Economic analysis and life cycle approach to compare drying reed beds and conventional treatments for sludge management

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Abstract
Sludge treatment wetlands (STW) emerge as a promising sustainable technology with low energy requirements and operational costs. In this work economic and environmental aspects of STW are investigated to compare alternatives for sludge management. To this end, cost analysis and life cycle assessment (LCA) were carried out considering dimensions and operation criteria of full-scale systems located in Spain. Four scenarios are considered: 1) STW with direct land application, 2) STW with compost post-treatment, 3) centrifuge with compost post-treatment and 4) sludge transport to an intensive wastewater treatment plant. The economic analysis shows that in small facilities (500-2000 PE), constructed wetlands with direct land application is the most favorable solution (less than 0.15 €/m^3 of water treated). The costs are slightly increased if post-treatment is required (between 0.18 and 0.15 €/m^3). On the other hand, centrifugation costs decrease at increasing wastewater flow rates, as a result of high implementation costs (from 0.28 to 0.15 €/m^3). According to the SimaPro LCA, STW with direct land application correspond to the solution with lower environmental impact. In all scenarios global warming is a significant impact category, which is attributed to fossil fuel and electricity consumption; while greenhouse gas emissions from STW are insignificant. On the whole, STW is the most appropriate solution to manage waste sludge produced in decentralized and small communities (<2000 PE), mainly post-treatment is not required prior to land application.

Keywords
Constructed wetlands; biosolids; composting; drying reed beds; sanitation, wastewater

INTRODUCTION

A major concern of intensive sewage treatment processes is the large production of waste sludge, which is generally managed by complex and costly operations. Its production is highly variable depending on the wastewater treatment used, for instance conventional activated sludge processes produce from 60 to 80 g of total solids (TS) per person per day (Von Sperling and Gonçalves, 2007). During the last years, sludge generation has increased dramatically by the fast growth of world population and industrialisation (Hong et al., 2009). According to Fytili and Zabanitou (2008), sludge production has increased in the European Union by 50 % since 2005. Therefore, optimisation of sludge management becomes a key element in the wastewater treatment sector.

Conventional sludge stabilisation and dewatering technologies (i.e. anaerobic digestion followed by centrifugation or filtration) are costly and energy demanding, which is troublesome particularly in small facilities (<2,000 population equivalent (PE)). This is a matter of concern, since the number of small wastewater treatment plants (WWTP) in operation will continue to increase within the next years, including municipalities below 500 PE (Council of the European Union, 2000). Nowadays, the solution adopted in many small facilities is sludge transport to the nearest WWTP with a conventional sludge treatment line, posing high operation costs and high potential environmental impacts. In this context, simplified *in situ* treatments are needed.

Sludge treatment wetlands (STW) consist of shallow tanks filled with a gravel layer and planted with emergent rooted wetland plants such as *Phragmites australis* (common reed). Sludge is spread and stored on the surface of the beds where most of its water content is lost by evapotranspiration of the plants and by water draining through the gravel filter layer, leaving a concentrated sludge residue on the surface. When the maximum storage capacity is reached, after a final resting period, the final biosolids are withdrawn to start a new operating cycle. Evolution of sludge composition results from dewatering and mineralisation processes (Nielsen, 2003). The resulting final product is suitable for land application (Nielsen and Willoughby, 2005); although in the practice in some cases it is post-treated to improve sludge stabilisation and hygienisation (Zwara and Obarska-Pempkowiak, 2000).

In comparison with common mechanical dewatering technologies like centrifuges, sludge treatment wetlands emerge as a promising alternative (Uggetti et al., 2010), which has low energy requirements, reduced operation and maintenance costs, and in principle causes little environmental impact. However, a systematic evaluation of the environmental performance of this technology has not yet been reported.

