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PhD Thesis

SEWAGE SLUDGE TREATMENT IN CONSTRUCTED WETLANDS

Technical, economic and environmental aspects applied to small communities of the Mediterranean Region

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Preface

This work is part of two research projects on the assessment of process performance, design and operation criteria of Sludge Treatment Wetlands, financed by the Spanish Ministry of Environment (MMARM, Projects A335/2007 and 087/PC08) and by the Catalan Water Agency (ACA). Both projects focused on the evaluation of different Sludge Treatment Wetlands (STW) configurations' performances in term of sludge dewatering, mineralisation and hygienisation. This work rises from the lack of knowledge of this technique, mainly in the Mediterranean Region where until now little experience has been developed.

During the last years, the construction and operation of new Wastewater Treatment Plants (WWTP) has led to a significant increase of sludge production. As a response, many efforts have been done in order to find a sludge treatment able to provide a final product suitable for land application (fulfilling legislation requirements). In fact, sludge valorisation in agriculture is the preferred option nowadays, ensuring the return of organic constituents, nutrients and microelements to crop fields. Moreover, reasonable investment as well as operational and maintenance costs are important aspect to take into account especially in small communities with less than 2,000 Population Equivalent (PE). In this sense, sludge treatment wetlands (STW) are regarded as a suitable technology for sludge management from both, an economic and environmental point of view.

The main objective of this research work was to assess the suitability of STW for sludge treatment and management; with special focus on small WWTP (communities <2,000 PE) of the Mediterranean Region. For this purpose, technical, environmental and economic aspects of the treatment were investigated. A comparison with conventional treatment for sludge management is presented in order to establish the most favourable solution for the Catalan context. As the final result from this work, design and operations criteria are exposed as a guide for STW implementation in small Mediterranean communities.

The research work was basically conducted on four full-scale systems located in Catalonia (Nord-est of Spain). Moreover one sampling campaign was carried out in two Danish systems in order to compare treatment performances. Additionally, during the development of the research project 087/PC08, a pilot plant was built and implemented at the Technical University of Catalonia and experiments were carried out on this plant during more than two years. Chapters are here presented in their current order, which differs from the chronological order in which experiments were performed.

Initially, an extensive review on the state of art of the technology is presented in order to investigate the systems' functioning and performance (Chapter 3). The main characteristics and operational aspects of the technology are here described, including a summary of the main results reported in the literature. Furthermore, STW's efficiency is compared to conventional treatments.

STW's performance in terms of sludge dewatering, mineralization and hygienisation was investigated in three full-scale systems (Chapter 4) and in a pilot plant with different configurations (Chapter 5). These chapters are aimed at evaluating STW efficiency in the Mediterranean Region. To this end physic-chemical parameters are analysed in different campaigns along the seasons of the year.

Dewatering data collected from the pilot plant and from the full-scale systems were used to establish a dewatering model in STW (Chapter 6). Terzaghi's consolidation equation was implemented with plant evapotranspiration in order to predict water loss from the sludge layer within the wetlands. This model could be a useful tool for the determination of the most effective feeding frequency in order to enhance sludge dewatering. This will result in the reduction of the sludge layer increasing rate, thus emptying procedures and operation costs.

In Chapter 7 the properties of the biosolids from STW are assessed as organic fertiliser. To this aim three full-scale systems located in Spain and Denmark were sampled and analysed with special focus on the final product stabilisation, nutrient concentration and hygienisation, as proposed by the legislation currently in force.

Chapter 8 and 9 are devoted to the environmental and economic aspects. In Chapter 8, sampling and analysing techniques used to determine greenhouse gases emissions from croplands and natural wetlands were adapted to quantify spatial and temporal evolution of methane and nitrous oxide emissions from a STW. Data collected were used for the determination of the Global Warming Potential of the treatment.

On the other hand, Chapter 9 presents technical, economic and environmental assessments comparing STW with other alternatives for sludge management in small communities. Here, the performance of the full-scale STW were characterised during two years. While for the economic and environmental assessments, four scenarios were considered for the comparison of STW with centrifuge and sludge transport without previous treatments.

In Chapter 10 a general discussion of the results is presented. Thus, the whole discussion focuses on the most relevant aspects emerged from the work developed. This chapter aims at giving a general overview on the STW's processes. The discussion is centralised on the development of design and operation criteria resulted from the experience acquired along this work.

Finally, Chapter 11 presents the conclusions drawn from this thesis.

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1. Introduction

Sludge production and management

Wastewater treatment consists in a number of processes aimed at removing pollutants from water. These processes result in a liquid or semisolid by-product commonly referred as sewage sludge. The large amount of sludge produced and the high organic matter concentration make sludge management a major concern in wastewater treatment operation.

Sludge production and composition depends on the influent's characteristics and the wastewater treatment type. Sludge production in conventional activated sludge processes ranges from 60 to 80 g of total solids per person per day (Von Sperling and Gonçalves, 2007). In Europe from the implementation of the Urban Wastewater Treatment Directive 91/271/EEC (Council of the European Union, 1991) a growing number of WWTP with secondary treatment have been constructed and operated specially in municipalities above 2,000 PE. The Water Framework Directive (Council of the European Union, 2000) encouraged wastewater treatment even in municipalities below 500 PE. As a result, sludge production has increased in the European Union by 50% since 2005 (Fytili and Zabaniotou, 2008).

In order to manage the increasing amount of sludge produced in Spain, the following hierarchy was proposed (Consejo de Ministros, 2001): 1) valorisation in agriculture, 2) valorisation in energy generation, and 3) landfilling. Following this hierarchy, by the end of 2011 at least 70% of the sludge produced is expected to be valorised in agriculture, while energetic valorisation or landfill will be employed for less than 15% of sludge produced in Spain. It becomes clear that agricultural valorisation is nowadays preferred to landfilling, since sludge recycling ensures the return of organic constituents, nutrients and microelements to crop fields, which eases the substitution of chemical fertilizers (Oleszkiewicz and Mavinic, 2002). Sludge disposal onto agricultural land is regulated by the European Sludge Directive (Council of the European Union, 1986), which controls land application of sewage sludge according to heavy metals concentrations. Recent regulation proposals are more restrictive in terms of heavy metals, and also consider micropollutants and microbial faecal indicators (Environment DG, EU, 2000).

Sewage sludge composition

Primary sludge is defined in a conventional WWTP as the product of the primary settling tank and is characterised by high total solid (TS) content (3.0-7.0% TS) and high organic matter concentration (60-80% volatile solids VS/TS) (Wang et al., 2008). Primary sludge requires further treatments due to its high instability. Sludge from biological reactors is commonly known as secondary sludge and is generally characterized by low dry solids content (0.5-2.0% TS) and by being partially stabilised (50-60% VS/TS) (Wang et al., 2008).

The chemical composition of sludge is determined by the amount of total solids, volatile solids, grease and fats, proteins and nutrients (nitrogen, phosphorus and potassium). Table 1.1 presents a typical composition of primary and secondary sludge.

Table 1.1 Characteristics of primary and secondary sludge (Wang et al., 2008).

Parameter	Primary sludge	Secondary sludge
TS (%)	3.0-7.0	0.5-2.0
VS (% TS)	60-80	50-60
Nitrogen (N, % TS)	1.5-4.0	2.4-5.0
Phosphorus (P_2O_5,% TS)	0.8-2.8	0.5-0.7
Potassium (K_2O,%TS)	0-1.0	0.5-0.7
pH	5.0-8.0	6.5-8.0

Solids characteristics that affect the suitability for land application include organic content, nutrients, pathogens, metals and microcontaminants. Although several organic and mineral constituents in the sludge may have fertilising characteristics, others, as metals, trace organic contaminants and pathogens, are associated to sanitary and environmental risks.

With respect to nutrients, a relative high concentration of nitrogen (2.4-5.0% TS) is quite common in secondary sludge. However, the mineral and organic nitrogen chemical forms are strictly dependent on the wastewater origin and treatment. Thus, in considering fertilising properties, only the nitrogen mineral fraction should be taken into account, due to its readily availability form for crops. Others macronutrients, such as phosphorus and potassium, are usually found in mineral form. The major proportion of phosphorus expressed as P_2O_5 ranges from 0.8-2.8 % TS in primary sludge to 0.5 to 0.7% TS in secondary sludge, while potassium concentrations (K_2O) range from 0-1.0 and 0.5-0.7 in primary and secondary sludge, respectively.

Heavy metals are currently the only parameter limited by the legislation, due to their potential toxic effects. Heavy metals concentration in sludge is highly variable, depending on the wastewater source and treatment. Concentrations are generally lower in domestic wastewater than in many industrial watewaters. Trace organics (toluene, phenol, naphthalene among others) are generally present in industrial effluents and are receiving major attention as potential pollutants of soil, plants and water as a consequence of land application of sludge.

On the other hand, the pathogenic organisms may come from human or animal sources. Epidemiological surveys demonstrated that bacteria, viruses, helminth eggs and protozoan cysts contained in wastewaters pose risks to human and animal health. Pathogens are not currently limited by the legislation, though the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000) proposes limits values for *Salmonella* spp. (absence in 50 g) and *E.coli* (6 $\log_{10}$ reduction to less than $5 \cdot 10^2$ CFU/g).

