Technical and environmental evaluation of an integrated scheme for the co-treatment of wastewater and domestic organic waste in small communities

L. Lijó*[1,2], S. Malamis[3], S. González-García[1], F. Fatone[4], M.T. Moreira[1] and E. Katsou[2]

[1]Department of Chemical Engineering, Institute of Technology, University of Santiago de Compostela, Spain.

[2]Department of Mechanical, Aerospace and Civil Engineering, Brunel University London, UK.

[3]Department of Water Resources and Environmental Engineering, School of Civil Engineering, National Technical University of Athens, Greece.

[4]Department of Biotechnology, University of Verona, Italy.

*Corresponding author email: lucia.lijo@usc.es

Abstract

A technical and environmental evaluation of an innovative scheme for the co-treatment of domestic wastewater and domestic organic waste (DOW) was undertaken by coupling an upflow anaerobic sludge blanket (UASB), a sequencing batch reactor (SBR) and a fermentation reactor. Alternative treatment configurations were evaluated with different waste collection practices as well as various schemes for nitrogen and phosphorus removal. All treatment systems fulfilled the required quality of the treated effluent in terms of chemical oxygen demand (COD) and total suspended solids (TSS) concentrations. However, only the configurations performing the short-cut nitrification/denitrification with biological phosphorus removal met the specifications for water reuse. The environmental assessment included the analysis of impacts on climate change (CC), freshwater eutrophication (FE) and marine eutrophication (ME). A functional unit (FU) of 2,000 people receiving treatment services was considered. The most relevant sources of environmental impacts were associated to the concentration of dissolved methane in the UASB effluent that is emitted to the atmosphere in the SBR process (accounting for 37% of impacts in CC), electricity consumption, mainly for aeration in the SBR (representing 13% of the impacts produced in CC), and the discharge of the treated effluent in receiving waters (contributing 98% and

57% of impacts in FE and ME, respectively). The scheme of separate waste collection together with biological nitrogen removal and phosphorus uptake via nitrite was identified as the best configuration, with good treated effluent quality and environmental impacts lower than those of the other examined configurations.

Keywords

Decentralised treatment processes; treated effluent reuse; environmental profile; nutrient removal; resource recovery; wastewater and domestic organic waste

Nomenclature

Abbreviation	Description
AOB	Ammonium oxidising bacteria
BNR	Biological nutrient removal
BOD_5	Five-day biochemical oxygen demand
CC	Climate change
COD	Chemical oxygen demand
DNBPR	Denitrifying via nitrite biological phosphorus removal
DO	Dissolved oxygen
DOW	Domestic organic waste
DPAO	Denitrifying phosphorus accumulating organism
EBPR	Enhanced biological phosphorus removal
FE	Freshwater eutrophication
FWD	Food waste disposer
GHG	Greenhouse gas
HRT	Hydraulic retention time
LCA	Life cycle assessment
ME	Marine eutrophication
ND	Nitrification/denitrification
NOB	Nitrite oxidising bacteria
OLR	Organic loading rate
PAO	Phosphorus accumulating organism
PE	Population equivalent
SBR	Sequencing batch reactor
scND	Short-cut nitrification/denitrification
sNUR	Specific nitrogen uptake rate
sPUR	Specific phosphorus uptake rate
SRT	Solids retention time
TN	Total nitrogen
TP	Total phosphorus
TS	Total solids
TSS	Total suspended solids
UASB	Upflow anaerobic sludge blanket
VFA	Volatile fatty acid
vNLR	Volumetric nitrogen loading rate
vPLR	Volumetric phosphorus loading rate
VS	Volatile solids
WWTPs	Wastewater treatment plants

1. Introduction

Centralised wastewater treatment may not be feasible or the most cost-effective option for all sites. For instance, due to geographical conditions and dispersed settlements, more than 9,000 wastewater treatment plants (WWTPs) in Italy are designed for 2,000 population equivalent (PE) or lower (Libralato et al., 2012). The European legislation on urban wastewater treatment defines discharge limits for biochemical oxygen demand (BOD_5), chemical oxygen demand (COD) and total suspended solids (TSS) for WWTPs serving PE higher than 2,000, while for lower agglomerations, it only states that appropriate treatment must be implemented (EEC, 1991). Moreover, when it comes to nutrient concentrations, limitations for total phosphorus (TP) and nitrogen (TN) are only specified for treated effluents from facilities with a treatment capacity larger than 10,000 PE discharging into sensitive recipients. The option of reusing the treated water from small scale WWTPs in agriculture is interesting, provided that the treated effluent is available near the potential points of use, thus, decreasing the costs of reclaimed water distribution systems (Hophmayer-Tokich, 2000). Currently, there is no European Union legislation concerning the use of reclaimed water; therefore, countries should apply national or regional regulations (Norton-Brandão et al., 2013).

Considering the requirements imposed for the treated effluent, the applied treatment process should accomplish a number of objectives: relatively low capital and operating expenses, reduced energy consumption and enhanced reuse potential of water and other valuable by-products, such as biogas. In this context, the application of anaerobic processes, i.e. upflow anaerobic sludge blanket (UASB), appears as a robust and attractive technology, particularly for hot climates (Latif et al., 2011). Compared to aerobic treatment, the UASB process has several advantages, such as low operating expenses, high efficiency, simplicity, flexibility, low requirements of space, energy and chemicals as well as reduced sludge production (Latif et al., 2011). However, there are still some barriers that limit the use of anaerobic processes, including the process instability at temperatures below 20°C, low pathogen removal, negligible nutrient removal, odours, long start-ups and the need for adequate post-treatment (Latif et al., 2011). In addition, it is important to consider the concentration of dissolved methane in the anaerobic effluent since low temperature raises methane solubility, which promotes its release into the environment (Cookney et al., 2016, 2012; Matsuura et al., 2015).

Biological nutrient removal (BNR) from the low strength anaerobic effluent can be applied as a polishing step (Frison et al., 2013b; Malamis et al., 2013). Biological nitrogen removal via nitrite has several benefits compared to conventional nitrification/denitrification such as 25% of oxygen savings during nitrification and 40% less need for organic carbon source during heterotrophic denitrification (Galí et al., 2007). Enhanced biological phosphorus removal (EBPR) can be performed using nitrite as electron acceptors (Katsou et al., 2015). Denitrifying via nitrite biological phosphorus removal (DNBPR) offers the possibility of integrating phosphorus and nitrogen removal in a robust process. In the presence of nitrite and lack of oxygen, nitrite is denitrified to gaseous nitrogen and simultaneously, phosphate is taken by denitrifying phosphorus accumulating organisms (DPAOs) (Peng et al., 2011). DPAOs are able to accumulate significant amounts of polyphosphate under anoxic conditions, similarly to the phosphorus accumulating organisms (PAOs) in the conventional EBPR process.

Due to the substantial organic matter removal attained in the anaerobic treatment, the addition of an external carbon source is required in the subsequent aerobic process for effective BNR (Frison et al., 2013a). The latter opens up the possibility of integrating the management of domestic organic waste (DOW) with sewage. The use of organic waste (i.e. fermented liquids) from households as external carbon source achieves satisfactory rates of denitrification and phosphorus accumulation, while decreasing operational costs (Frison et al., 2013a). Food waste disposers (FWDs) are being promoted as an alternative practice for the collection of DOW (Iacovidou et al., 2012). Specifically, the implementation of FWDs entails reduced transport requirements and odours when compared to the conventional collection (Battistoni et al., 2007; Bernstad et al., 2013). However, the environmental assessment of FWD use is required, with specific focus on energy demand, water consumption and increased organic loads in the WWTP (Battistoni et al., 2007; Marashlian and El-Fadel, 2005).