In this study, STW costs and environmental impact are investigated and compared to conventional treatments for sludge management in small communities (<2,000 PE). Economic and environmental assessments have been carried out assuming design and operation criteria of full-scale systems located in Spain. Four scenarios are considered and compared: 1) STW with direct land application of the final product, 2) STW with compost post-treatment, 3) centrifugation with compost post-treatment, 4) sludge transport to an intensive WWPT without previous treatment. Our aim was to demonstrate the suitability of STW for small communities, not only in terms of process performance but also in terms of costs and environmental impacts.

MATERIALS AND METHODS

Economic assessment

Economic aspects of STW are compared with sludge management alternatives which are currently used in small WWTP in our zone: centrifugation, as representative of mechanical dewatering techniques, and transport to a larger WWTP with sludge treatment line. According to the common practice adopted nowadays in Spain, STW followed by composting is also considered (scenario 2). Each scenario is evaluated for sewage treatment capacities of 100, 200 and 400 m^3/d of wastewater treated, theoretically corresponding to 500; 1,000 and 2,000 PE.

According to design and operation criteria of STW located in Spain (Uggetti et al., 2009), we considered between 4 and 12 beds, with an average surface of 50 m^2 and height of 1.6 m. Taking into account the 20 cm layer of gravel and sand, the sludge storage capacity results in 50 m^3. In this study, sludge loading rate of 50 kg TS/m^2·year and 5 year operating cycles are assumed, although longer operating cycles are reported in other countries like Denmark (Nielsen, 2003). Emptying procedures involve final biosolids withdrawal with a power shovel and transport to final destination. STW operation is thereafter re-started without replanting.

Table 1 summarises sludge flow rates for each scenario. Secondary sludge generation in the WWTP is calculated by the Huisken equation. The difference between sludge production in STW and centrifuge is due to the TS concentration of the final product, 25 and 20 % TS, respectively (Uggetti et al., 2010). CH_4 emissions were measured by gas chromatography (Thermo Finnigan Trace, GC 2000) in samples collected from representative STW by positioning a Linvall Hood of 1 m^2 surface area as described by Sarkar and Hobbs (2002).

Table 1. Sludge flow rates considered in the economic and environmental assessment.

	Wastewater treated		
	100 m^3/d	**200 m^3/d**	**400 m^3/d**
Waste activated sludge (sludge generation) (m^3/year) (all scenarios)	275	550	1100
Sludge production in STW (m^3/year) (scenarios 1 and 2)	33	66	132
Sludge production in centrifuge (m^3/year) (scenario 3)	41	82	165
Pump electricity consumption in STW (kWh/year) (scenarios 1 and 2)	25	50	105
Pump electricity consumption in centrifuge (kWh/year) (scenario 3)	30	60	125
Centrifuge electricity consumption (kWh/year) (scenario 3)	140	280	560
CH_4 emission rate from STW ($mg/m^2 \cdot s$) (scenarios 1 and 2)	< 88	< 88	< 88

Life cycle assessment

The aim of the LCA model developed is to compare the environmental impact of STW with sludge management alternatives commonly used in small WWTP in our zone. Therefore, the same scenarios as in the economic analysis are considered.

The function of the system is to manage secondary sludge produced in an activated sludge unit with extended aeration. For this reason, the functional unit is defined as the management of 1 ton of sewage sludge (wet weight).

Since the study is focused on sludge management, secondary sludge is selected as input material; and only the impact generated by sludge management in the facility is accounted for. This includes the sludge treatment line of the WWTP (STW or centrifuge) and transport to post-treatment in a composting plant (scenarios 2 and 3) or treatment in an intensive WWTP (scenario 4), assuming a distance of 30 km in all cases. Treatments outside the WWTP (composting in scenarios 2 and 3; and sludge treatment in a larger WWTP in scenario 4) are not included in the model.

Raw materials required for systems' construction and energy consumption for systems' operation are taken into account. On the contrary, the systems' boundaries exclude the construction phase, which only accounts for minor environmental impacts compared to the operation phase of WWTP, according to previous LCA studies (Lundie et al., 2004 and Lassaux et al., 2007). The end of life is included for the centrifuge, as it should be replaced over the period considered (20 years); but not for STW, since their lifespan is longer than the 20 years period considered in this study.