Overview on sludge treatments

Main processes used for sludge handling may be grouped into stabilisation and dewatering treatments. Stabilisation treatments aim at reducing the biodegradable fraction of organic matter, thus reduce the risk of putrefaction, as well as diminishing the concentration of pathogens. On the other hand, dewatering techniques are used to decrease sludge volume, hence sludge disposal costs and environmental risks associated.

The most commonly used process flow for sludge treatment include: 1) preliminary operations as storage, grinding, screening; 2) thickening; 3) stabilization; 4) conditioning; and 5) dewatering (Figure 1.1)

Preliminary operations

Preliminary operations are necessary to remove plastics or other materials and to provide a relatively constant and homogeneous feed to subsequent process facilities. They are often required.

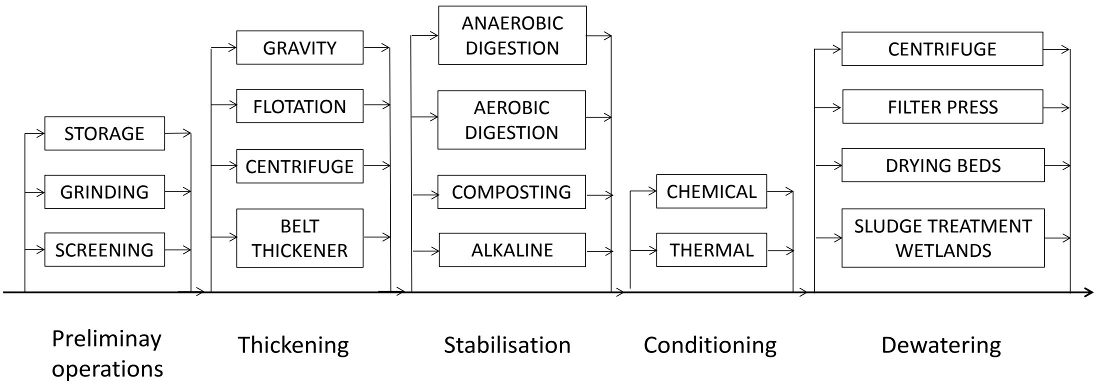


Figure 1.1 General process flow for sludge treatment (adapted from Metcalf and Eddy, 2003).

Sludge thickening

Thickening is a procedure used to increase the solid content of sludge by removing a portion of the liquid fraction. Sludge volume is usually reduced in this phase of the treatment by 2-3% of the original volume. Thickening can be achieved in circular sedimentation tanks by gravity or by flotation, performed naturally of by air injection. Moreover, centrifuge or belt thickeners are often sometimes used in medium and large facilities with good results.

Sludge stabilisation

Stabilization processes are aimed at inhibiting or reducing the potential of putrefaction and eliminating offensive odours as well as reducing pathogens. Those processes act on the volatile or organic fraction of the solids, improve sludge dewaterability and produce methane as source of energy. Anaerobic and aerobic digestion, composting and alkaline stabilization are the principal methods employed for sludge stabilisation in plants ranging in size, from small to very large.

Anaerobic and aerobic digestions consist in the biological degradation of organic matter by means of anaerobic or aerobic microorganisms. In anaerobic digestions, organic materials are biodegraded through a complex anaerobic microbiological process leading to the production of a more stabilised organic material and biogas with high methane material, which can be used for the generation of heat or energy. On the other hand, in aerobic digestion, aerobic microbes are the responsible of organic matter degradation in an open air reactor. The stabilisation process in this case is normally faster than in anaerobic condition, but with higher energy requirements. Digestion can lead to a final product with volatile solids concentration around 50-60%.

Composting can be used as a stabilisation process itself or following anaerobic or aerobic digestion, in order to improve biosolids quality. In this process, dewatered sludge is mixed with some organic support, such as wood shavings or sawdust, with the aim of enhancing organic material decomposition to a stable end product suitable for land application. Approximately 20 to 30% of the volatile material is converted to carbon dioxide and water. During the composting process, pathogens are removed due to the high temperature reached (up to 50-70°C).

Alkaline stabilisation rends the sludge unsuitable for the survival of microorganisms. Usually lime is added to untreated sludge, increasing pH in order to halt or retard the microbial reactions. The process can also inactivate virus, bacteria, and other microorganisms present.

Sludge conditioning

Sludge is sometimes conditioned in order to improve sludge characteristics and enhance subsequent dewatering. It can be chemical or thermal. The first consist of the addition of coagulants such as iron chlorite, lime, aluminium sulphate and organics polymers, which lead to the coagulation of solids, with the corresponding desorption of water. On the other hand, thermal conditioning is achieved by heating the sludge during brief periods and under pressure, which results in solids coagulation, the rupture of the gel structure and the reduction of sludge affinity for water, together with sludge sterilisation.

Sludge dewatering

Sludge dewatering is mainly used to reduce the moisture content to ease sludge handling and reduce transportation costs. Several techniques are used as dewatering devices for removing moisture. Some of them rely on natural evaporation and percolation (drying beds or sludge treatment wetlands), while in others dewatering is mechanically assisted (centrifugation or thermal drying).

Drying beds consists of concrete tanks, usually rectangular. Drainage is enabled in these systems by a draining medium (around 0.50 m height) of sand or gravel. Stabilised sludge is spread onto the filter medium, where solids are retained while water content is reduced by percolation. Drainage pipes are located on the bottom of tanks, allowing water collection. Dry sludge is withdrawn when tanks are fulfilled and either disposed in a landfill or used as soil conditioner. Drying beds have simple operation and are typically used to dewater digested and settled sludge.

Sludge Treatment Wetlands (or drying reed beds) is a relative new technique which combines drying beds and constructed wetlands for wastewater treatment. In these systems a filter layer allows water percolation, while the presence of emergent aquatic plants enhances water evapotranspiration and sludge mineralisation. The fundaments of this treatment are explained in detail in Chapter 2.

Centrifugation is widely used to force solid/liquid separation by centrifugal force. Here, a fast settling stage of solid particles is followed by a compaction under the prolonged action of centrifugation. Biosolids that outcome from the treatment are in the form of cake with 14 to 40% TS, depending on the source of sludge (Gonçalves et al., 2007). This product can be landfilled or post-treated before reuse. The main inconveniences of this treatment are high energy requirement and maintenance costs.

Thermal drying is based on the evaporative removal of interstitial water and is able to remove up to 98% of the sludge water content. This is one of the most efficient and flexible treatments used for moisture reduction of organic industrial and domestic sludge. Under the condition of high temperature and pressure, proteins are hydrolysed causing cell destruction, organic compound solubilisation and free ammonia emissions. This method is not sludge sensitive and results in a high concentrated product normally suitable for incineration, landfill and land application, depending on heavy metals and pathogens concentration.

Sludge management in Catalonia

In Catalonia (North-est of Spain) in response to the Urban Wastewater Treatment Directive 91/271/EEC (Council of the European Union, 1991) around 50% of the existing WWTP (170) were constructed between 2000 and 2006. According to the Catalan Water Agency (2010b), 143,000 t of sludge (dry weight) were produced in 2007 in Catalonia, mainly from medium or small facilities (<1,500 t dry matter/year).

The typical sludge management flow in Catalonia (Figure 1.2) consists of sludge dewatering followed, when possible, by sludge digestion. Digested sludge is commonly post-treated in a composting plant or by thermal drying. Depending on the properties of biosolids, they can be used for land application (both as fertiliser or soil conditioner) or landfilled.

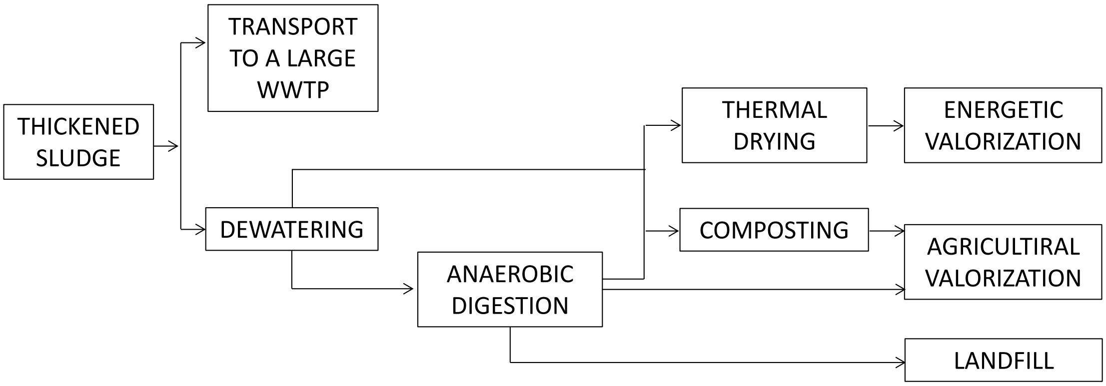


Figure 1.2 Sludge treatment and management flow in Catalonia (adapted from Catalan Water Agency, 2011a).

In small facilities (<2,000 PE) sludge management represent an important problem because sludge stabilisation and dewatering technologies are costly and energy demanding. This is a matter of concern for water authorities, since the number of small wastewater treatment plants (WWTP) in operation will continue to increase within the next years, including municipalities below 500 PE. Only in Catalonia, 1,500 facilites are planned to be constructed by 2014 (Figure 1.3) (Catalan Water Agency, 2007 and 2011b).