Several works on the environmental performance of WWTPs have been published following the life cycle assessment (LCA) approach. Rodriguez-Garcia et al. (2011) assessed the environmental impact of 24 WWTPs, classifying them in six different typologies by the quality requirements according to their final use or discharge point. Besides, LCA has also been applied for the environmental assessment of integrated processes for waste and wastewater management. Weichgrebe et al. (2008) compared, in terms of energy and environmental impact, the conventional, separate treatment of wastewater and organic waste with their combined treatment by psychrophilic anaerobic digestion and aerobic post-treatment, which showed greenhouse gas (GHG) savings compared to conventional wastewater treatment. Nakakubo et al. (2012) compared different technologies for the disposal of sewage sludge and food waste in order to identify the best option regarding the reduction of GHG emissions. Similarly, Righi et al. (2013) analysed the environmental profile of a decentralised scheme for the management of sewage sludge and biodegradable municipal solid waste (MSW). However, no LCA study has been conducted on the assessment of the environmental performance of anaerobic – aerobic processes for domestic sewage and DOW at community level.

This work evaluates the feasibility of an integrated system designed for the decentralised co-management of wastewater and DOW in a small community of 2,000 PE. Various scenarios were evaluated including (i) alternatives in the collection of DOW regarding the integration rates of FWDs within the community, (ii) different nitrogen removal processes and (iii) the potential of including phosphorus removal in the treatment scheme.

2. Materials and methods

2.1. Integrated treatment system: UASB – SBR configuration

The selection of the treatment configuration was based on the results of a pilot scale UASB-SBR system operating at the premises of the University of Verona, taking into consideration cost criteria, legislative aspects for the treated effluent and DOW management in Italy, as well as the characteristics of the small community in terms of waste collection and sewage management. The treatment scheme included: (i) an UASB reactor to treat sewage and produce biogas (ii) a fermentation process to produce volatile fatty acids (VFAs) from DOW, a sequencing batch reactor (SBR) to remove nutrients from the UASB effluent and produce reusable water (iii) composting to treat the excess sludge and convert it to compost to be applied as soil conditioner. Nutrient removal in small scale wastewater treatment systems is not required by European Legislation; however, depending on the relevant National and or Regional Law of countries, compliance to specific nitrogen and phosphorus limits of the treated effluent before discharge to specific water recipients. Mass balances were developed for the whole treatment scheme for total solids (TS), volatile solids (VS), COD and nutrients (TN and TP) to model the different streams of the treatment system. The flowchart of the baseline treatment scheme is shown in Figure 1.

Figure 1 around here

UASB process

The average sewage flow was 400 m^3/d, assuming a production of 200 L wastewater/capita d for a community population of 2000 people. The production rates of COD, N and P were taken as 120 g COD/capita d, 12 g N/capita d and 1.8 g P/capita d, respectively. The UASB reactor operated at ambient temperature (22±2°C), at an average organic loading rate (OLR) of 1.4-2.1 kgCOD/$m^3_{reactor}$ d, a hydraulic retention time (HRT) of 8 h and an upflow velocity of 1 m/s. According to the experimental results, the UASB produced 7.2-13.2 L/d of biogas with a methane content ranging between 60-65%, which corresponded to an average experimental methane yield of 0.26 m^3CH_4/kg $COD_{removed}$. This value is lower compared to the theoretical value of methane yield of 0.35 m^3 CH_4/kg $COD_{removed}$ since it does not include the dissolved methane present in the UASB effluent which is not recovered. For calculation purposes, global removal efficiencies of 77% and 70% were considered for COD and TSS respectively, assuming that 1 kg of COD degraded produced 0.26 m^3 of methane. The dissolved methane derived from the operation of the UASB at moderate temperature was also taken into account in terms of its environmental impact, by considering concentrations of 20 mg CH_4/L under supersaturation conditions (Souza et al., 2011). The biogas produced in the UASB was treated in a biotrickling filter to remove hydrogen sulphide. The biogas was burnt in a boiler for heat production, appropriate for small and decentralised systems. The boiler had a thermal efficiency of 90% and 10% of losses.

Fermentation process

Considering that the UASB effluent had a very low COD/N ratio (2.5 kg COD/kg N), fermentation of DOW was applied to produce VFAs, which would then be fed to the SBR to promote nutrient removal from the

UASB effluent. Furthermore, the surplus of fermented DOW was sent to the UASB in order to increase the OLR and, thus, biogas production. A production rate of 0.30 kg DOW/capita d was considered (Bolzonella et al., 2003). Assuming a collection efficiency of 83%, 500 kg of DOW are separately collected at household level and transported to the treatment facility on a daily basis for a community of 2,000 inhabitants. Regarding its physicochemical composition, TS of DOW was 25%, including 1,200 mg COD/g TS, 25 mg N/g TS and 3 mg P/g TS. Moreover, its content in carbohydrates was 600 g/kg VS, in proteins 200 g/kg VS and in sugars 160 g/kg VS. In the baseline configuration, DOW was firstly ground to produce a homogeneous mixture and then diluted with water up to 6% TS. The fermenter was fed at an average OLR of 11 kg COD/m^3 d and operated at 35°C and at a HRT of 5.2 d (Katsou et al., 2015; Lee et al., 2014; Traverso et al., 2000). During the fermentation process, organic matter is converted to acids (e.g. acetic acid, butyric acid, lactic acid etc.) while CO_2 is also released as a result of metabolic processes. Furthermore, hydrolysis of organic nitrogen and subsequent ammonification takes place, pH increases and some ammonia is released into the atmosphere. It was assumed that 8.5% of COD was converted into carbon dioxide and methane, and losses of TN (2%) as ammonia and TS (2%) take place (Battistoni et al, 2002).

Dewatering unit

The fermented DOW and the excess sludge from the UASB and SBR were separated into a liquid and a solid fraction by applying a screw-press. Therefore, a liquid fraction rich in VFAs was produced to be used in the SBR and a solid fraction to be further treated in the composting unit. The separation efficiencies of fermented DOW and sludge are different. More specifically, in the case of fermented DOW 65% of TS, 40% of COD and TN and 50% of TP are transferred to the solid stream and the remaining in the liquid fraction; when sludge was separated 95% of TS, COD, TN and TP ended up in the solid fraction (Albertson et al., 1991; Battistoni et al., 2002). The produced solid fraction was sent to the composting unit, while the liquid fraction was stored in an equalisation tank with a HRT of 10 h before being fed to the SBR.

SBR

The SBR was applied as a post-treatment stage of the UASB effluent and of the liquid stream generated from the screw-press. The SBR cycle comprised of filling, the sequential operation under anaerobic, aerobic and anoxic conditions, settling and decanting. The system operated at low dissolved oxygen level (around 1 mg/L) in order to perform short-cut nitrification/denitrification (scND) instead of the conventional nitrification/denitrification (ND) process. Previous work has shown that the combination of a suitable vNLR and low DO can result in effective via nitrite nutrient removal from domestic sewage (Katsou et al., 2015). The calculation of the oxygen demand was based on the organic carbon and ammonia load.

