitrate was responsible for 72% of the chemical burden at Källby (Table S22 in SD). Since both biosolids storage and use of chemicals are responsible of a significant share of the carbon footprint burdens, it is important to include them in the assessment (Fig. 1). Emissions occurring from effluent when emitted into water contributed to the total burden between 3% in Holbæk and 11% in Ryaverket

(Fig. 2e). About 80% of these impacts were generated by CH₄ emissions (Table S35 in SD).

Although the use of biosolids on land involved both burdens (emissions of CH₄ and N₂O, and the consumption of fuel for spreading biosolids) and savings (avoided production of fertilisers and carbon sequestering) (Fig. 2), the net carbon footprint of this process was negative for all cases where biosolids were land-applied (Table S36 in SD). Savings from carbon sequestering were included in the land application sub-process (Table S36 in SD). The avoided production of mineral nitrogen fertiliser accounted for approximately 90% of the mineral fertiliser savings, while for phosphorus fertiliser the figure in this regard was approximately 10% (Table S36 in SD). This is in contrast with the results of an LCA study of Danish WWTPs, in which most of the savings, caused by mineral fertiliser substitution, were found to be due to the substitution of mineral phosphorus fertiliser rather than nitrogen fertiliser (Niero et al., 2015). Comparing the results of this study to those found by Niero et al. (2015) was a complicated undertaking, because some important inventory data were not reported in the last-mentioned study. For example, the authors did not report the mineral nitrogen fertiliser substitution rate in a consistent way, nor did they mention the chosen processes for mineral fertiliser substitution or emissions associated with the land application of mineral fertiliser – all factors that would have influenced the results. Although the land application of biosolids occurs outside the WWTP, the analysis showed the importance of this process unit in terms of carbon footprint and demonstrated the importance of including land application in the assessment (Fig. 1).

Processes such as the transportation and incineration of screenings made a negligible contribution to the carbon footprint of each WWTP (Fig. 2 and Tables S17-S20 in SD). This result supported the choice to leave out the downstream processes of the screenings incineration, which would have made even smaller carbon footprint impacts, due to the almost complete loss of carbon and nitrogen during the incineration process.

In general, the carbon footprint evaluation of the seven WWTPs showed that savings were caused mainly by substituting mineral fertilisers and energy sold to the grid. Ash and sand recycling contributed with a negligible carbon footprint (Fig. 2f and

Tables S17-S20 in SD). Details about the contribution analysis are available in Section S3 in SD.

3.2. Sensitive model parameters and scenario analysis

Depending on the specific WWTP, the perturbation analysis involved between 22 model parameters for Källby and 40 for Ryaverket (Tables S37-S38 in SD). Although many parameters were site-specific, the perturbation analysis showed that some parameters were sensitive for nearly all WWTPs. Direct GHG emissions were typical examples in this regard, with SRs between 0.22 and 0.53 for N₂O direct emissions, and SRs between 0.13 and 0.51 for CH₄ direct emissions (Tables S37-S38 in SD). Furthermore, the carbon footprint potentials of the energy mix, energy taken from the grid, energy sold to the grid and CH₄ emissions from biosolids storage were among the most sensitive parameters (Tables S37-S38 in SD). These findings are in line with a study by Yoshida et al. (2014a), which reports high parameter sensitivities such as direct GHG emissions and electricity consumption in an LCA for a Danish WWTP. The literature shows similar results also when evaluating the carbon footprint of a small WWTP in Spain and three large Korean facilities (Garfi et al., 2017; Piao et al., 2016), pointing out the relevance of these parameters disregarding size and location of the WWTP. The value of the most sensitive parameters should be chosen carefully in a life cycle inventory step, because small changes could lead to large changes in the net carbon footprint. For this reason, accurate site-specific values are preferred.

The importance of site-specific information was revealed in the scenario analysis, in which different approaches to calculating direct GHG emissions were compared with using measured direct GHG emissions (Fig. 6). The Danish national guideline (DNG) approach gave a net carbon footprint between 1.7 times lower for Lundtofte to 3.9 times lower for Väjjö compared to the baseline approach using site-specific measurements, which was due to underestimated emissions of both CH₄ and N₂O (Fig. 6). Conversely, the IPCC approach gave a net carbon footprint between 1.4 and 7.2 times higher than the baseline approach (Avedøre and Källby, respectively) (Fig. 6). Although the IPCC approach underestimated N₂O emissions, it largely overestimated CH₄ emissions, resulting in an overall overestimation of the net carbon footprint. The very different net carbon footprints obtained using direct GHG

measurements and direct GHG estimation schemes showed that misleading results would be obtained if direct GHG emissions were estimated with current Danish and IPCC guidelines. The scenario analysis studying CH₄ emissions during digester malfunctioning showed that the net carbon footprint burdens increased by about 320 times at Avedøre and 125 times at Väjö (Section S5 in SD). This finding illustrated the importance of monitoring direct GHG emissions during normal as well as non-routine operational conditions, to obtain representative annual emission rates.