Figure 1.3 WWTP in full- operation (in green), under construction (in red), projected (in yellow) and planned (in blue) in Catalonia (Spain) (from Catalan Water Agency, 2010a).

Nowadays, in Spain, different solutions are adopted in small communities depending on the sludge quality (MARM, 2010):

- Primary sludge is normally stored in a sedimentation tank and transported to the nearest WWTP with a conventional sludge treatment line, posing high environmental risks and costs.
- Secondary sludge can be treated in centrifuge, which is a rather common option employed when the investment and operation costs are not excessive. In case centrifugation solution is not feasible, sludge is transported to a bigger WWTP able to treat it.

However, during the last 30 years, Sludge Treatment Wetlands (STW) have been employed as an alternative treatment for sludge management both in small and large facilities. Recent experiences, mainly coming from North Europe, presents excellent results in sludge dewatering and stabilisation with low energy demand and maintenance costs.

In spite of the favourable climate, this solution is far less common in the Mediterranean Region, where it might be successfully adopted as an alternative treatment for small communities.

2. Objectives

The aim of this PhD Thesis was to study Sludge Treatment Wetlands (STW) as an alternative technology for sludge management, with special focus on small communities (<2000 PE) of the Mediterranean Region. Technical, environmental and economic aspects were evaluated in three full-scale treatment systems and in a pilot plant located in Catalonia (Spain); and compared with conventional technologies for sludge management. Finally, design and operation criteria for the implementation of STW in small Mediterranean communities were proposed.

The specific objectives of this work were:

- To summarise the state of the art of STW, including the main design and operation characteristics, and a comparison with conventional sludge treatment technologies.

- To evaluate the performance of three full-scale systems located in Catalonia (Spain), in terms of sludge dewatering, mineralisation and hygienisation.

- To compare the performance of different STW configurations by means of a pilot scale experiment in order to optimise design and operation parameters.

- To model sludge dewatering within STW with the aim of optimising operation patterns.

- To characterise the biosolids produced at the end of the treatment cycle in order to assess their suitability for land application.

- To develop a methodology to measure greenhouse gas emissions from STW and determine the Global Warming Potential of the technology.

- To compare STW and conventional sludge treatments from a technical, economic and environmental point of view.

- To propose design and operation criteria for the implementation of STW in small Mediterranean communities.

3. State of art

This chapter is based on the article:

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

Sludge management has become a key issue in wastewater treatment, representing some 20–60% of the operational costs of conventional wastewater treatment plants. The high water content of the sludge results in large daily flow rates to be handled and treated. Thus, the search for methods to improve sludge volume reduction continues to be of major interest. The technology known as sludge treatment wetlands has been used for sludge dewatering since the late 1980s. Major advantages include its low energy requirements, reduced operating and maintenance costs, and a reasonable integration in the environment. However, the number of plants in operation is still low in comparison with conventional technologies. This study represents a review of the state of the art of sludge treatment wetlands. The main characteristics and operational aspects of the technology are described, including a summary of the main results reported in the literature. Finally, the efficiency of sludge treatment wetlands versus conventional treatments is compared.

Introduction

Sludge management has become a key issue in urban and industrial wastewater treatment for two main reasons: 1) large amounts of sludge are generated as a waste or by-product of wastewater treatment processes; and 2) solid waste management and disposal are among the most complex problems of wastewater treatment facilities. In general, sludges and biosolids resulting from wastewater treatment operations are in liquid form, typically containing 0.5% to 15% total solids (TS). Most of them are organic compounds, with a broad range of volatile solids (VS) contents (50-80% VS/TS), commonly from 75% to 80% of TS (Von Sperling and Gonçalves, 2007). Sludge production and characteristics are highly dependent on the wastewater composition and the treatment used.

The main sludge treatment operations are aimed at increasing the concentration of total solids in order to reduce the sludge volume (i.e. sludge thickening and dewatering) or decreasing the concentration of volatile solids and stabilising the biodegradable fraction of organic matter (i.e. sludge stabilisation via anaerobic digestion or composting) (Werther and Ogada, 1998). Decreasing sludge volume by means of dewatering technologies reduces the costs of sludge handling, transportation and final disposal. Furthermore, sludge dewatering is always required prior to treatments such as composting, incineration or landfilling. Dewatering may be carried out by using conventional mechanical processes such as centrifugation and filtration, or by using other processes such as water evaporation, evapotranspiration (ET) and percolation. Sludge treatment wetlands are extensive treatments that achieve sludge dewatering and mineralization by means of the latter processes.

In general, sludge treatment systems involve high costs, ranging from 20% to 60% of the total operating cost of wastewater treatment plants (WWTP) (Wei et al., 2003; Von Sperling and Andreoli, 2007). This is particularly critical in the case of WWTP of small rural communities, which, in practice, may then transport raw sludge to larger WWTP instead of implementing their own sludge treatment line. The use of sludge treatment wetlands may provide an opportunity to treat the sludge within the WWTP of this type of communities.

Sludge treatment wetlands, are rather new sludge treatment systems based on treatment wetlands (TW). TW are being used in many regions of the world for wastewater treatment (Caselles-Osorio et al., 2007), and are made up of shallow ponds, beds or trenches filled with a gravel layer and planted with emergent rooted wetland vegetation such as Phragmites australis (common reed) (Cole, 1998).

Sludge treatment wetlands have been used in Europe for sludge dewatering and stabilisation since the late 1980s. The largest experience comes from Denmark, where there are over 140 full-scale systems currently in operation (Nielsen, 2008). Other systems implemented in northern Europe are located in Poland (Hardej and Ozimek, 2002; Obarska-Pempkowiak et al., 2003), Belgium (De Maeseneer, 1997) and the United Kingdom (Edwards et al., 2001). In the Mediterranean region, full-scale systems are operating in Italy (Giraldi et al., 2008; Bianchi et al., 2010), France (Liénard et al., 1995; Troesch et al., 2008a) and Spain (Uggetti et al., 2009a). Several pilot plant trials have been carried out in Palestine (Nassar et al., 2006), Cameroon (Kengne Noumsi et al., 2006) and, more recently, in Greece (Stefanakis et al., 2009; Melidis et al., 2010), France (Vincent et al., 2010), China (Cui et al., 2008), Thailand (Koottatep et al., 2005; Panuvatvanich et al., 2009) and Brazil (Magri et al., 2010). The US experience of sludge treatment wetlands has been reported by Kim and Smith (1997), Burgoon et al. (1997), Summerfelt et al. (1999) and Begg et al. (2001).

The aim of this review is to present the current state of the art of sludge treatment wetlands. The main design characteristics and operational aspects of the technology are described, including a summary of the main results reported in the literature. Finally, the efficiency of sludge treatment wetlands is compared with that of conventional mechanical sludge treatments.

General aspects of sludge treatment wetlands

Sludges from different sources have been treated in wetlands, including anaerobic digesters (Nielsen, 2003), aerobic digesters (De Maeseneer, 1997), conventional activated sludge systems (Nielsen, 2003, Obarska-Pempkowiak et al., 2003; Troesch et al., 2009a), extended aeration systems (Begg et al., 2001; Edwards et al., 2001; Nielsen, 2003, Uggetti et al., 2009a), septic tanks (Summerfelt et al., 1999; Tresch et al., 2009b; Vincent et al., 2010), and Imhoff tanks (Zwara and Obarska-Pempkowiak, 2000). In general, sludge conditioning with chemical or thermal treatments is not needed for the operation of the wetlands (Nielsen, 2003).

Sludge is directly spread into the basins from the aerations tanks (Nielsen, 2003) or is previously homogenised in a buffer tank before its discharge into the wetlands (Figure 3.1). From this tank, the sludge is diverted into one of the beds (wetlands), following a semi-continuous regime. The number of beds may vary, according to the treatment capacity of the facility, between 3 (Uggetti et al., 2009a; Uggetti et al., 2009b) and 18 (Nielsen, 2003), which correspond to 400 and 123,000 population equivalent (PE), respectively. Beds surface is also highly variable, between 4.5 m^2 (Summerfelt et al., 1998) and more than 1000 m^2 (Nielsen, 2007). Beds may be constructed in rectangular concrete basins (Uggetti et al., 2009b) or soil excavated basins (Nielsen, 2003) as shown in Figure 3.2.

The bottom of each basin is covered with a waterproof membrane to seal off the bed and prevent leaching. A minimum slope of 1% is desirable to ease leachate collection through a number of perforated pipes, which are placed along the bottom of the bed (Figure 3.3).

These pipes also enhance aeration through the gravel filter and sludge layers. Sludge is fed by means of pipes which may be located in a corner of the bed, along one of the bed sides or in the middle of the basin (upflow vertical pipes).