Composting process

Sludge composting took place in an enclosed system equipped with a biofilter (Colón et al., 2009). Wheat straw was used as a bulking agent and was mixed with sludge in order to improve aeration, to provide a C/N ratio in the range of 25:1-35:1 and adjust the moisture content of the mixture in the range of 60-65% (Hernandez et al., 2006; Tremier et al., 2005). The addition of the bulking agent also prevented the compost mixture from excessive compacting. The straw had the following characteristics: 90% of TS out of which 90% were VS, 60% of total carbon, 0.9% of TN and 0.1% of TP (Rihani et al., 2010).

2.2. Integrated treatment schemes for wastewater and DOW management

Alternative approaches were examined to identify the best treatment configuration from a technical and environmental point of view. More specifically, three options were analysed considering different integration levels of FWDs in the community, diverse nitrogen removal options in the SBR and the possibility of including phosphorus removal.

DOW collection

The collection of DOW in the community was considered with various FWDs integration rates. These disposal units are equipped with a shredding system, allowing effective collection of DOW, which is pumped together with wastewater to the treatment plant.

- Configuration 1 involved the separate collection of wastewater and DOW (0% FWDs integration). Wastewater was pumped from the households to the WWTP, whereas DOW was separately collected at households and transported by trucks to the WWTP.

- Configuration 2 included FWD integration in 50% of the households in the community (Evans et al., 2010). The remaining DOW that was not managed through FWDs was transported by trucks to the treatment plant.

- Configuration 3 considered complete integration of FWDs (100%) in the community. Wastewater and DOW streams produced were delivered together to the WWTP.

The introduction of FWDs has been reported to cause an additional load of 60 g TSS/capita d, 95 g COD/capita d, 2.1 g N/capita d and 0.3 g P/capita d in the influent wastewater (Bernstad et al., 2013; De Koning, 2003). The application of FWDs leads to an increase in tap-water consumption (up to 4.5 L/capita d) required for pumping wastewater and DOW to the WWTP (Bernstad et al., 2013; Rosenwinkel and Wendler, 2001). A primary settler was implemented before the UASB to receive the mixture of wastewater and shredded DOW for the effective settling of the primary sludge (Figure 2). The removal efficiencies of COD, TSS, TP and TN in the primary settler were assumed to be 30%, 50%, 10% and 5%, respectively (Tchobanoglous et al., 2014). The produced sludge was fed to the fermentation unit to produce VFAs, while the supernatant was fed into the UASB.

Alternative processes for nitrogen removal

Biological nitrogen removal was integrated in the scheme by applying conventional nitrification/denitrification (ND) and short-cut nitrification/denitrification (scND) in the SBR.

Phosphorus removal

Biological phosphorus removal using oxygen and nitrate or nitrite as electron acceptors in the SBR was also evaluated. Nitrogen and phosphorus removal accomplished under anoxic conditions require lower amounts of external carbon source and energy compared to aerobic conditions (Malamis et al., 2013). The tested configurations are summarised in Table 1.

Table 1 around here

2.3. Environmental impact of the decentralised schemes

This section includes the quantification of the environmental impact of each configuration for the identification of the most favourable one from an environmental point of view, using the LCA methodology (ISO 14040, 2006). The functional unit (FU) selected was the treatment of the wastewater and DOW produced by a community of 2,000 PE per day.

System boundaries

The processes considered within the system boundaries of the tested configurations are outlined in Figure 2. The generation of waste streams (wastewater and DOW) was excluded from the environmental analysis, since it does not affect the resource valorisation. The sewer network has an important contribution to the total environmental impact of wastewater management (Doka, 2007). However, in this work, the sewer system was excluded for the purpose of comparison because it was considered to be similar for all scenarios.

In LCA studies, when waste treatment systems are converted into alternatives for resource recovery, they are usually credited by considering the avoided environmental impacts of producing a different product with the same function (Finnveden et al., 2005). In this manner, the environmental benefits of the production of valuable products can be quantified. The produced heat from biogas was partially used to heat the fermentation reactor, while the surplus heat can be exploited for heating nearby households 8 months per year. Fuel oil was assumed as the fuel used for accounting the environmental credits, since it was considered the most appropriate for a small and decentralised community.

Numerous studies have demonstrated the horticultural properties of compost, being able to substitute peat in the production of ornamental plants (Ceglie et al., 2015; Russo et al., 2011), although its fertiliser capacity is lower than that of other organic substrates such as manure or digestate (De Vries et al., 2012). Therefore, it was assumed that the produced compost can be used as soil conditioner avoiding the extraction, transport and use of a similar quantity of peat (Boldrin et al., 2009; Saer et al., 2013).

Figure 2 around here

Inventory data

Inventory data regarding all inputs and outputs for each configuration were based on experimental results from the UASB-SBR pilot plant and mass balances. A description of the bibliographic sources used to build the life cycle inventory is given in Table 2. A detailed description of inventory data of the base case can be found in Table S2 of the supplementary material.

Table 2 around here

The ecoinvent® database (2016) was used to introduce background data for production of electricity, heat from fuel oil and peat (Dones et al., 2007), manufacturing of chemicals (Althaus et al., 2007), transportation (Spielmann et al., 2007) and waste disposal (Doka, 2007). Concerning the production of electricity, the process included in the database has been updated using data for the average electricity generation and import/export data for Italy in 2014 (Terna Rete Italia, 2015).

Impact assessment methodology

This section describes the methodology used to select representative impact categories. Direct and indirect GHG emissions and eutrophication attributed to the discharge of the treated effluent have important environmental impacts in most WWTPs (Rodriguez-Garcia et al., 2011). Hence, among all the available impact categories within the LCA methodology, only three were considered: climate change (CC), freshwater eutrophication (FE) and marine eutrophication (ME). In particular, CC estimates the contribution of the system to the global warming effect and it is influenced by the amount of direct and indirect GHGs. The categories of FE and ME measure the potential enrichment of nutrients in water bodies (freshwater and marine water, respectively). FE is affected by phosphorus-based substances, while ME accounts for nitrogen-based compounds. Finally, the potential impacts regarding CC were determined by considering the characterisation factors provided by IPCC (2013), while the potential damages due to FE and ME were measured through the characterisation factors reported by the ReCiPe Midpoint H methodology (Goedkoop et al., 2009).

The relation between the background processes and CC used in the study is shown in Table S3 of the supplementary material.

3. Results and discussion

3.1. Effluent quality, bioenergy generation and compost production

This SBR had a HRT of 10 days, a solids retention time (SRT) of 18 days, a volumetric nitrogen loading rate (vNLR) of 0.15 kg N/m^3 d and a volumetric phosphorus loading rate (vPLR) of 0.022 kg P/m^3 d. These parameters were considered to be invariable among all the configuration schemes Regarding dissolved oxygen (DO) concentration in the aerobic reactor, the SBR performing BNR via nitrate operated at DO concentrations of 2 mg/L; whereas the DO level was kept close (and even below) 1 mg/L in the process via nitrite. It was observed that under these conditions the ratio of NO_2-N/NOx-N gradually increased and was steadily maintained above 99% during the operation of the SBR. In the ND configurations, sNUR was on average 2.02 g N/kg VSS h, while in the scND configurations, the sNUR was on average 4.93 g N/kg VSS h, as supported by experimental results. The lower needs of external carbon source in the BNR process via nitrite can maintain higher average sNUR in the reactor. When enhanced biological phosphorus removal was performed, the pathway schemes integrating processes via nitrite resulted in slightly higher specific phosphorus uptake rates (sPUR) compared to the processes via nitrate: 3.85 g P/kg VSS h and 3.19 g P/kg VSS h, respectively. Table 3 shows the characteristics of the treated effluent as this is calculated for each scenario in terms of COD, TSS, TN and TP.