Inventory data on systems' design and operation are the same as for the economic analysis. Data concerning the embodied environmental aspects of materials, transport use and other processes were taken from the Ecoinvent system process database. The LCA analysis was carried out with the software SimaPro 7.1 by PRé Consultant, using the CML 2 baseline method (Guinée, 2001). Impact categories evaluated include Abiotic Resource Depletion, Acidification, Eutrophication and Global Warming Potential (Climate Change), amongst others.

RESULTS

Economic assessment

Table 2 shows investment and operation costs of scenarios 1, 2 and 3 (scenario 4 does not have investment costs, hence it is not included). The results are expressed in €/m^3 of wastewater treated. STW investment costs include soil occupation and excavation, wetlands construction, pump and pipe installation, gravel placement and plantation. The most significant costs of the centrifuge include machine assembly and installation, room construction and polyelectrolyte preparation. Notice that STW investment costs increase with the treatment capacity, from 60,000 to 190,000 € for 500 and 2,000 PE systems, respectively. On the other hand, centrifuge costs increase only slightly, from 90,000 to 120,000 €. Therefore, the difference between investment costs becomes more evident for 2,000 PE facilities and are more competitive for centrifuges.

Table 2. Investment and operation costs of scenarios 1, 2, 3 and 4: (1) STW, (2) STW + compost, (3) centrifuge + compost and (4) transport to WWTP. Costs are expressed in €/year.

		Wastewater treated		
		100 m^3/d	**200 m^3/d**	**400 m^3/d**
Scenario 1	Investment cost	60.169	99.490	189.736
	Operation cost	4.012	7.259	13.199
Scenario 2	Investment cost	60.169	99.490	189.736
	Operation cost	5.007	9.237	17.161
Scenario 3	Investment cost	88.722	90.448	114.950
	Operation cost	7.952	13.564	20.939
Scenario 4	Investment cost	-	-	-
	Operation cost	11.348	21.018	38.430

The economic analysis considering a life cycle of 20 years is shown in Fig. 1. It is calculated assuming 3% increase of operation costs and applying 5% interest tax to the total cost. In this case, amortisation of investment and STW emptying costs are also included. From a long term perspective, the benefit of biosolids' direct land application (scenario 1) emerges versus compost post-treatment (scenario 2), with lower costs (0.021 €/m^3) in all cases. Investment and operation costs of the centrifuge (0.28 €/m^3) are more expensive than other solutions (0.24 €/m^3 for transport and 0.16-0.18 €/m^3 for STW) for communities of 500 PE. However, centrifugation costs decrease at increasing treatment capacity (to 0.20 and 0.15 €/m^3 for 1,000 and 2,000 PE systems, respectively), hence treatment costs are the same as STW for 2,000 PE systems. Transport may be considered as an alternative to centrifugation only for systems with less than 850 PE (0.28 €/m^3 versus 0.24 €/m^3). Likewise, STW costs are 0.05-0.07 €/m^3 lower than this option. It is worth mentioning that the economic evaluation of this scenario is correlated with sludge production (and humidity), as well as the distance to nearest WWTP with sludge treatment line. In this study, an average distance of 30 km was adopted, based on circumstances normally observed in our zone.

This analysis underlines the economic advantage of STW with respect to conventional treatments exemplified by centrifugation in facilities up to 2,000 PE. However, this technology is currently adopted for sludge management in systems up to 30,000 PE in Italy (Peruzzi *et al.*, 2007) and 60,000-125,000 PE in Denmark (Nielsen, 2003). Certainly, the results are specific for each country, depending on the costs (i.e. electricity), as well as design and operation criteria of STW and weather conditions, affecting the efficiency of the treatment. For instance, operating cycles of 5 and 10 years are described in Spain and Denmark, respectively. Longer operating cycles reduce operation costs of STW, resulting in additional economic advantage for communities above 2,000 PE.