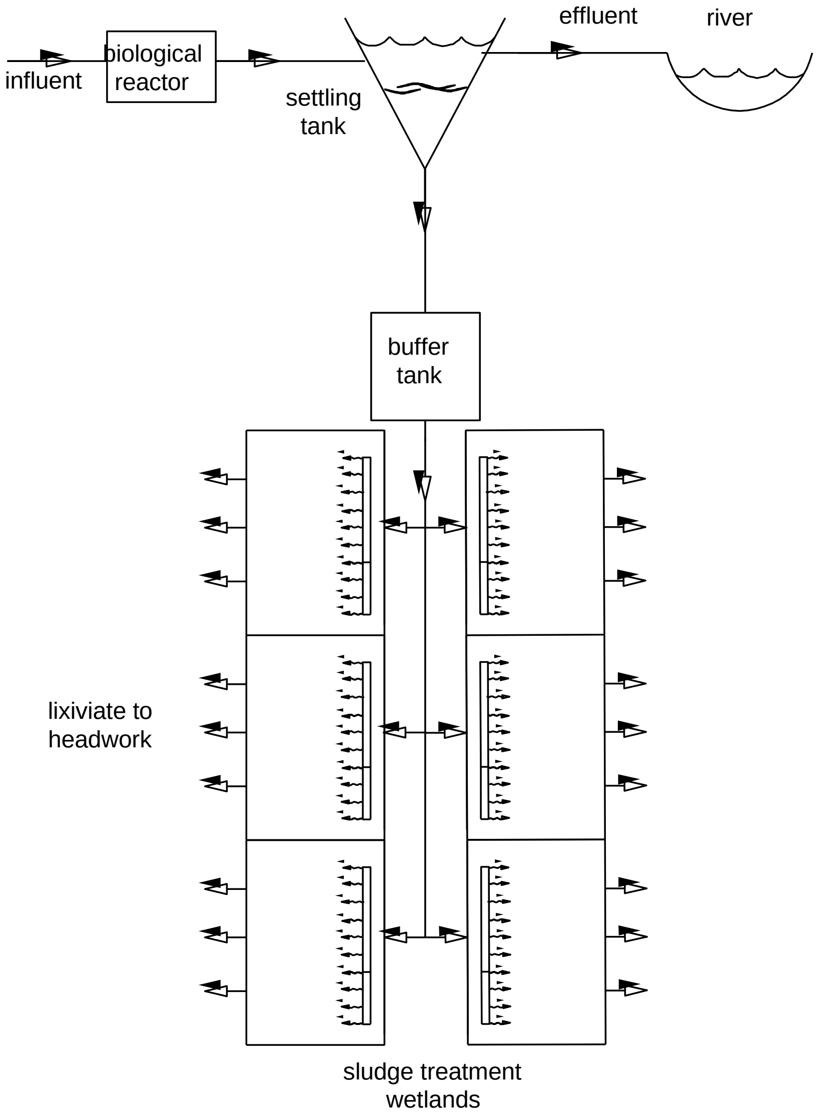


Figure 3.1 Schematic diagram of a sludge treatment wetland system for the treatment of activated sludge from an extended aeration process.

Figure 3.2 Concrete wetland on the top and excavated wetland on the bottom, located in Skovby and Hadsten (Denmark) respectively.

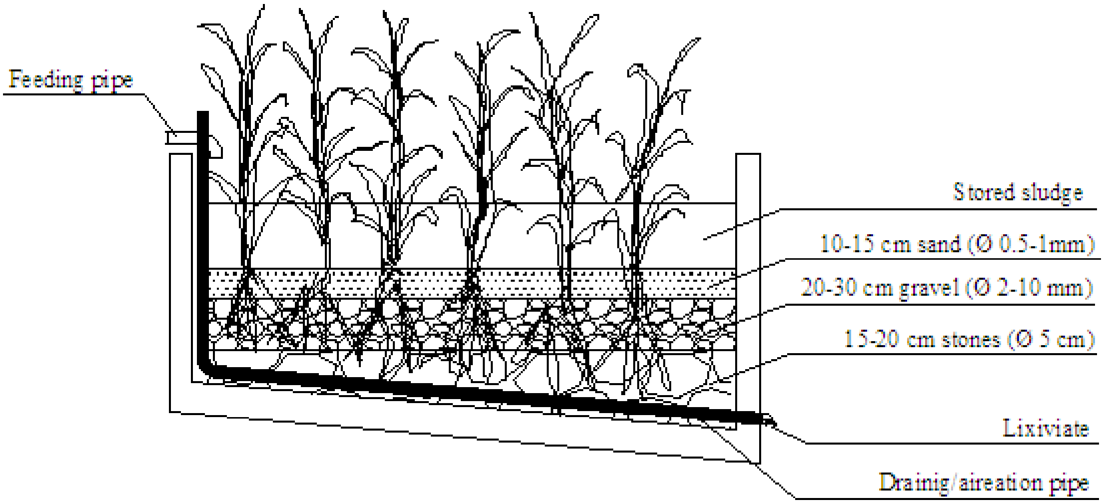


Figure 3.3 Schematic diagram of a sludge treatment wetland.

In the wetlands, drying is undertaken as a batch process in such a way that sludge is fed each time to one of the beds during a feeding period that may last 1–2 days (or even 1–2 weeks).

After the feeding period, the bed rests and the sludge is dewatered, while the influent sludge is discharged to another bed. Resting periods may last a few days or weeks, depending on the treatment capacity, weather conditions, age of the system, dry matter content and thickness of the sludge (Nielsen, 2003). In the following cycle, sludge is spread anew on the residual layer during the same feeding period, and dewatered during the subsequent resting time. After each load, solids remain on the surface and the sludge is dewatered mainly due to water percolation through the sludge residue and the granular medium. Residual water content is further reduced by plant evapotranspiration (Nielsen 1990; De Maeseneer, 1997). By increasing the number of beds in the system it is possible to establish longer rotation series, and hence longer resting periods, which increases the dryness of the sludge residue.

In addition, during a resting period, stored sludge forms a dry surface film that is cracked due to the plants' movement (Figure 3.4). Usually plants reduce the surface cracking decreasing the number of large cracks and increasing small-medium cracks. The fractures on the sludge layer enhance water evaporation and oxygen transfer, which promotes a more uniform porosity along the bed and sludge mineralisation at the bottom level. Indeed, oxygen transfer by the plants from the air to the roots and through the cracked surface and via filter aeration creates aerobic conditions in some zones of the sludge layer, promoting the existence of aerobic microorganisms and ultimately improving sludge mineralisation (Nielsen, 2003; Nielsen, 2005).

Figure 3.4 Detail of the cracked surface of a wetland.

At the present moment, there is no standard recommended strategy for loading and resting periods. Giraldi et al. (2008) indicated that the management of facilities may be optimised by implementing numerical models that consider both dewatering and mineralisation processes.

The sludge layer height of the wetlands increases at a certain rate, and when the layer approaches the maximum height, feeding is stopped during a final resting period (from 1–2 months to 1 year), aimed at improving final sludge dryness and mineralisation. The final product is subsequently withdrawn (i.e. with a power shovel). Attention must be paid not to withdraw the lower layer of sludge residue in which the remaining plant roots will regenerate the vegetation without requiring replanting (Nielsen, 2003).

The result of sludge dewatering and stabilisation processes is a final product that is suitable for land application, either directly or after additional composting. In general, heavy metal concentrations in this product are within the limits for unrestricted land application of the sludge (Uggetti et al., 2009a), although they obviously depend on the sewage composition. To some extent, sludge treatment wetlands provide faecal microbial inactivation, but additional hygienisation might be needed for an unrestricted application of the product in agricultural crop fields (Zwara and Obarka-Pempkowiak, 2000; Nielsen, 2003; Uggetti et al., 2009a).

Some aspects of the treatment have not yet been investigated. From an environmental point of view, the impact of greenhouse gas emissions should be dealt with in future research studies (i.e. plant carbon uptake and subsequent CO_2 and CH_4 release from the sludge layer). From a technical and economic point of view, the development of numerical models for predicting water removal from sludge would enhance the optimisation of loading and resting patterns, and constitute a powerful tool for the design and management of full-scale facilities.

Characteristics of the design

Range of application

Over the last 20 years, sludge treatment wetlands have been implemented at WWTP in communities ranging from 400 to 1500 PE in Spain (Uggetti et al., 2009) and Poland (Obarska-Pempkowiak et al., 2003) to 30,000 PE in Italy (Peruzzi et al., 2007) and even 60,000–125,000 PE in Denmark (Nielsen, 2003). Therefore, the systems' capacity is not a limiting factor, and wetlands may be used if sufficient land is available. The surface area required for sludge treatment varies between 1.5 and 4 PE per m^2 (De Maeseneer, 1997). However, some parameters as sludge composition and climate conditions should be taken into account when dimensioning sludge treatment facilities (Nielsen, 2005).

Design and configuration of wetlands

At present, there are no standard values of design factors and configurations of sludge treatment wetlands. The main design factor is the sludge loading rate, which dictates the required surface area and therefore allows the dimensioning of the systems. Table 3.1 shows the number of beds, surface area of each basin and the maximum sludge loading rate from several studies on sludge treatment wetlands.

Nielsen (2003) set loading criteria to a maximum of 60 kg dry matter/m^2·year, and recommended a maximum of 50 kg dry matter/m^2·year for sludge with high fat content or with low age (< 20 days). Similar loading rates were adopted by Burgoon et al. (1997), who increased the solids loading from 9.8 kg/m^2·year during the start-up phase to a design load of 65 kg/m^2·year. According to Edwards et al. (2001), who studied the solids loading in a pilot plant in the UK, the design load of 60 kg dry matter/m^2·year may be exceeded during the summer as a result of the higher plant evapotranspiration (ET). Higher loading rates (up to around 100 kg dry matter/m^2·year) have also been applied, according to the sludge and climate conditions (Crites and Tchobanoglous, 1998). Actually, loading rates higher than 200 kg dry matter/m^2·year were recently applied in a full-scale system Greece (Melidis et al., 2010) and in pilot plants located in Cameroun (Kengne Noumsi et al., 2006), Thailand (Panuvatvanich et al., 2009) and Brazil (Magri et al., 2010). However, 60 kg dry matter/m^2·year is the loading rate nowadays recognised for STW design in Europe. During the start-up period (after plantation of beds), which may last some months (up to three years according to Nielsen, 2003), the loading rate might be lower than the design rate in order to enhance plant growth and vegetation development, and to protect the plants from any possible stress (Nielsen, 2003; Burgoon et al., 1997).