Table 3 around here

All the scenarios achieved COD levels between 36.4 and 69.5 mg/L and TSS concentrations from 14.1 - 25.9 mg/L in the treated effluent; therefore the treated effluent met the EU limits of COD and TSS for discharge into water bodies. However, the quality of the treated effluent regarding nutrients and TSS was not appropriate for reuse. Regardless of the waste collection strategy, only the systems which performed the BNR through the short-cut nitrification/denitrification together with biological phosphorus uptake via nitrite (scND-P configurations) is able to reduce the nutrients to the levels required by existing National standards in Europe in. In any case, to comply with the reuse criteria tertiary treatment (coagulation and sand filtration) followed by appropriate disinfection is required. The configurations applying scND achieved 85-86% nitrogen removal, while the nitrogen removal efficiency was 67-85% for ND configurations. Phosphorus removal, when applied, was higher than 80% for BNR via nitrite and around 43-73% via nitrate. The relation between the carbon source supplied and the one required for the BNR process is the reason behind these results, since in some configurations such as Configuration 1-ND, the carbon source required for the denitrification process is significantly higher than the one that is available and is thus supplied by the system. In this case, high levels of external carbon source were required for BNR in the conventional treatment system. The COD consumed for denitritation ranged from 49.6-57.2 kg COD/day, while the COD required for conventional denitrification via nitrate varied from 61.2-99.8 kg COD/day. The latter was not enough to remove nitrogen and this is the reason why the nutrient concentrations of the treated effluent are higher in the conventional nitrification/denitrification processes. Diverting fermented DOW liquid from the UASB to the SBR resulted in lower biogas production in the UASB (Table 4). More specifically, the application of conventional ND allowed recirculation rates of fermented liquid to the UASB from 0%-11% of the amount of fermented liquid produced, while the respective recirculation rates were up to 45% for the scND scheme. As a result, when scND was performed, the average biogas production was usually higher. Regarding the food waste collection options, the use of FWDs (Configurations 2 and 3) increased the COD levels at the head of the plant. After primary settling, the settled sludge was fed to the fermentation unit to produce VFAs, while the clarified effluent was sent to the UASB. Part of the fermented liquid was sent to cover the BNR needs of the SBR and the remaining part to the UASB to increase biogas recovery. More specifically, 59% and 52% of the inlet COD was fed to the UASB in configurations 2 and 3, respectively.

Table 4 around here

The treatment and disposal of sludge is an important issue in WWTPs (Wei et al., 2003). The most commonly applied methods for sludge disposal at EU level include landfills, land application and incineration. In the examined systems, sludge was valorised through composting. The compost properties must be in line with the quality assurance protocol. As seen in Table 4, sludge production was directly affected by the food waste collection system. The partial or total application of FWDs (Configurations 2 and 3) resulted in higher sludge production compared to the separate collection schemes (Configuration 1) (around 18-19% increase). The larger sludge production is attributed to the operation of the primary settler required when FWDs are used. The implementation of the primary settler implies the separation of primary sludge that is further sent to the fermentation reactor. Furthermore, the amount of sludge produced was 2-4% lower in the scND configurations. Finally, the schemes with EBPR produced more sludge than the respective ones without EBPR (around 3-7% increase).

3.2. Environmental profile of the UASB – scSBR configuration

Table 5 summarises the LCA characterisation results for the scND configuration per functional unit, split up by the processes involved. Positive values indicate environmental burdens, whereas negative values are indicative of environmental credits.

Table 5 around here

Despite the differences in environmental results among the examined impact categories, it is important to highlight the general positive effect of avoided processes. Avoided peat use had a modest contribution, while avoided heat production from fuel oil played an important role in offsetting the GHG emissions. In LCA studies, the choice of avoided products has a strong influence in the results. For this reason, a detailed analysis of the influence of these methodological assumptions was performed in section 3.5. In addition, the treatment scheme under assessment was evaluated excluding environmental credits; therefore, each impact category was examined in detail considering only negative loads in order to identify the system components with greater environmental impacts. Figure 3 summarizes the relative contributions of each process to CC for the baseline scenario.

Figure 3 around here

Regarding the environmental impact in CC (467 kg CO_2 eq/FU), the electricity requirements contributed up to 13% of the global impact produced in CC. From the total electricity consumed in the treatment plant, aeration in SBR accounted for 60%. The environmental impact of the consumption of electricity is directly linked to the electricity mix of the specific country under study. In this case, the Italian electricity profile produces 0.46 kg CO_2 eq/$kWh_{produced}$. If the treatment plant would have a solar system installed, the emissions would be only 0.08 kg CO_2 eq/$kWh_{produced}$ (Dones et al., 2007). Direct emissions in the SBR unit (76 kg CO_2 eq/FU) were significant contributors to the environmental profile of the treatment scheme, representing 37% of the total impact produced in CC. Dissolved methane from the UASB effluent is by far the most influential compound (201 kg CO_2 eq/FU), followed by nitrous oxide emitted from the SBR (11 kg CO_2 eq/FU). Emissions derived from the composting unit also contributed with 13% of the impacts in CC; these environmental impacts were related to direct emissions of methane and nitrous oxide that were generated during biomass decomposition. Despite the fact that composting is an aerobic process, methane emissions may occur, especially for enclosed systems, in anaerobic pockets of the substrate/mixture that is composted (Boldrin et al., 2009). Methane and nitrous oxide emissions derived from the application of compost on land were accounted according to Bruun et al. (2006); representing 13% of the total impact produced in CC. In addition, these processes were also a source of carbon dioxide; particularly, the composting and the SBR process. However, due to the natural carbon cycle, carbon from biogenic sources can be considered as climate-neutral, since the equivalent amount of carbon dioxide emitted from an organic source is previously uptaken in the photosynthesis. Other minor sources of GHG emissions were waste disposal in landfill (10%), infrastructure (4%), transportation (3%) and biogas losses (2%).

The relative contributions of each process to the eutrophication related categories are outlined in Figure 4. The discharge of treated effluent was the main source for eutrophication emissions, contributing up to 98% in FE and 57% in ME. Emissions of phosphorus from the treated effluent were responsible for FE, while nitrogen emissions from the treated effluent were related to ME. In addition, leachates of nitrate derived from the application of compost on land had an important contribution in ME (37%).

Figure 4 around here

3.3. Towards increased environmental sustainability

The different treatment configurations were analysed in terms of their environmental profile to identify the most sustainable scheme. Characterisation results for each configuration are given in Table 6. Figure 5a, b and c summarizes the comparative results for each impact category.

Table 6 around here

Figure 5 around here

In terms of CC, the profile of the system depended on the collection scheme (Figure 5a). The partial use of FWDs (Configuration 2) resulted in GHG emissions between 1.80 and 2.17 times higher than in the baseline scenario. As shown in Table 5, the use of FWDs resulted in less methane generation and subsequently, in lower environmental credits due to avoided heat production that highly affected this category (Table 4). Additionally, this collection system increased the production of sludge, increasing the environmental impacts from direct emissions of nitrous oxide and methane from composting and land application processes. Finally, the implementation of FWDs was associated with additional energy consumption compared to the separate collection of DOW (Table 2). Concerning the removal of nutrients, the denitrification process entailed different emissions and aeration requirements, resulting in different electricity consumption (Table 2). In particular, energy consumed for air supply was 14% higher in the Configurations performing nitrogen removal via nitrate than via nitrite. As a consequence, ND configurations exhibited 15% more environmental impact on average regarding CC than scND configurations.