3.3. Comparison of wastewater treatment plants

For plant comparison, the net carbon footprint per Mg of input material of each facility was normalised by the pollution load in terms of carbon, total nitrogen and phosphorous, thus expressed as kg CO₂ eq (kg C influent), kg CO₂ eq (kg TN influent), kg CO₂ eq (kg P influent). The comparison was also facilitated by performing an uncertainty propagation analysis in the EASETECH models, in order to obtain the uncertainty of the net carbon footprint (section S6 in SD). Specific uncertainty parameter values are available in Tables S40-S46 in SD. Fig. 7 shows the net carbon footprint values for the seven WWTPs along with corresponding uncertainty for all adopted functional units, so that results are expressed according to the pollution load and removal. Net carbon footprint uncertainties were between $\pm 8\%$ for Holbæk and $\pm 36\%$ for Lundtofte (Table S47 in SD). These two extreme values were most likely caused by N₂O emission rates, which showed the smallest and the largest variations among the different WWTPs, in Holbæk and Lundtofte, respectively (Table S9 in SD). This explanation is also supported by the fact that for both WWTPs, the most sensitive parameter was direct N₂O emissions (Table S37 in SD). Applying the law of the total variance, direct GHG emissions were the parameters that most contributed to the uncertainty of the carbon footprints. The contribution of the direct GHG emissions was between 70 and 99% in Ryaverket and Lundtofte, respectively (Table S48 in SD). The fact that the direct GHG emissions are responsible of the largest part of the net carbon footprint uncertainty highlights the importance of using plant specific reliable and representative annual emission rates.

Due to the influence of the energy system on the carbon footprint results (section 3.1), only WWTPs operating in the same energy system, i.e. country, could be

compared. Although different functional units provided different rankings among the WWTPs, common features were found. Among the Danish WWTPs, Lynetten reported the smallest and Lundtofte the largest net carbon footprint. Although a previous study on Danish WWTPs reported that plants using biosolids on land had lower net carbon footprints than WWTPs using incineration (Niero et al., 2015), this study found that Holbæk (the only Danish plant without biosolid incineration) reported one of the highest net carbon footprints of all among the Danish WWTPs (Fig. 7). When the results were expressed according to carbon and nitrogen load and removal, the Swedish WWTPs showed comparable net carbon footprints (Fig. 7). However, when expressed through the phosphorus load and removal, Växjö reported a net carbon footprint that was smaller than the value of Källby and Ryaverket (Fig. 7 and Table S47 in SD). Details on the uncertainty analysis are available in Section S6 in SD.

3.4. Suggestions for a sound carbon footprint assessment of wastewater treatment plants

Sections 2.1 and 2.2 describing in detail the life cycle assessment steps and the life cycle assessment modelling adopted, respectively, are valuable guidelines for performing a sound carbon footprint of WWTPs. However, a more general guideline is reported in section S7 in the SD.

4. Conclusions and perspectives

This study determined the carbon footprint of seven Scandinavian wastewater treatment plants (WWTPs) using different technologies for wastewater and sewage sludge treatment. The internationally standardised life cycle assessment approach was applied, using plant-specific data, including measured plant-integrated CH₄ and N₂O emissions. The chosen functional units were 1 Mg of treated input material, and the removal of 1 kg of carbon, total nitrogen and phosphorus in 2015.

- None of the plants could be considered carbon-neutral because net carbon footprint values were all positive: between 0.15 and 0.66 kg CO₂ eq (Mg of input material)⁻¹, between 0.9 and 2.2 kg CO₂ eq (kg C removed)⁻¹, between 6.5 and 12.6 kg CO₂ eq (kg TN removed)⁻¹, and between 31.3 and 94.8 kg CO₂ eq

(kg P removed)⁻¹. This means that in 2015 the investigated WWTPs overall produced environmental burdens during their operations.