Table 3.1 Number of beds, total surface area and sludge loading rate in different studies on full-scale sludge treatment wetlands.

Number of Beds	Surface of each bed (m^2)	Sludge loading rate (kg TS/m^2·year)	Reference
25	1,000	65	Burgoon et al. (1997)
-	495	-	Begg et al (2001)
2	240	-	Obarska-Pempkowiak et al. (2003)
8	500	60	Nielsen (2005)
10	1,050	60	Nielsen (2007)
8	468	22-44	Troesch et al. (2008b)
3	66	55	Uggetti et al. (2009a)
6	54	51	Uggetti et al. (2009a)
7	25	125	Uggetti et al. (2009a)
2	140	284	Melidis et al. (2010)

An accurate dimensioning of the systems requires an assessment of sludge loading rates and of the duration of feeding and resting periods in order to determine the required surface area and the number of beds. These two characteristics will ultimately determine the life span of each operating cycle. However, feeding patterns and resting periods are not standardised. While some Danish systems were fed for 7–8 days and rested for 55–56 days, others were fed for 2–3 days and rested for 14–21 days (Nielsen, 2005; Nielsen, 2007). Similarly, 2 days of feeding were followed by 14 days of rest in a full-scale system in France (Troesch et al., 2008b). There are even studies on systems that were loaded only 3–8 times per year (Summerfelt et al., 1998; Obarska-Pempkowiak et al., 2003). According to Nielsen (2003) is important to operate rapid loading of short duration (pump performance must result in a water level of 0.15m in one hour) followed by a resting period of the residual sludge. Furthermore, the number and size of the beds is not standard; for instance, the literature reports systems with 2 beds of 240 m^2 (Obarska-Pempkowiak et al., 2003) and others with 25 beds of 1000 m^2 (Burgoon et al., 1997). There are no specific design criteria for the shape of beds, although they tend to be rectangular with a variable length-to-width ratio.

The most common value for the total depth of the beds is around 2.4 m (0.6–0.7 m of filter medium and 1.5–1.6 m for sludge accumulation). In fact, the height of the bed should ensure a treatment capacity of at least 1 m of sludge residue, corresponding to a maximum sludge layer increasing rate of approximately 10 cm/year with an operating cycle life span of 8–10 years (Begg et al., 2001; Nielsen, 2003). A rapidly growing layer of sludge is often a sign of operational problems (Nielsen, 2005). Sludge feeding is usually stopped when the sludge layer is 20 cm below the top of the walls of the beds. Nielsen (2003) recommends a basin depth above the filter layer of no less than 1.70–1.80 m, which allows 1.50–1.60 m of sludge residue to be accumulated.

Granular medium

The granular medium constitutes a filter with a total height ranging from 30 cm (Burgoon et al., 1997) to 50–60 cm (Begg et al., 2001; Nielsen, 2003; Obarska-Pempkowiak et al., 2003; Summerfelt et al., 1999). The filter has several layers of granular media set in increasing size from top to bottom, through which water percolates (Figure 3.3). Leachate is collected by means of draining pipes, which are located at the bottom of the granular medium. While stones (diameter of around 5 cm) at the bottom protect draining pipes, gravel (diameter from 2 to 10 mm) and sand (diameter from 0.5 to 1 mm) in the upper layers provide a primary physical filtration and rooting medium for plants at the beginning of system operation (Begg et al., 2001; Edwards et al., 2001). In fact, the sand layer at the top of the filter acts as a separation interphase between the granular medium and the sludge layer. The sand layer retains solids, which prevents clogging processes that would impede water percolation through the medium pores (Platzer and Mauch, 1997). Likewise, it is important to maintain the capillarity connection between the sludge and the filter layer in order to avoid hydraulic failures, otherwise insufficient dewatering may occurs (Nielsen, 2005). Moreover, when treated sludge is removed (i.e. at the end of each cycle), the sand layer protects the main filter layer, which does not need to be replaced.

For a standard granular medium with a height of 30–60 cm, the most common layer heights are 15–20 cm for stones, 20–30 cm for gravel and 10–15 cm for sand. Filter heights are variable and do not seem to significantly affect the treatment efficiency. In recent studies, Troesch et al. (2008a and 2008b) evaluated the effect of replacing the sand layer with a compost layer of 5–10 cm in a pilot plant. The results indicated that the vegetal compost layer was a better growing media for plants, but it had a lower filtration capacity than the sand altering the filtering function. Nevertheless, dewatering efficiencies were similar with both media, further research in this sense may be useful to corroborate compost effects as a medium layer.

Draining pipes are opened to promote air movement through the pipes and granular medium (Figure 3.5). The significance of such a passive aeration system was shown by Lienard et al. (1995). According to experimental results conducted in unplanted beds, dewatering may be accelerated by means of a mechanical aerator that injects air into the sand layer in order to crack the sludge layer (Yamaoka and Hata, 2003).

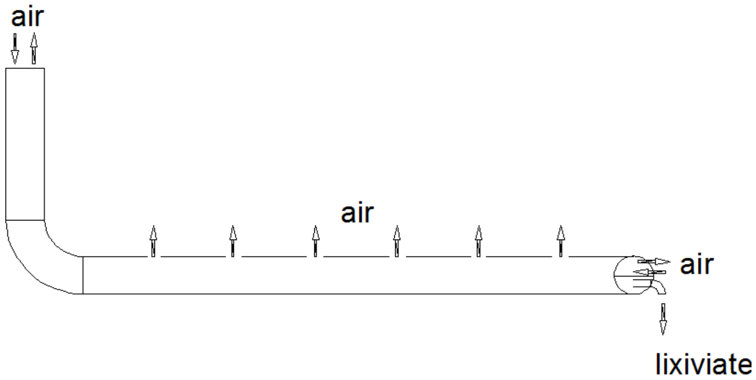


Figure 3.5 Detail of the draining and aeration pipe.

Plants

Plants are a key element of sludge treatment wetlands, since they assist sludge dewatering and mineralisation. Their impact on the efficiency of the process was clearly demonstrated in the work of Edwards et al. (2001), in which planted and unplanted beds were compared. In this study planted beds showed higher TS concentration (20-21%) than unplanted (18%) and higher slugde height reduction (84-86% in planted beds and 81% in unplanted).

Plant species used in treatment wetlands have to be able to grow in watery, muddy, anaerobic conditions and at the same time they must be able to tolerate oscillations in water level, high salinity and variations between high and low pH (De Maeseneer et al., 1997). However, it is important to provide suitable condition for vegetation growth by applying the right sludge loading rate during the start-up phase. Plantation density may vary between 4 rhizomes/m^2 (Edwards et al., 2001) and 15 rhizomes/m^2 (Magri et al., 2010).

The most widely used species in treatment wetlands for wastewater as well as sludge treatment is the common reed (*Phragmites australis*) (Puigagut et al., 2007). Hardej and Ozimek (2002) evaluated the effect of sewage sludge on growth and morphometric parameters of *Phragmites australis* and demonstrated the high adaptation capacity of the common reed to the sewage sludge environment, observing that the shoot density was over two times greater than that commonly found in natural systems. Cattail (*Typha* sp.) has also been extensively used in wastewater treatment wetlands, in particular due to its high initial growth rate (De Maeseneer et al., 1997). However, according to Magri et al., (2010) and to Koottatep et al. (2005), *Typha* sp. showed a hardest adaptation in sludge systems, compared to other species. Other species like *Cyperus papyrus* L. and *Echinochloa pyramidalis* (Lam.) demonstrate a good adaptation to high sludge loading rates (up to 300 kg dry matter/m^2·year) in a study conducted in Cameroun (Kengne Noumsi et al., 2006).

Plants contribute to sludge dewatering mainly by ET. Chazarenc et al. (2003) estimated ET values ranging from 4 to 12 mm/day in a 1 m^2 wastewater pilot plant planted with *Phragmites australis* located in France. For the same plant species, higher values were found during summer in two pilot scale experiments, where ET oscillates between 25 and 38 mm/day in the North of Italy and between 32 and 50 mm/day in the South (Borin et al., 2010). According to these results, ET values are extremely variable depending on the season, air temperature, wind velocity and relative humidity. Considering the Radiation Method (Shaw, 1994), the evapotransiration of a reference crop (ET_0) is a function of mean humidity, wind conditions, temperature, altitude and solar radiation (sunshine hours). Therefore, at lower latitudes high ET values may be more favourable for sludge dewatering. In accordance with Allen et al. (1998), to know the specific ET for a plant species, one approach is to adjust the ET_0 by means of an appropriate crop coefficient (k_c), which varies depending on the season and the geographical location. The k_c relates the ET_0 with the ET of a given plant species. Hedges et al. (2008) found k_c values of around 1 for *Phragmites australis* in the UK; while, according to Borin et al. (2010), k_c varies between 2 and 8 depending on the season, but following the same pattern in North and South Italy.