As shown in Figure 4, the discharge of the treated effluent was the most important contributor to FE and ME impact categories. Therefore, the treated effluent quality can explain the differences observed in Figures 5b and 6c among the examined configurations. A reduction of 28%-82% in the FE category was observed in the systems that perform EBRP, since this impact category is only influenced by phosphate-based emissions. Finally, the configurations that perform nitrogen removal via nitrate together with EBPR resulted in high nitrogen levels in the final effluent since the carbon source was not enough for complete denitrification. This adversely affecting ME, which was 44-90% higher compared to the baseline scenario.

3.4. Sensitivity analysis

A sensitivity analysis was performed to analyse three selected, key parameters: (i) the COD removal efficiency in the UASB process, (ii) the efficiency in the separate collection of DOW at household level and (iii) the bulking agent used for composting. The outcomes from the sensitivity analysis are presented in Tables S4, S5 and S6 of the supplementary material, respectively.

COD removal efficiency in UASB. COD removal of 77% was considered in the UASB for calculations as the base case. In sensitivity analysis 1 (SA1), it has been considered that only around 50-55% of the COD input was removed, which implied lower biogas and sludge production in the UASB. In addition, in spite of the higher COD concentration in the UASB effluent compared to the base case, the quality of the treated effluent has low nutrient concentrations in Configurations 2 and 3-scND-P allowing the potential reuse. However, this resulted in higher energy consumption for aeration and sludge production in the SBR process. Therefore, the decrease in biogas production (29-35% lower compared to the base case) was not only attributed to lower COD removal but also to lower flow of fermented liquid recirculated to the UASB reactor. Moreover, total sludge production in the treatment scheme was higher, which was attributed to the sludge production in SBR (48-77% more), which ended up in a larger production of compost (10-19%). From an environmental perspective, all these changes entailed an increase in GHG emissions by 65 kg CO_2 eq/d with respect to base case.

DOW collection efficiency. In the base case assessment, it was assumed that 83% of DOW produced at household level was separately collected and delivered to the treatment facility. In Configurations 1, this meant that from the total 600 kg DOW produced each day, 500 kg were delivered to the treatment facility; in Configurations 2 this meant that 250 kg DOW/d were delivered to the facility because the remaining 300 kg were collected through FWDs. Configurations 3 were not influenced since all DOW produced was collected in FWDs. However, the values of collection efficiency can vary from one community to another. Therefore in sensitivity analysis 2 (SA2), it was assumed that only 40% of the produced DOW was successfully separated in the households. The lower collection efficiency resulted in lower amount of DOW sent to the fermentation reactor, implying less available carbon source and/or biogas production, but also greater amount of DOW sent to landfill. Under this assumption, although 100% of the fermented liquid is sent to the SBR in Configurations 1, it is not enough for efficient nutrient removal and the treated effluent is characterized by elevated nutrient concentration. Conversely, the collection efficiency had a lower influence in Configurations 2, where the supply of DOW is guaranteed by the implementation of FWDs in 50% of the households. In these cases, the quality of the effluent was similar in comparison to the base case due to the effective collection of DOW in the FWDs, which allowed the proper supply of carbon source in the SBR. Moreover, the total amount of sludge produced and compost are lower especially in Configurations 1 due to the lower amount of DOW handled; however, due to the higher amount of organic waste sent to landfill and the lower amount of biogas produced, the environmental impacts produced in Configurations 1 were on average around 40 kg CO_2 eq/FU higher.

Bulking agent used for composting. In the base case, wheat straw was used as bulking agent in the composting process. In sensitivity analysis 3 (SA3), wheat straw was substituted by sawdust. The change in the bulking agent meant different compost mixture composition, resulting in different emissions from composting and from compost application. Wheat straw had a composition in terms of 10% moisture, TC, TN and TP of 60%, 0.9% and 0.1%, respectively as percent of dry solids; the composition of sawdust was 20% moisture, 60% TC, 0.2% TN and 0.03% TP. The lower content in nutrients resulted in (i) lower amount of sawdust required to achieve the appropriate C/N ratio and (ii) lower emissions of nutrient-based compounds derived from the composting process and the application of compost on land (including emissions of nitrous oxide and ammonia and leachates of nitrate and phosphate). In terms of CC, this change meant a reduction in GHG emissions of 6% in average, while in ME the global impacts produced by the treatment schemes proposed were reduced up to 10%. However, no significant changes occurred in terms of FE (<0.1%), since almost all of the effects produced in this impact category (>98%) were allocated to the discharge of the treated effluent.

3.5. Assessment on the reliability of the environmental results

The influence of the selection of important parameters in the environmental balance was assessed. A comparison between the baseline case and alternative scenarios was performed to identify sensible variations in the results.

Biogas losses. Fugitive biogas emissions from anaerobic processes are usually included in the environmental analysis. These emissions directly affect CC, not only due to direct methane emissions, but also by decreasing the potential heat production from biogas. In the baseline scenario of the current study, 1.5% of biogas produced was taken into account as biogas losses in accordance to De Vries et al. (2012). Poeschl et al. (2012) considered that these losses can vary from 1 to 1.8%. A sensitivity analysis was performed to assess the influence of different rates of biogas losses in CC (i.e. 1% and 1.8%). The decrease of the emissions to 1% of the biogas produced can save from 3-4 kg CO_2 eq/FU; whereas when the biogas losses were 1.8%, the environmental profile can increase by 5-7 kg CO_2 eq/FU. Therefore, this assumption had a slight effect on the definition of the environmental profile (±1%).

Avoided products. As described in Section 3.2., credits from the avoided products played an important role in offsetting the environmental impacts of the applied treatment scheme, especially regarding CC. Alternative avoided products were analysed to identify their impact. The baseline case where the avoided heat was produced from fuel oil at small-scale in Europe, was compared with the substitution of heat produced from different fuels, such as natural gas and hard coal (Dones et al., 2007). The substitution of peat for compost is usually done on a 1:1 volume basis (Boldrin et al., 2009). In this study, identical density was assumed for compost and peat. However, Boldrin et al. (2009) stated that compost and peat densities are very variable and can be different; it is possible that 1 tonne of compost can replace the use of 0.2-1 tonne of peat. Accordingly, an equivalence of 0.2, 0.6 and 0.8 t peat/t compost was considered. Concerning avoided heat, the production of heat from fuel oil generates 0.32 kg CO_2 eq/kWh (base case), while the environmental impact of heat production from natural gas and hard coal is 0.26 and 0.57 kg CO_2 eq/kWh, respectively. Considering natural gas as the substitute fuel, the environmental impacts can increase around 13-24 kg CO_2 eq/FU; whereas, when considering hard coal, it can be improved by 35-63 kg CO_2 eq/FU; meaning an environmental profile 5-13% lower compared with the base case. Therefore, the substitution of heat from fuel oil to heat from hard coal has a considerable effect. With regard to avoided peat, the lowest replacement ratio (0.2 t peat/t compost) means an increase of the environmental profile up to 3% (~15 kg CO_2 eq/FU).