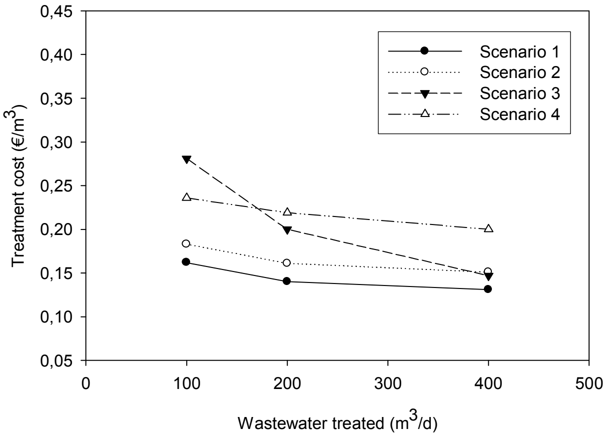

Figure 1. Investment and operation costs over 20 years of operation of scenarios 1, 2, 3 and 4: (1) STW, (2) STW + compost, (3) centrifuge + compost and (4) transport to WWTP.

Life cycle assessment

In LCA analysis the environmental impacts attributed to materials or processes are grouped according to the so-called impact categories. LCA results are therefore expressed as a quantification of the potential contribution of materials and processes to each impact category. Fig. 2 shows the main impact categories of this LCA model (Abiotic Resource Depletion, Acidification, Eutrophication and Global Warming Potential (Climate Change)), with comparative results for each scenario. The results are presented in absolute values in the units corresponding to each impact category. Within each impact category, the total impact as well as the individual contribution of raw materials, energy and transport are included separately. This interpretation is useful to determine the most influent element of the process that could eventually be modified to reduce the global impact.

In general, within each category the total impact is distributed following the same pattern: transport (scenario 4) has the highest impact, from 3 to 6 times higher than centrifuge with compost post-treatment (scenario 3) and STW with compost post-treatment (scenario 2). The impact of STW with direct use of the final product (scenario 1) is negligible in comparison with the other scenarios, with values between 1,000 and 6,000 times lower. According to this analysis, STW appear as the most favourable solution in every impact category. For this scenario 1 the biggest impact is caused by raw materials employed in system's construction; while direct greenhouse gas emissions (Table 2), as well as indirect emissions derived from energy consumption and transport, have a smaller contribution. On the whole, STW impact is negligible in comparison with the rest. If post-treatment is required, the total impact of STW (scenario 2) and centrifuge (scenario 3) is similar, due to sludge transport to post-treatment. From an environmental point of view, centrifuges and filter bands do not have relevant differences (Gallego *et al.*, 2008), therefore scenario 3 should be representative of conventional mechanical dewatering treatments.

Global Warming Potential accounts for a high contribution mainly in scenarios 2, 3 and 4 (1,100; 1,300 and 6,000 kg CO_2eq/t wet weight, respectively) due to fossil fuel and electricity consumption. In STW, the contribution of CH_4 emissions to this impact category is negligible, as a result of the low CH_4 found in these type of systems (Table 1).