Plants contribute to sludge mineralisation through the transport of oxygen from the aerial parts to the belowground biomass. This oxygen is released in the rhizosphere, which creates aerobic microsites in the bulk sludge layer and thus ensures appropriate conditions for aerobic degradation processes and other oxygen-dependent reactions like nitrification (Vymazal, 2005). Plants also indirectly contribute to aerobic mineralisation through stems, which as a result of their movement (by the wind) crack the surface of dry sludge and prompt aeration of the upper sludge layers. In addition, the effect of the movement of the stems and the complex root system support pore maintenance within the sludge layer and preserve drainage efficiency through the gravel filter (Nielsen, 2003). However, oxygen transported by the plant in STW has not yet been quantified.

Leachate

The water content in a typical secondary sludge (with 5% TS) can be classified as pore water (66.7%), capillary water (25%), and adsorbed and structurally bound water (8.3%) (Nielsen, 2003). After each sludge feeding event into the wetlands, there is a rapid water loss due to percolation, which mainly consists of pore water removal (Gonçalves et al., 2007). According to Nielsen (2005), in a facility with operational problems, loading events are not immediately followed by water percolation, thus the dewatering is slowly and incomplete. It is important to control leachate quantity and quality since percolated water is collected by draining pipes as leachate and is returned to headworks.

Analysis of leachates from full-scale sludge treatment wetlands has demonstrated their general low organic matter content (around 100 mg/L of COD) (Huertas et al., 2004; Troesch et al. 2008b, Troesch et al., 2009a; Cui et al., 2008). However, in some cases higher values were found (between 500 and 6000 mg/L) (Kengne Noumsi et al. 2006; Vincent et al., 2010; Troesch et al., 2009b). Variations are probably due to the organic loading rate, which is strictly dependent upon the wastewater influent quality and the treatment which it has been submitted.

A high concentration of nitrate (up to between 100 and 200 mg NO_3^-N/L) was detected in different studies (Koottatep et al., 2005; Troesch et al., 2009b; Vincent et al., 2010), highlighting the important role of STW in organic nitrogen and ammonia removal (around 87-92%) mainly due to the nitrification (Panuvatvanich et al., 2009). However, further investigations are required in order to determine nitrification rates and improve bed configuration and operation conditions for nitrogen recovery and removal.

Treatment efficiency

Sludge dewatering

The main goal of sludge treatment wetlands is dewatering, which transforms the sludge from a liquid to a solid waste (or by-product), resulting in the so-called sludge cake. The higher the solids content of the sludge, the lower the volume and disposal costs. The efficiency of several sludge treatment wetland facilities for sludge dewatering, as shown by the increase in totals solids (TS) concentration in the sludge, can be observed in Table 3.2. In general, TS concentration increases from 1–4% in influent sludge to 20–30% within the wetlands. Even higher TS values were reported from Polish systems (58%), although they might be a result of a higher TS concentration in feeding sludge (4–10%) and the fact that it was a primary sludge.

Treatment efficiency of STW in terms of dewatering is comparable to that of conventional treatments such as centrifuges, vacuum filters and belt presses (Table 3.3). The most efficient dewatering treatments are centrifuges and filter presses, which may lead to TS concentrations as high as 35%, usually using polymers for sludge conditioning. Nevertheless, most values are within the range of, and sometimes even lower than, those observed for sludge treatment wetlands. Furthermore, such treatments involve higher energy requirements and operation and maintenance costs.

Table 3.2 Total solids (TS) and Volatile solids (VS) concentration observed in several sludge treatment wetlands.

Systems' location	Source of the sludge	TS (%)		VS (%TS)		Reference
		Influent	Wetlands	Influent	Wetland	
Fort Campbell, USA	Anaerobic digestion	3	32 *	-	46 *	Kim and Smith (1997)
Pilot plant in Rugeley, Staffordshire, UK	Biological Aerated Filter (BAF) and raw slurry solids	4	20	74	52	Edwards *et al.* (2001)
Darzlubie, Poland	Imhoff tank	4-10	58	~60	45	Obarska-Pempkowiak *et al.* (2003)
Helsinge, Denmark	Activated sludge and activated sludge from settling tank	0.5-0.7	20	-	-	Nielsen (2003 and 2007)
Alpens, Spain	Activated sludge, extended aeration	0.7-1.5	22-25 *	52-67	39-42 *	Uggetti *et al.* (2009a)
Sant Boi de Lluçanès, Spain	Activated sludge, extended aeration	3	20-28 *	52-42	36-40 *	Uggetti *et al.* (2009a)
Seva, Spain	Activated sludge, contact-stabilization	0.3-2	15-20 *	58-59	46-50 *	Uggetti *et al.* (2009a)

** Average from different depths*

Table 3.3 Typical TS content of the sludge product from conventional sludge dewatering treatments.

Treatment	Source of the sludge	TS (%)	Reference
Centrifuge	Activated sludge	14-20	Gonçalves et al. (2007)
	Anaerobic digestor	15-35	Metcalf and Eddy (2003)
	Aerobic digestor	8-10	Metcalf and Eddy (2003)
Vacuum filter	Activated sludge	12-18	Gonçalves et al. (2007)
	Anaerobic digestor (mixture)	17-23	Gonçalves et al. (2007)
Belt Press	Activated sludge	12-18	Gonçalves et al. (2007)
	Anaerobic digestor (mixture)	17-23	Gonçalves et al. (2007)
	Anaerobic digestor	12-30	Metcalf and Eddy (2003)
	Aerobic digestor	12-25	Metcalf and Eddy (2003)
Filter press	Activated sludge	27-33	Gonçalves et al. (2007)
	Anaerobic digestor (mixture)	29-35	Gonçalves et al. (2007)
Composting	Aerobic digestion and centrifuge	83	Bertrán et al. (2004)
	Dewatered raw sludge	44	Ruggieri et al. (2008)

Sludge stabilisation

The efficiency of several sludge treatment wetland facilities, as shown by the decrease in volatile solids (VS) concentration in the sludge, can be observed in Table 3.4. During sludge treatment within the wetlands, a VS reduction of 25–30% can be achieved, reaching final VS concentrations of between 40% and 50%.

VS removal yields depend on influent sludge VS concentration. For instance, sludge from extended aeration activated sludge systems has lower VS content than that from other treatments (i.e. conventional activated sludge); hence VS removal within the wetlands is lower when this type of sludge is treated. Consequently, the efficiency in terms of VS removal of the wetlands might be slightly lower than that of aerobic digestion (40%–55%) or anaerobic digestion (35%–50%) (Metcalf and Eddy, 2003; Von Sperling and Gonçalves, 2007). Nevertheless, final VS concentrations are nearly those of anaerobic digestion, as can be seen in Table 3.4, where VS contents of sludge treated in conventional systems are reported.

On the other hand, VS contents in compost are considerably higher (60%–70%) than in sludge from other treatments, including wetlands (Table 3.4). However, the nature of organic solids in mature compost is completely different from that of raw sludge, since it is mainly composed of poorly biodegradable complex polymers (i.e. non-putrescible materials).

Table 3.4 VS content of the influent, the effluent and removal efficiency of conventional sludge stabilisation treatments.

System	Type of sludge	VS			Reference
		Influent (% TS)	Effluent (% TS)	Removal efficiency (%)	
Full-scale mesophilic anaerobic digestion	Primary sludge	-	-	47	Krugel et al. (1998)
Full-scale thermophilic anaerobic digestion	Primary +secondary sludge	-	-	60-80	Krugel et al. (1998)
Full-scale mesophilic and thermophilic anaerobic digestion	Primary + secondary sludge	-	53-55	-	Zábranská et al. (2000)
Full-scale mesophilic anaerobic digestion	Scondary sludge	58-70	-	13-27	Bolzonella et al. (2005)
Full-scale mesophilic anaerobic digestion	Secondary sludge	62	54	-	Mininni et al. (2006)
Pilot plant thermophilic anaerobic digestion	Primary + secondary sludge	78	64	33	Ferrer et al. (2008)
Pilot plant thermophilic anaerobic digestion	Primary + secondary sludge	73	-	56-60	Palatsi et al. (2009)
Composting pilot plant	Secondary sludge	-	62	-	Bertrán et al. (2004)
Composting laboratory experiment	Primary + secondary sludge	74	71	-	Ruggieri et al. (2007)

Indeed, this is a main parameter in compost quality assessment. The results of non-stabilising sludge dewatering treatments, such as centrifuges, will depend on previous processes. When centrifuged sludge has not been previously stabilised, final VS values can be as high as 70%.

Sludge stability is used to define the extent to which readily biodegradable organic matter has been decomposed (Lasaridi et al., 1998). Referred to compost, stability is a quality parameter related to the microbial decomposition or microbial respiration activity of composted matter (Komilis et al., 2009), which may be determined by means of respiration indexes (Barrena et al., 2009a, Ponsá et al., 2008). Giraldi et al. (2009a) and Peruzzi et al. (2009) investigated the readily biodegradable fraction of sludge and the activity of microorganisms by the determination of the Water Soluble Carbon (WSC) and the DHase activity (DHase). These authors found an important decrease of both parameters during sludge treatment within wetlands, suggesting the stabilization of organic matter. In addition, they observed some degree of sludge mineralization and humification already after 1 year of treatment.