Treated effluent reuse. The quality of the treated effluent in the scND-P configurations met the specifications for water reuse in Italy provided that effective tertiary filtration and appropriate disinfection take place (Section 3.1.). Therefore, the treated water can be reused for irrigation instead of being discharged in water bodies. This practice reduces the impact of direct discharge of nutrients; however, it entails other potential environmental burdens from the filtration and disinfection as well as the use of agricultural machinery and emissions derived from the treated effluent discharge on land. In the sensitivity analysis, it has been considered that the effluent is further treated in a sand filter using aluminium sulphate as coagulant, followed by UV disinfection, as described in (Meneses et al., 2010). In addition, derived emissions were computed using the methodology described in IPCC (2006).

As shown in Figure 6, the use of treated effluent for irrigation had an adverse impact in the environmental profile regarding CC (by 18-20%) due to the tertiary treatment as well as the use of agricultural machinery for irrigation. On the contrary, the performance was improved by 69-83%% and 37-41%% for FE and ME, respectively, due to the reduction of direct P and N emissions into water recipients.

Figure 6 around here

LCA works dealing with wastewater treatment have also identified environmental benefits (i.e. replacement of mineral fertilisers) from the use of reclaimed water for agricultural purposes (Meneses et al., 2010). The application of scND-P configurations can result in savings of 3.85-4.10 kg N/FU and 0.95-1.59 kg P_2O_5/FU of nitrogen and phosphorus based fertilisers. This results in a reduction of approximately 44.7 kg CO_2 eq/FU, 8.24 g P eq/FU and 10.5 g N eq/FU for CC, FE and ME, respectively, which enhances the environmental profile of the systems by 11-17%.

3.6. Wastewater treatment alternatives in small communities

Several systems have been reported in literature for wastewater treatment in small and decentralised communities. The most common treatment scheme for wastewater treatment in small communities is constructed wetlands (Barros et al., 2008; Chan et al., 2008; Wu et al., 2011; Ye and Li, 2009). Other configurations have also been proposed, such as trickling filter, activated sludge, membrane bioreactor or extended aeration (Molinos-Senante et al. (2012) and an integrated step-feed biofilm process (Liang et al., 2010). A review of different schemes designed for the treatment of domestic wastewater at decentralised level can be found in Table 7. In more detail, Nogueira et al. (2009) compared the economic and environmental profile of energy-saving and intensive wastewater treatment systems. Energy-saving technologies such as slow rate infiltration plants and constructed wetlands exhibited better results compared to the activated sludge processes. Yildirin and Topkaya (2012) evaluated the environmental behaviour of constructed wetlands, vegetated land and activated sludge (with and without phosphorus removal), which reported similar results in CC impact but also in terms of the eutrophication-related categories.

Table 7 around here

Regarding more advanced treatment technologies, Zeeman et al. (2008) analysed the operational performance of UASB for the separate treatment of both grey and black water. Grey water was treated in a UASB-SBR system, while a struvite precipitation process was applied after the UASB process for black water. The comparison of the proposed treatment configurations with conventional sanitation showed energy savings of 200 MJ/PE year and phosphorus recovery via struvite of 0.14 kg P/PE year. In comparison with our study, important water reductions related with the use of vacuum toilets are shown. The production of grey and black water was in the range of 60-90 L/PE year and 6.8-7.5 L/PE year, respectively, while a production of 73 m^3/PE year was considered in the current work. Energy consumption was estimated as 151 MJ/PE year. Despite the differences among the treatment systems examined in the current work, similar results were obtained in configurations 1 (160 MJ/PE year), while in configurations 2 and 3, energy consumption was higher (300 MJ/PE year). Alternatives of the conventional SBR were also analysed in the literature, including the performance of a sequencing batch membrane bioreactor (SBMBR) (Krampe, 2013). One of the advantages of coupling a membrane to a SBR is the reduced cycle time as a result of the elimination of the settling phase and complete elimination of suspended solids in the treated effluent. However, they are associated with higher operating costs due to membrane fouling.

4. Conclusions

The technical evaluation of the systems revealed:

- The co-management of wastewater and DOW is feasible for a small community (i.e. up to 2,000 PE), regardless of the applied collection scheme; the treated effluent met the discharge requirements.
- The removal of nitrogen via nitrite with EBPR in the SBR upgraded the treated effluent quality allowing its reuse for agricultural purposes. Nitrogen and phosphorus uptake rates were higher in the processes removing nutrients via nitrite
- Configurations performing denitrification via nitrite allowed higher levels of fermented liquid recirculation in the UASB, resulting in higher biogas generation.

The environmental assessment of the alternative processes in the integrated systems showed:

- Climate change achieved the lowest results in Configuration 1-scNSD (467 kg CO_2 eq/FU) and the highest GHG emissions were produced in Configuration 2-ND-P (622 kg CO_2 eq/FU). The environmental impacts were mainly attributed to the energy requirements for FWD operation and SBR aeration. The use of FWDs increased GHG emissions by 57% - 67% compared to the separate collection, while denitrification via nitrate entailed 15% higher impacts in CC compared to nitrogen removal via nitrite.

- Impacts in eutrophication related categories derived from the discharge of the treated effluent. Thus, the collection scheme does not affect the environmental performance. The systems which perform nitrogen removal via nitrite and EBPR via nitrite resulted in better environmental profile concerning FE and ME.

Considering technical and environmental aspects, it can be concluded that the separate collection of waste combined with nitrogen removal and phosphorus uptake via nitrite is the best configuration for the combined treatment of wastewater and DOW in a small community of 2,000 PE. However, the collection efficiency highly influences the performance of the treatment scheme.

Acknowledgements

This research was supported by the EU projects: LIFE$^+$ LIVE-WASTE (LIFE 12 ENV/CY/000544) and PIONEER (PCIN-2015-227) and by the BBVA programme "2015 edition of the BBVA Foundation Grants for Researchers and Cultural Creators" (2015-PO027). L. Lijó would like to thank the COST Action ES1202 for a Short Term Scientific Mission grant. Dr. S. González-Garcia would like to express her gratitude to the Spanish Ministry of Economy and Competitivity for financial support (Grants references JCI-2012-11898 and RYC-2014-14984). The authors (L. Lijó, S. González-García and M.T. Moreira) belong to CRETUS (AGRUP2015/02) and the Galician Competitive Research Group GRC 2013-032.

5. References

Albertson, O., Burris, B., Reed, S., Semon, J., Smith, J.E., Wallace, A., 1991. Dewatering municipal wastewater sludges, Pollution Technology. New Jersey, USA.

Althaus, H.J., Hischier, R., Jungbluth, N., Osses, M., Primas, A., 2007. Life cycle inventories of Chemicals. Ecoinvent report N°8, v2.0 EMPA. Dübendorf, Swizerland.

Barros, P., Ruiz, I., Soto, M., 2008. Performance of an anaerobic digester-constructed wetland system for a small community. Ecol. Eng. 33, 142–149. doi:10.1016/j.ecoleng.2008.02.015

Battistoni, P., Fatone, F., Passacantando, D., Bolzonella, D., 2007. Application of food waste disposers and alternate cycles process in small-decentralized towns: a case study. Water Res. 41, 893–903. doi:10.1016/j.watres.2006.11.023

Battistoni, P., Pezzoli, S., Bolzonella, D., Pavan, P., 2002. The AF-BNR-SCP process as a way to reduce global sludge production: comparison with classical approaches on a full scale basis. Water Sci. Technol. 46, 89–96.