- Direct GHG emissions were the largest contributors to the carbon footprint accounting for between 44 and 71% of the total burden. This implies that the carbon footprint of a WWTP could be reduced significantly if GHG mitigation actions focused on reducing direct GHG emissions. Additionally, direct GHG emissions were very sensitive model parameters and largely responsible for uncertainty in the carbon footprint evaluation, accounting for between 70 and 99% of the total uncertainty. This highlights the importance of using plant specific reliable and representative annual emission rates.
- Using default emission factors for estimating direct GHG emissions instead of measured site-specific emission rates led to a net carbon footprint up to four or seven times smaller and higher, respectively.
- A scenario analysis, including direct emissions measured during digester malfunctioning, showed a carbon footprint burdens up to 320 times higher than normal conditions, thereby illustrating the importance of a monitoring strategy covering emission variations in the inventory year.
- In the inventory year, plants gained more carbon footprint savings by delivering methane directly to the natural gas grid, rather than first converting it to heat and electricity and then delivering these two energy sources to the energy grid. From a GHG mitigation perspective, the energy system should be analysed and the WWTP should sell to the grid a type of energy that has a larger carbon footprint potential per kWh than the type of energy consumed. However, the carbon footprint potential of a specific type of energy could change over time, due to changes in the energy system.
- The use of electricity taken from the grid had very different impacts for WWTPs operating in different energy systems: the share for the Swedish WWTPs was 2% of the total carbon footprint burden, whereas for the Danish WWTPs, the share was between 16 and 28%. This difference was due to a smaller carbon footprint potential of the grid electricity mix in Sweden than in Denmark. Therefore, the energy system where the WWTPs operate is important for any carbon footprint evaluation.

- Only WWTPs operating in the same energy system were comparable.
- Biosolids storage and its application onto agricultural land resulted not only in emissions being released into the environment, but also in savings in terms of carbon sequestering and mineral fertiliser substitution. Although this process unit is placed outside the WWTP, its inclusion in the assessment was important for obtaining a sound analysis.
- The carbon footprint of the WWTPs can be used as a plant environmental benchmark, albeit this would require systematic and plant-specific data collection, transparent inventory reporting and consistent assessment modelling, as suggested in this study.

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Appendix A. Supplementary data

Supplementary data (SD) contains detailed descriptions of all life cycle assessment steps.

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FIGURES

Fig. 1. System boundaries to the study. All processes considered in the study are shown; however, not all processes shown were relevant in relation to all seven wastewater treatment plants.

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Fig 2. A contribution analysis of the carbon footprint for the investigated wastewater treatment plants in 2015. The carbon footprint (kg CO₂ eq) is expressed per Mg of plant

871 input material (a) and per kg carbon (b), total nitrogen (c) and phosphorus (d) removed
872 during treatment. The net carbon footprint is the sum of carbon footprint burdens
873 (positive values) and carbon footprint savings (negative values). (e) Contribution of
874 processes resulting in burdens to the total carbon footprint burden (%). (f) Contributions
875 of processes resulting in savings to the total carbon footprint saving (%).

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Fig. 3. Contribution analysis for the carbon footprint of energy consumed and energy generated and sold to the grid at the investigated wastewater treatment plants in 2015. The carbon footprint of energy (kg CO₂ eq) is expressed per Mg of plant input material (a) and per kg carbon (b), total nitrogen (c) and phosphorus (d) removed during treatment. The net carbon footprint of energy is the sum of burdens (positive values) and savings (negative values).

Fig. 4. Energy consumption for all investigated wastewater treatment plants in 2015. The energy consumption (kWh) is expressed per Mg of plant input material (a) and per kg carbon (b), total nitrogen (c) and phosphorus (d) removed during treatment.

Fig. 5. Energy balance of all investigated wastewater treatment plants in 2015. Energy consumption is reported as positive value, while energy sold to the grid is reported as negative value. The net energy is the sum of energy consumption and energy sold to the grid. (a) Net energy and its composition expressed as $(\text{kWh (Mg of input material)}^{-1})$. (b) Net energy and its composition expressed as $(\text{kWh (kg C removed)}^{-1})$. (c) Net energy and its composition expressed as $(\text{kWh (kg TN removed)}^{-1})$. (d) Net energy and its composition expressed as $(\text{kWh (kg P removed)}^{-1})$.