Nutrients, heavy metals and microbial faecal indicators

The treatment of sludge resulting from wastewater treatment facilities must allow this waste to be converted into a by-product such as an organic fertiliser or soil conditioner suitable for agricultural crop fields or land reclamation. However, the accumulation of organic and inorganic pollutants in the sludge may impede such applications due to the environmental and health hazards that it may involve. Thus, the assessment of such potential pollutants (i.e. heavy metals, faecal microbial indicators, etc.) is needed before land application of the treated sludge, as stated in the European Sludge Directive (Council Directive 86/278/EEC) and recommended in the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000).

The presence of these pollutants is extremely variable depending on the wastewater composition and treatment system. Moreover, while faecal microbial contamination can be reduced with a number of disinfection techniques, heavy metal removal from sludge is complex and expensive. Therefore, an appropriate sludge management strategy must lead to high-quality products that enable sludge recycling. Indeed, agricultural use of sludge is only recommended when harmful effects to soil, agricultural products, human health and the environment can be avoided.

As shown in Table 3.5, a considerable variation in heavy metal concentrations from sludge treatment wetlands is reported in the literature. In fact, the concentration of heavy metals in the sludge is very much dependent on sewage composition. Nevertheless, in almost all cases heavy metals are below the legal limits (Council Directive 86/278/EEC; Environment DG, EU, 2000). Thus, according to the results shown in Table 3.5, sludge from treatment wetlands might be recycled in soils by means of agricultural or land restoration applications.

Table 3.5 Heavy metal concentrations observed in several sludge treatment wetlands. Influent aludge values are expressed in mg/L and wetland values in mg/kgTS.

Systems' Location		Cr	Ni	Cu	Zn	Cd	Pb	Reference
Buckland, USA	Influent	0.14	0.82	11	3	-	0.5	Begg *et al.* (2001)
	Wetland	42	23	1,906	684	3	154	
-	Wetland	-	-	215	1,836	12	341	De Maeseneer (1997)
USA Fort Campbell	Wetland	29*	14*	408*	444*	8	66*	Kim and Smith (1997)
Darzlubie, Poland	Wetland	22	67	28	871	2	31	Obarska-Pempkowiak *et al.* (2003)
RudkøbinDenmark	Wetland	39-99	-	260-470	410-1,100	-	-	Nielsen (2003)
Oratoio, Italy	Wetland	40-73	28-52	383-467	1,108-1,357	< 6	93-121	Peruzzi *et al.* (2007)
Alpens, Spain	Influent	35.8	27.9	227	348	0.41	4.29	Uggetti *et al.*, (2009a)
	Wetland	55	30	390	550	0.6	52	
St Boi de Lluçanès, Spain	Influent	36.4	50.2	183	609	0.66	1.99	Uggetti *et al.*, (2009a)
	Wetland	50	36	160	530	0.7	43	
Seva, Spain	Influent	52.1	25	232	897	0.76	0.95	Uggetti *et al.*, (2009a)
	Wetland	60	40	230	690	1	80	
Law thresholds		-	300-400	1000-1,750	2,500-4,000	20-40	750-1,200	Council Directive 86/278/EEC
Law thresholds (proposal)		800	200	800	2,000	5	500	Environment DG,EU,2000

** Average from different depths*

Heavy metals uptake by plants is likely to be the main biological removal mechanism (Sheoran and Sheoran, 2006) in the case of harvesting. De Maeseneer et al. (1997) reported that the amount of heavy metals uptaken by *Phragmites australis* is lower than in the case of *Salix fragilis* and *Salix trandra*. Peruzzi et al. (2007) detected a slight increase in the heavy metal concentration in *Phragmites australis* shoots after 400 days of systems' operation; being significantly lower than heavy metals concentration in sludge. Furthermore, heavy metals concentrations in sludge and in *Phragmites australis* are generally below disposal standards and do not pose a problem for sludge and reed disposal or recycling (Begg et al., 2001).

With regard to faecal microbial indicators, Obarska-Pempkoviak et al. (2003) showed that *Escherichia coli* in sludge was decreased and pathogenic *Salmonella* bacteria inactivated after 8 months of sludge treatment, indicating an improvement of the microbial condition of treated sludge with respect to influent sludge from an Imhoff tank.

Nielsen (2007) analysed the reduction of faecal bacteria indicators in the sludge from treatment wetlands after a period of 1–4 months from the last loading. The results showed a decrease in concentrations to values below 2 MPN/100g for *Salmonella*, below 10 CFU/g for *Enterococci* and below 200 MPN/100g for *E. coli*. Moreover, the sludge accumulated in depth (> 25 cm) did not appear to be recontaminated by *Salmonella* and Enterococci from subsequent sludge loadings.

Currently, however, only heavy metals concentrations are regulated for land application of sewage sludge (Council of the European Union, 1986). Since treated sludge may have considerable amounts of pathogens, depending on the treatment processes used, limit values for faecal bacteria indicators have also been proposed (Environment DG, EU, 2000). According to this proposal, advanced treated sludge should not contain *Salmonella* in 50 g (wet weight) and the treatment should achieve at least a 6 $\log_{10}$ reduction in *E.coli* to less than $5 \cdot 10^2$ CFU/g. On the other hand, conventional treatments should achieve at least a 2 Log_{10} reduction in *E.coli*.

Comparison of treatment efficiency

For the sake of comparison, Table 3.6 shows the average composition of treated sludge after dewatering and stabilisation in sludge treatment wetlands (Uggetti et al., 2011), composting (Bertrán et al., 2004) and centrifugation in treatment facilities in the region of Catalonia (Spain). Data of sludge from wetlands were collected in three full-scale STW of different size (Figure 7.1); one located in Spain (Seva, 1,500 PE) and two in Denmark (Greve, 50,000 PE and Hadsten, 12,000 PE). The main characteristics of the facilities are summarised in Table 7.1. The comparison has been made between these treatments because centrifugation is among the most widely used dewatering techniques, even in small community WWTP (< 2000 PE), and sludge treatment wetlands are sometimes regarded as a form of passive composting (Metcalf and Eddy, 2003).

Table 3.6 Sludge composition after treatment in treatment wetlands (Uggetti et al., 2009b), centrifugation and composting (Bertrán et al., 2004).

	Sludge treatment wetlands (average of 3 systems)	Centrifuge	Composting
pH	6.14	6.91 (1:2.5)	7.5
CE dS/m (1:5)	1.2	4.2	4.1
TS (%)	22	18	83
VS (% TS)	45	73.4	62
TNK (% TS)	0.1	6.4	2.5
Ptotal (% TS)	0.2	1.8	2.3
Cu (mg/kg)	284	518	388
Zn (mg/kg)	941	807	1,087
Pb (mg/kg)	66	60	110
Cd (mg/kg)	0.93	2	1.5
Ni (mg/kg)	46	15	54
Cr (mg/kg)	-	40	95
Hg (mg/kg)	2.3	4	-

With reference to TS, it is evident that the dryness obtained with sludge treatment wetlands (22%) is similar to that of centrifuges (18%), but significantly lower than that of standard compost (83%).

Organic matter contents (as % VS/TS) range between 45% and 73%. The lowest values are obtained with treatment wetlands (45%), which clearly indicate sludge stabilisation during treatment. On the other hand, higher values for compost (62%) may be explained by an increase in complex organic compounds as a result of sludge stabilisation and mineralisation during the composting process (e.g. an increase in humus-like substances, etc.). Since centrifugation is only aimed at sludge dewatering, the final values of the VS/TS are the highest (73%) and must indeed be similar to those of influent sludge.

Regarding nutrients, a certain amount of nitrogen (2.5–6.4% TKN/TS) and phosphorus (1.1–2.3% Ptotal/TS) is found in centrifuge and composting, while STW nutrient concentration are significantly lower (0.1% TNK/TS and 0.2 %TP/TS). Heavy metals are somewhat higher in compost compared with centrifugation and sludge treatment wetlands (Table 3.6), possibly as a result of higher material dryness and organic matter mineralization. As mentioned above, the concentration of heavy metals in sludge treatment wetlands is well below the limits for unrestricted land application of the sludge (Council Directive 86/278/EEC; Environment DG, EU, 2000).

Conclusions

Sludge management plays a key role in wastewater treatment, which accounts for major operational costs. In this context, sludge treatment wetlands are a potential alternative to conventional sludge treatments. Among others, some major advantages are their low or lack of energy requirements, reduced operation and maintenance costs, and a reasonable integration in the environment. However, despite the favourable climatic conditions of the Mediterranean region for sludge treatment wetlands, the technology is little used at present.

The comparison of sludge treatment wetlands with other (conventional) technologies like centrifugation and composting suggests a high efficiency of the system in terms of sludge dewatering (around 30% TS) and stabilisation (40-50% VS), which leads to a final product that may be suitable for agricultural crop fields and land reclamation, even without further composting of the treated sludge. This provides an opportunity for on-site sludge treatment, especially in WWTP of small communities.

However, this study highlights the lack of standard configuration and design criteria for sludge treatment wetlands. Little up-to-date information on surface loading rates or loading patterns is available, particularly for Mediterranean locations. Therefore, further research is required to improve the knowledge of the design and management of these systems in order to enhance the treatment efficiency.