Bernstad, A., Davidsson, A., Tsai, J., Persson, E., Bissmont, M., la Cour Jansen, J., 2013. Tank-connected food waste disposer systems--current status and potential improvements. Waste Manag. 33, 193–203. doi:10.1016/j.wasman.2012.09.022

Boldrin, A., Andersen, J.K., Møller, J., Christensen, T.H., Favoino, E., 2009. Composting and compost utilization: accounting of greenhouse gases and global warming contributions. Waste Manag. Res. 27, 800–12. doi:10.1177/0734242X09345275

Bolzonella, D., Pavan, P., Battistoni, P., Cecchi, F., 2003. The under sink garbage grinder: a friendly technology for the environment. Environ. Technol. 24, 349–59. doi:10.1080/09593330309385567

Bruun, S., Hansen, T.L., Christensen, T.H., Magid, J., Jensen, L.S., 2006. Application of processed organic municipal solid waste on agricultural land – a scenario analysis. Environ. Model. Assess. 11, 251–265. doi:10.1007/s10666-005-9028-0

Ceglie, F.G., Bustamante, M.A., Ben Amara, M., Tittarelli, F., 2015. The Challenge of Peat Substitution in Organic Seedling Production: Optimization of Growing Media Formulation through Mixture Design and Response Surface Analysis. PLoS One 10, e0128600. doi:10.1371/journal.pone.0128600

Chan, S.Y., Tsang, Y.F., Chua, H., Sin, S.N., Cui, L.H., 2008. Performance study of vegetated sequencing batch coal slag bed treating domestic wastewater in suburban area. Bioresour. Technol. 99, 3774–3781. doi:10.1016/j.biortech.2007.07.018

Colón, J., Martínez-Blanco, J., Gabarell, X., Rieradevall, J., Font, X., Artola, A., Sánchez, A., 2009. Performance of an industrial biofilter from a composting plant in the removal of ammonia and VOCs after material replacement. J. Chem. Technol. Biotechnol. 84, 1111–1117. doi:10.1002/jctb.2139

Cookney, J., Cartmell, E., Jefferson, B., McAdam, E.J., 2012. Recovery of methane from anaerobic process effluent using poly-di-methyl-siloxane membrane contactors. Water Sci. Technol. 65, 604–610. doi:10.2166/wst.2012.897

Cookney, J., Mcleod, A., Mathioudakis, V., Ncube, P., Soares, A., Jefferson, B., McAdam, E.J., 2016. Dissolved methane recovery from anaerobic effluents using hollow fibre membrane contactors. J. Memb. Sci. 502, 141–150. doi:10.1016/j.memsci.2015.12.037

De Koning, J., 2003. Effects on wastewater treatment focused on additional production of biogas 1–10.

De Vries, J.W., Vinken, T.M.W.J., Hamelin, L., De Boer, I.J.M., 2012. Comparing environmental consequences of anaerobic mono- and co-digestion of pig manure to produce bio-energy--a life cycle perspective. Bioresour. Technol. 125, 239–48. doi:10.1016/j.biortech.2012.08.124

Doka, G., 2007. Life Cycle Inventories of Waste Treatment Services. Ecoinvent report N°13. Dübendorf, Swizerland.

Dones, R., Bauer, C., Bolliger, R., Burger, B., Faist-Enmenegger, M., Frischknecht, R., Heck, T., Jungbluth, N., Röder, A., Tuchschmid, M., 2007. Life cycle inventories of energy systems: results fro current systems in Switzerland and other UCTE countries. Ecoinvent report N°5. Dübendorf, Swizerland.

ecoinvent® database, 2016. http://www.ecoinvent.org/ [WWW Document].

EEC, 1991. Directive 91/271/EEC concerning urban waste-water treatment. Off. J. Eur. Communities 10.

Evans, T.D., Andersson, P., Wievegg, Å., Carlsson, I., 2010. Surahammar: a case study of the impacts of installing food waste disposers in 50% of households. Water Environ. J. 24, 309–319. doi:10.1111/j.1747-6593.2010.00238.x

Finnveden, G., Johansson, J., Lind, P., 2005. Life cycle assessment of energy from solid waste - part 1: general methodology and results. J. Clean. 13, 213–229. doi:10.1016/j.jclepro.2004.02.023

Frison, N., Di Fabio, S., Cavinato, C., Pavan, P., Fatone, F., 2013a. Best available carbon sources to enhance the via-nitrite biological nutrients removal from supernatants of anaerobic co-digestion. Chem. Eng. J. 215–216, 15–22. doi:10.1016/j.cej.2012.10.094

Frison, N., Katsou, E., Malamis, S., Bolzonella, D., Fatone, F., 2013b. Biological nutrients removal via nitrite from the supernatant of anaerobic co-digestion using a pilot-scale sequencing batch reactor operating under transient conditions. Chem. Eng. J. 230, 595–604. doi:10.1016/j.cej.2013.06.071

Galí, A., Dosta, J., Mata-Alvarez, J., 2007. Optimisation of Nitrification-Denitrification Process in a SBR for the Treatment of Reject Water Via Nitrite. Environ. Technol. 28, 565–571. doi:10.1080/09593332808618817

Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A. De, Struijs, J., Zelm, R. Van, 2009. ReCiPe 2008, A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level. University of Leiden, Radboud University Nijmegen, RIVM, Bilthoven, Amersfoort, Netherlands.

Hernandez, T., Masciandaro, G., Moreno, J.I., Garcia, C., 2006. Changes in organic matter composition during composting of two digested sewage sludges. Waste Manag 26, 1370–1376. doi:10.1016/j.wasman.2005.10.006

Hophmayer-Tokich, S., 2000. Wastewater Management Strategy: centralized v . decentralized technologies for small communities 27.

Iacovidou, E., Ohandja, D.-G., Gronow, J., Voulvoulis, N., 2012. The Household Use of Food Waste Disposal Units as a Waste Management Option: A Review. Crit. Rev. Environ. Sci. Technol. 42, 1485–1508. doi:10.1080/10643389.2011.556897

IPCC, 2006. Guidelines for National Greenhouse Gas Inventories - Chapter 11 N2O emissions from managed soils, and CO2 emissions from lime and urea application 1–54.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fith Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York, USA.

ISO 14040, 2006. Environmental Management-Life Cycle Assessment- Principles and Framework, Geneve, Switzerland.

Katsou, E., Malamis, S., Frison, N., Fatone, F., 2015. Coupling the treatment of low strength anaerobic effluent with fermented biowaste for nutrient removal via nitrite. J. Environ. Manage. 149, 108–17. doi:10.1016/j.jenvman.2014.09.008

Krampe, J., 2013. Cycle-time determination and process control of sequencing batch membrane bioreactors. Water Sci. Technol. 67, 2083–2090. doi:10.2166/wst.2013.096

Latif, M.A., Ghufran, R., Wahid, Z.A., Ahmad, A., 2011. Integrated application of upflow anaerobic sludge blanket reactor for the treatment of wastewaters. Water Res. 45, 4683–4699. doi:10.1016/j.watres.2011.05.049

Lee, W.S., Chua, A.S.M., Yeoh, H.K., Ngoh, G.C., 2014. A review of the production and applications of waste-derived volatile fatty acids. Chem. Eng. J. 235, 83–99. doi:10.1016/j.cej.2013.09.002

Liang, H., Gao, M., Liu, J., Wei, Y., Guo, X., 2010. A novel integrated step-feed biofilm process for the treatment of decentralized domestic wastewater in rural areas of China. J. Environ. Sci. 22, 321–327. doi:10.1016/S1001-0742(09)60111-X

Libralato, G., Volpi Ghirardini, A., Avezzù, F., 2012. To centralise or to decentralise: An overview of the most recent trends in wastewater treatment management. J. Environ. Manage. 94, 61–68. doi:10.1016/j.jenvman.2011.07.010