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900 Fig. 6. Scenario analysis of direct greenhouse gas emissions for the investigated wastewater treatment plants. The carbon footprint is
 901 expressed per functional unit, equal to 1 Mg of site-specific input material in 2015. Three scenarios are shown for each facility: the baseline
 902 scenario (Measurements), which includes plant-integrated and site-specific emission measurements, the Danish scenario (DNG estimation),
 903 which includes emission estimation according to the Danish national guidelines (DNG), and the international scenario (IPCC estimation),
 904 which includes emission estimation according to the guidelines provided by the Intergovernmental Panel on Climate Change (IPCC)
 905 (Doorn et al., 2006; Thomsen, 2016).

Fig. 7. Net carbon footprint of the investigated wastewater treatment plants expressed with different functional units referring to 2015. The functional unit of 1 Mg of treated material was normalized by carbon, total nitrogen and phosphorus influents in Figure (a1), (a2) and (a3), respectively. Figures (b), (c) and (d) show the results according to the functional units expressing carbon, total nitrogen and phosphorus removal capabilities, respectively. Error bars are the results of 100,000 Monte Carlo simulations calculating the propagation of uncertainty of the most sensitive model parameters.

914 **TABLES**

915 Table 1. Description of the investigated wastewater treatment plants in the inventory, 2015.

Plant (country)	Population equivalent (x 1000)	Additional processed material ^a	Wastewater treatment	On-site sludge treatment	Off-site biosolids treatment	Pollutant removal (%)				Energy output and use ^b
						COD	BOD	TN	P	
Lynetten (Denmark)	750	Sewage sludge	Bio Denipho	Digested and incinerated	None	93	NA	87	84	Methane to natural gas grid; Heat to district heating grid
Avedøre (Denmark)	265	Sewage sludge	Bio Denitro	Digested and incinerated	None	95	NA	86	92	Electricity to the grid
Lundtofte (Denmark)	150	None	Bio Denitro: Anaerobic MBBR	Digested and incinerated	None	90	96	88	87	Methane to natural gas grid
Holbæk (Denmark)	60	Landfill leachate	SBR and Deammonification process	Digested and biosolids stored for 2 days	Biosolids storage for 26 weeks before UoL	95	99	94	93	Electricity to the grid
Ryaverket (Sweden)	805	Sewage sludge; Food waste	Activated sludge; Nitrifying trickling filters; Post-Denitrifying MBBR	Digested and biosolids stored for 3 weeks	Biosolids storage for 23 weeks before UoL	86	94	65	94	Methane to natural gas grid and as vehicle fuels; Heat to district heating grid
Källby (Sweden)	120	Sewage sludge	Activated sludge	Digested and biosolids sent off- site daily	Biosolids storage for 26 weeks before UoL	NA	99	84	98	Methane to natural gas grid and as vehicle fuel; Heat to district heating grid
Växjö (Sweden)	95	Sewage sludge; Food waste	Activated sludge; HYBAS; Deammonification process	Thermal hydrolysis processed, digested, and biosolids stored for 26 weeks	None	90	NA	66	98	Methane to natural gas grid and as vehicle fuel

NA: Not available. MBBR: Moving Bed Bio Reactor. SBR: Sequencing Batch Reactors. HYBAS: Activated sludge combined with MBBR. UoL: Use on land of biosolids.

^a Material originating from external facilities and processed in the plant. ^b “Energy output” means energy produced on-site and sold to the energy system. Energy produced and used on-

site is not considered an output of the wastewater treatment plant. The pollutant removal (%) refers only to quantities in the wastewater: $\left(\frac{\text{influent mass}-\text{effluent mass}}{\text{influent mass}} \cdot 100\right)$.

916 Table 2. Carbon footprint potential of energy consumed and produced in the studied
917 wastewater treatment plants.

Type of energy	Energy system	Value	Unit	Reference
Electricity mix	Denmark	2.30E-1	kg CO ₂ eq (kWh) ⁻¹	National energy ministry
	Sweden	2.40E-2		
District heating mix	Denmark	1.08E-1	kg CO ₂ eq (kWh) ⁻¹	National energy ministry
	Sweden	2.40E-2		
Natural gas mix	Europe	4.94E-1	kg CO ₂ eq (kWh) ⁻¹	ELCD database 2.0
Diesel	Europe	2.48E-7	kg CO ₂ eq (kg) ⁻¹	ELCD database 3.0
Gasoline	Europe	7.19E-9	kg CO ₂ eq (kg) ⁻¹	ELCD database 3.0
Heating oil	Europe	2.48E-7	kg CO ₂ eq (kg) ⁻¹	ELCD database 3.0

918 More detailed references are available in Supplementary Data, Table S10.