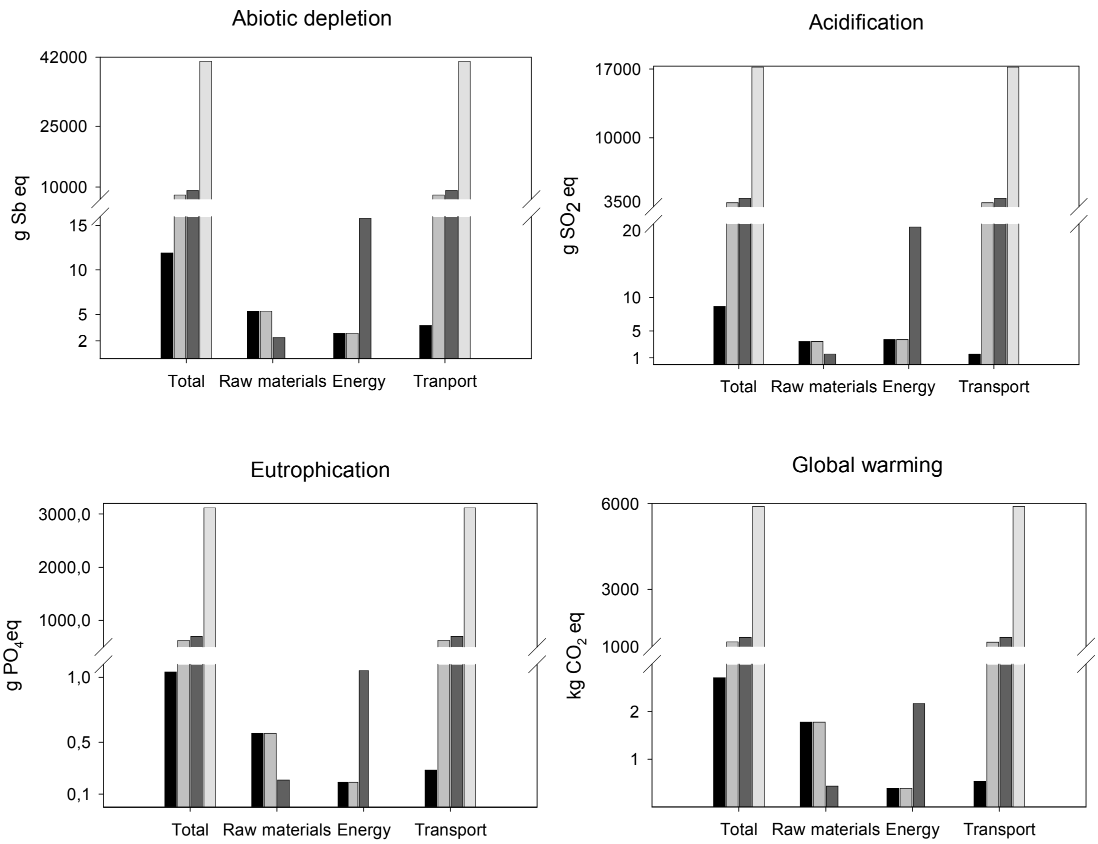

Figure 2. Life Cycle Assessment results grouped by impact categories for scenarios 1, 2, 3 and 4: (1) STW, (2) STW + compost, (3) centrifuge + compost and (4) transport to WWTP.

If we look at individual contributions of raw materials, energy and transport within each scenario (Fig. 2), other trends are observed. Scenario 1 is characterised by a high consumption of raw materials (basically steel and gravel), which accounts for the highest contribution in all impact categories. On the other hand, lower impacts are attributed to the energy consumption for sludge pumping into the STW, and transport during STW emptying operation.

Scenario 2 has the same contribution as scenario 1 with respect to raw materials and energy, but in this case transport accounts for the highest impact, which is attributed to the compost post-treatment. In scenario 3, the centrifuge has low raw materials requirements, but significantly higher energy consumption for sludge dewatering and pumping. Like in scenario 3, transport to compost post-treatment has the highest contribution to the total impact. As in the economic study, sludge transport to an intensive WWTP (scenario 4) is characterised by the highest environmental impact in all categories. Indeed, the reduction of sludge volume after dewatering (scenarios 1-3) has a positive environmental impact with respect to untreated sludge transport.

The results of this assessment show the economic and environmental benefits of STW compared to conventional mechanical dewatering and transport of untreated sludge. STW are less advantageous if compost post-treatment is required, as with mechanical dewatering techniques, due to the impact associated to sludge transport. However, the impacts of composting may differ between partially stabilised sludge from STW and dewatered sludge from centrifuges. For this reason, further LCA studies should include the post-treatment stage as well as final disposal of biosolids. As indicated by Cambell (2000), the most important criterion in the selection of sludge management alternatives is that the solution must be appropriate to the local conditions of each site.