Malamis, S., Katsou, E., Di Fabio, S., Bolzonella, D., Fatone, F., 2013. Biological nutrients removal from the supernatant originating from the anaerobic digestion of the organic fraction of municipal solid waste. Crit. Rev. Biotechnol. 34, 244–57. doi:10.3109/07388551.2013.791246

Marashlian, N., El-Fadel, M., 2005. The effect of food waste disposers on municipal waste and wastewater management. Waste Manag. Res. 23, 20–31. doi:10.1177/0734242X05050078

Matsuura, N., Hatamoto, M., Sumino, H., Syutsubo, K., Yamaguchi, T., Ohashi, A., 2015. Recovery and biological oxidation of dissolved methane in effluent from UASB treatment of municipal sewage using a two-stage closed downflow hanging sponge system. J. Environ. Manage. 151, 200–209. doi:10.1016/j.jenvman.2014.12.026

Meneses, M., Pasqualino, J.C., Castells, F., 2010. Environmental assessment of urban wastewater reuse: Treatment alternatives and applications. Chemosphere 81, 266–272. doi:10.1016/j.chemosphere.2010.05.053

Molinos-Senante, M., Garrido-Baserba, M., Reif, R., Hernández-Sancho, F., Poch, M., 2012. Assessment of wastewater treatment plant design for small communities: Environmental and economic aspects. Sci. Total Environ. 427–428, 11–18. doi:10.1016/j.scitotenv.2012.04.023

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. J. Clean. Prod. 32, 157–172. doi:10.1016/j.jclepro.2012.03.026

Nogueira, R., Brito, a. G., Machado, a. P., Janknecht, P., Salas, J.J., Vera, L., Martel, G., 2009. Economic and environmental assessment of small and decentralized wastewater treatment systems. Desalin. Water Treat. 4, 16–21. doi:10.5004/dwt.2009.349

Norton-Brandão, D., Scherrenberg, S.M., van Lier, J.B., 2013. Reclamation of used urban waters for irrigation purposes – A review of treatment technologies. J. Environ. Manage. 122, 85–98. doi:10.1016/j.jenvman.2013.03.012

Peng, Y.Z., Wu, C.Y., Wang, R.D., Li, X.L., 2011. Denitrifying phosphorus removal with nitrite by a real-time step feed sequencing batch reactor. J. Chem. Technol. Biotechnol. 86, 541–546. doi:10.1002/jctb.2548

Poeschl, M., Ward, S., Owende, P., 2012. Environmental impacts of biogas deployment – Part I: life cycle inventory for evaluation of production process emissions to air. J. Clean. Prod. 24, 168–183. doi:10.1016/j.jclepro.2011.10.039

Righi, S., Oliviero, L., Pedrini, M., Buscaroli, A., Della Casa, C., 2013. Life Cycle Assessment of management systems for sewage sludge and food waste: centralized and decentralized approaches. J. Clean. Prod. 44, 8–17. doi:10.1016/j.jclepro.2012.12.004

Rihani, M., Malamis, D., Bihaoui, B., Etahiri, S., Loizidou, M., Assobhei, O., 2010. In-vessel treatment of urban primary sludge by aerobic composting. Bioresour. Technol. 101, 5988–5995. doi:10.1016/j.biortech.2010.03.007

Rodriguez-Garcia, G., Molinos-Senante, M., Hospido, a, Hernández-Sancho, F., Moreira, M.T., Feijoo, G., 2011. Environmental and economic profile of six typologies of wastewater treatment plants. Water Res. 45, 5997–6010. doi:10.1016/j.watres.2011.08.053

Rosenwinkel, K.H., Wendler, D., 2001. Influences on the anaerobic sludge treatment by co-digestion of organic wastes. Proc.of Sludge Manag. Enter. 3 rd Millenn. Int. Water Assoc. Spec. Conf. 25–28.

Russo, G., De Lucia, B., Vecchietti, L., Rea, E., Leone, A., 2011. Environmental and agronomical analysis of different compost-based peat-free substrates in potted rosemary. Acta Hortic. 891, 265–272.

Saer, A., Lansing, S., Davitt, N.H., Graves, R.E., 2013. Life cycle assessment of a food waste composting system: environmental impact hotspots. J. Clean. Prod. 52, 234–244. doi:10.1016/j.jclepro.2013.03.022

Souza, C.L., Chernicharo, C.A.L., Aquino, S.F., 2011. Quantification of dissolved methane in UASB reactors treating domestic wastewater under different operating conditions. Water Sci. Technol. 64, 2259–2264. doi:10.2166/wst.2011.695

Spielmann, M., Bauer, C., Dones, E., Tuchschmid, M., 2007. Life cycle inventories of transport services. Dübendorf, Swizerland.

Tchobanoglous, G., Burton, F.L., Stensel, H.D., 2014. Wastewater Engineering: Treatment and Resource Recovery, 5th editio. ed. McGraw-Hill Science, New York.

Terna Rete Italia, 2015. Dati statistici sull'energia elettrica in Italia - 2014. doi:10.1017/CBO9781107415324.004

Traverso, P., Pavan, P., Innocenti, L., Bolzonella, D., Mata-Alvarez, J., Cecchi, F., 2000. Anaerobic fermentation of source separated mixtures of vegetables and fruits wasted by supermarkets, in: Symp. On Environmental Biotechnology. Noordwijkerhout, The Netherlands.

Tremier, A., De Guardia, A., Massiani, C., Paul, E., Martel, J.L., 2005. A respirometric method for characterising the organic composition and biodegradation kinetics and the temperature influence on the biodegradation kinetics, for a mixture of sludge and bulking agent to be co-composted. Bioresour. Technol. 96, 169–80. doi:10.1016/j.biortech.2004.05.005

Wei, Y., Van Houten, R.T., Borger, A.R., Eikelboom, D.H., Fan, Y., 2003. Minimization of excess sludge production for biological wastewater treatment. Water Res. 37, 4453–4467. doi:10.1016/S0043-1354(03)00441-X

Weichgrebe, D., Urban, I., Friedrich, K., 2008. Energy- and CO2-reduction potentials by anaerobic treatment of wastewater and organic kitchen wastes in consideration of different climatic conditions. Water Sci. Technol. 58, 379–384. doi:10.2166/wst.2008.363

Wu, S., Austin, D., Liu, L., Dong, R., 2011. Performance of integrated household constructed wetland for domestic wastewater treatment in rural areas. Ecol. Eng. 37, 948–954. doi:10.1016/j.ecoleng.2011.02.002

Ye, F., Li, Y., 2009. Enhancement of nitrogen removal in towery hybrid constructed wetland to treat domestic wastewater for small rural communities. Ecol. Eng. 35, 1043–1050. doi:10.1016/j.ecoleng.2009.03.009

Yildirin, M., Topkaya, B., 2012. Assessing Environmental Impacts of Wastewater Treatment Alternatives for Small-Scale Communities. CLEAN - Soil, Air, Water 40, 171–178. doi:10.1002/clen.201000423

Zeeman, G., Kujawa, K., de Mes, T., Hernandez, L., de Graaff, M., Abu-Ghunmi, L., Mels, A., Meulman, B., Temmink, H., Buisman, C., van Lier, J., Lettinga, G., 2008. Anaerobic treatment as a core technology for energy, nutrients and water recovery from source-separated domestic waste(water). Water Sci. Technol. 57, 1207–1212. doi:10.2166/wst.2008.101