CONCLUSIONS

This study looked at economic and environmental aspects of sludge treatment wetlands for small communities (500-2,000 PE). From this evaluation, STW with direct land application emerge as the most cost-effective scenario, which is also characterised by the lowest environmental impact (almost negligible in comparison with the other options evaluated). The LCA highlights that in all scenarios global warming has a significant impact, which is attributed to fossil fuel and electricity consumption; while methane emissions from STW are insignificant. As a conclusion, sludge treatment in constructed wetlands with direct land application is the most appropriate solution to manage waste sludge in decentralised small communities.

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REFERENCES

Council of the European Union (1986). Council Directive 86/278/EEC of 12 June 1986 on the Protection of the Environment, and in Particular of the Soil, when Sewage Sludge is Used in Agriculture. *Official Journal of the European Union* L 181, 04/07/1986, 6-12.

Fytili, D. and Zabaniotou A. (2008). Utilization of sludge in EU application of old and new methods – A review. *Renewable & Sustainable Energy Reviews* 12, 116-140.

Gallego, A., Hospido, A., Moreira, M.T., Feijoo, G. (2008). Environmental performance of wastewater treatment plants for small populations. *Resources conservation and recycling* **52**, 931-940.

Guinée, J. (2001). Life Cycle Assessment: an operational guide to ISO standards. Centre of Environmental Science, Leiden University, Kluwer Academic Publisher, 2001.

Lassaux, S., Renzoni, R., Germanin, A. (2007). Life cycle assessment of water: from the pumping station to the wastewater treatment plant. *International Journal of LCA* **12** (2), 118-126.

Lundie, S., Peter, G.M., Beavis, P.C. (2004). Life Cycle Assessment of wastewater systems: Influence of system boundaries and scale on calculated environmental loads. *Environmental Science and Technology* **34** (1), 180-186.

Hong, J., Hong, J., Otaki, M., Jolliet, O. (2009). Environmental and economic life cycle assessment for sewage sludge treatment processes in Japan. *Waste Management* **29** (2), 696-703.

Nielsen, S. (2003). Sludge treatment in wetland systems. Proceedings of Conference: The Use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands (IWA). 8-10 May. Dias V and Vymazal J. (ed). Lisbon (Portugal).

Nielsen, S. and Willoughby, N. (2005). Sludge treatment and drying reed beds systems in Denmark. Water and Environmental Journal **19** (4), 296-305.

Peruzzi, E., Macci, C., Doni, S., Masciandaro, G., Peruzzi, P., Aiello, M., Caccianti, B. (2007). Wetland cane *(Phragmites australis)* for sewage sludge stabilization. In: *Proceedings of the International Conference on Multi Functions of Wetland Systems,* Padova, Italy. Borin, M. And Bacelle, S. (Eds).

Sarkar, U., Hobbs, S. E. (2003). Landfill odour: assessment of emissions by the flux footprint method. Environmental Modelling & software 18 (2), 155-163.

Uggetti, E., Llorens, E., Pedescoll, A., Ferrer, I., Castellnou, R., García, J. (2009) Sludge dewatering and stabilization in drying reed beds: characterization of three full-scale systems in Catalonia, Spain. Bioresource Technology, 100(17), 3882-3890

Uggetti, E., Ferrer, I., E., Llorens, García, J. (2010). Sludge treated wetlands: A review on the state of the art. *Bioresource Technology* **101** (9), 2905-2912.

Von Sperling, M., Gonçalves, R.F. (2007). Sludge characteristics and production. In: *Sludge Treatment and Disposal*. Cleverson, Von Sperling & Fernandes Eds. IWA Publishing, London, UK, 2007.

Zwara, W. and Obarska-Pempkowiak, H. (2000). Polish experience with sewage sludge utilization in reed beds. Water Science and Technology **41** (1), 65-69