Some aspects of the treatment have not yet been investigated. From an environmental point of view, the impact of greenhouse gas emissions should be dealt with in future research studies (i.e. plant carbon uptake and subsequent CO_2 and CH_4 release from the sludge layer). From a technical and economic point of view, the development of numerical models for predicting water removal from sludge would enhance the optimisation of loading and resting patterns, and constitute a powerful tool for the design and management of full-scale facilities.

4. Characterisation of full-scale systems in Catalonia

This chapter is based on the article:

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

Optimization of sludge management can help reducing sludge handling costs in wastewater treatment plants. STW appear as a new and alternative technology which has low energy requirements and reduced operating and maintenance costs. The objective of this work was to evaluate the efficiency of three full-scale STW in terms of sludge dewatering, stabilization and hygienisation. Samples of influent sludge and sludge accumulated in the wetlands were analysed for pH, Electrical Conductivity (EC), Total Solids (TS), Volatile Solids (VS), Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD), nutrients (Total Kjeldahl Nitrogen (TKN) and Total Phosphorus (TP)), heavy metals and faecal bacteria indicators. Leachate samples were also collected. There was a systematic increase in the TS concentration from 1-3% in the influent to 20-30% in the beds, which fits in the range obtained with conventional dewatering technologies. Progressive organic matter removal in the beds was also observed (VS decreased from 52-67% TS in the influent to 31-49% TS in the beds). Concentration of nutrients of the sludge accumulated in the beds was quite low (TKN 2-7% TS and TP 0.04-0.7% TS), and heavy metals remained below law threshold concentrations. *Salmonella* spp. was not detected in any of the samples, while E.coli concentration was generally lower than 460 MPN/g in the sludge accumulated in the beds. The studied systems demonstrated a good efficiency for sludge dewatering and stabilization in the context of small remote wastewater treatment plants.

Introduction

Sewage sludge is the main organic waste or by-product generated in wastewater treatment plants (WWTP). In general it has total solids concentration around 1-3%, depending on the treatment process (De Maeseneer, 1997). Although sludge production represents only 1-2% of the volume of wastewater treated, its management is highly complex and accounts for 20 to 60% of the total operating costs of WWTP (Von Sperling and Andreoli, 2007). For this reason, sludge management is a matter of concern in many countries, especially if sludge production is expected to increase in the next years as a result of the implementation of new WWTP (Von Sperling and Andreoli, 2007). Therefore, optimization of sludge management becomes a key element to reduce WWTP costs.

Taking into account that sludge moisture is typically as high as 97-99%, a first approach for its management is to decrease the volume by means dewatering processes, reducing in this way sludge disposal costs and environmental risks associated. A major problem of conventional sludge treatment and dewatering technologies is that they are costly and high energy demanding. Therefore, conventional technologies are not often feasible in small WWTP (< 2,000 population equivalent (PE)). In this case waste sludge is usually stored in lagoons and drying beds (Obarska-Pempkowiak et al., 2003) or it is transported to the nearest WWTP with a conventional sludge treatment line. In this context, STW appear as a new and alternative technology, which has low energy requirements, reduced operating and maintenance costs, and causes little environmental impact.

STW are based on constructed wetlands, which are being used for wastewater treatment in many regions of the world (Caselles-Osorio et al., 2007). CWs are land-based treatment systems that reproduce self-cleaning processes that occur in natural wetlands. STW consist of shallow tanks filled with a gravel layer and planted with emergent rooted wetland plants such as *Phragmites australis* (common reed) (Cole, 1998). During the last 30 years this technology has been successfull-y used and improved especially in Denmark (Nielsen, 2008).

In these systems, waste sludge is pumped and spread on the wetlands surface, 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. The roots of the plants contribute to the oxygen transfer through sludge layers creating aerobic microsites that promote sludge mineralization and stabilization (Reed et al., 1988). Furthermore, the complex root system maintains pores and small channels within sludge layer that preserve the drainage efficiency through the bed (Nielsen, 2003). When the sludge is dry, the movement of plant stems by the wind causes cracking of the surface of the beds and subsequently improves the aeration of the sludge layer.

Changes of sludge composition in time are the result from dewatering processes (draining and evapotranspiration) as well as the degradation of organic matter (Nielsen, 2003). Thus, besides dewatering, STW also allow for a certain degree of sludge mineralization. The resulting final product is suitable for land application (Nielsen and Willoughby, 2005), and might be further treated to improve sludge hygienisation (Zwara and Obarska-Pempkowiak, 2000), promoting in either case sludge reuse as opposite to sludge disposal in landfill or sludge incineration.

Most studies on STW show excellent results in Northern Europe (Edwards et al., 2001). This technology is by far less common in Mediterranean regions, including Spain, despite having much favourable climates for the performance of this type of systems. Nowadays, there are still only a few systems in Spain (Figure 4.1), according to our knowledge all of them located in the region of Catalonia and operated by the company Depuradores d’Osona SL

The objective of this study was to evaluate the efficiency of three full-scale STW in terms of sludge dewatering, mineralization and hygienisation. This work aims at gaining knowledge on the implementation and performance of STW in Mediterranean regions.

Figure 4.1 STW in Catalonia (Spain). From the top Alpens, Sant Boi de Lluçanès and Seva.

Materials and methods

Systems description

The facilities studied in this work are located in Alpens, Sant Boi de Lluçanès and Seva, all in the province of Barcelona (Catalonia, Spain). Main characteristics of the wastewater treatment plants and sludge treatment wetlands are summarized in Tables 4.1 and 4.2, respectively. Average temperature in the area is 6 °C in winter and 20 °C in summer. These WWTP receive varying amounts of subsurface water that infiltrates into the sewer systems.

The wetlands in Seva were set-up in 2000 by transforming the existing conventional drying beds. The first operating cycle was finished in 2004, when the sludge was removed and the process was re-started in 2005. On the other hand, the facilities in Alpens and Sant Boi de Lluçanès were already designed as STW, and the processes started to operate in winter 2006.

In all wetlands the draining filter layer is approximately 55 cm high, and consists in a bottom layer of gravel (diameter from 1 to 3 cm) of about 30 cm and a upper layer of sand (diameter from 0.3 to 1 mm) of about 25 cm. The beds were planted with common reed with a density of 4 plants/m^2. Biological waste sludge from the corresponding WWTP is periodically spread on top of the beds.

Table 4.1 Main characteristics of the wastewater treatment plants at Alpens, Sant Boi de Lluçanès and Seva.

	Alpens	**Sant Boi de Lluçanès**	**Seva**
Population equivalent	400(800 design)	600(1500 design)	1500
Type of treatment	Extended aeration	Extended aeration	Contact-stabilisation
Wastewater flow rate (m^3/d)	70-90	200-250	180 (summer) 400 (winter)
Sludge production (kg TS/d)	30	45	60
Sludge flow (m^3/day)	2	3	4.5

Sludge loading patterns during the period of this study were almost the same in the beds of Alpens and Sant Boi de Lluçanès (55 and 51 kg TS/m^2·year respectively) (Table 4.2). These beds are automatically fed and the sludge loading rates are within the range suggested in the literature (50-60 kg TS/m^2.year; Burgoon *et al.*, 1997; Edwards *et al.*, 2001; Nielsen, 2003). In Seva the beds are fed daily with a manual system and the sludge loading rate (125 kg TS/m^2·year) is significantly higher than recommended values. In all facilities only one bed is fed at a time, whilst the rest remain draining. In Alpens and Sant Boi de Lluçanès the beds are fed during 2 days, followed by 4 resting days in Alpens and 10 days in Sant Boi de Lluçanès. In Seva, due to the manual feeding system employed there is not a regular feeding pattern.

Table 4.2 Main characteristics of the studied wetlands at Alpens, Sant Boi de Lluçanès and Seva.

	Alpens	Sant Boi de Lluçanès	Seva
Number of wetlands	3	6	7
Total surface area (m^2)	198	324	175
Bed surface area (m^2)	66	54	25
Nominal height for sludge accumulation (m)	0.65	0.65	~0.80
Total nominal volume for sludge accumulation (m^3)	128	210	~140*
Sludge loading rate (kg TS/m^2·year)	55	51	125
Loading pattern	2 min every 4 h	2 min every 4 h	manual

*The bottom of Seva's beds is not levelled

Field methods

Three sampling campaigns were carried out, each one in a different season: fall 2007, and spring and summer 2008. Samples of influent sludge and sludge accumulated in the same representative wetland of each facility (in the three campaigns) were analysed to study process performance. Sludge height, or thickness, inside the beds was also measured.

In order to obtain representative composite sludge samples, each representative bed was divided into three sections along the length, namely inlet, middle and final zones. The sampling point of each section in Alpens and Sant Boi de Lluçanès was upon the middle of the width, as shown in Figure 4.2. In Seva the thickness of the sludge accumulated hampered sample collections from the middle, thus within each section samples were taken from both sides. Furthermore, in each sampling point, samples were taken from two depths, corresponding to a surface layer (from the surface of the sludge to half height) and a bottom layer (from half height to the bottom). Composite samples from each depth layer were obtained by mixing the sludge subsamples from different points (3 in Alpens and Sant Boi de Lluçanès, and 6 in Seva), as proposed by Obarska-Pempkowiak et al. (2003). Sludge core samples were collected using Eijkelkamp soil coring kit (Figure 4.3) and preserved at 4 °C until they were processed in the next days. During the second campaign a leachate sample was taken from the bed evaluated in each facility in order to evaluate its quality.

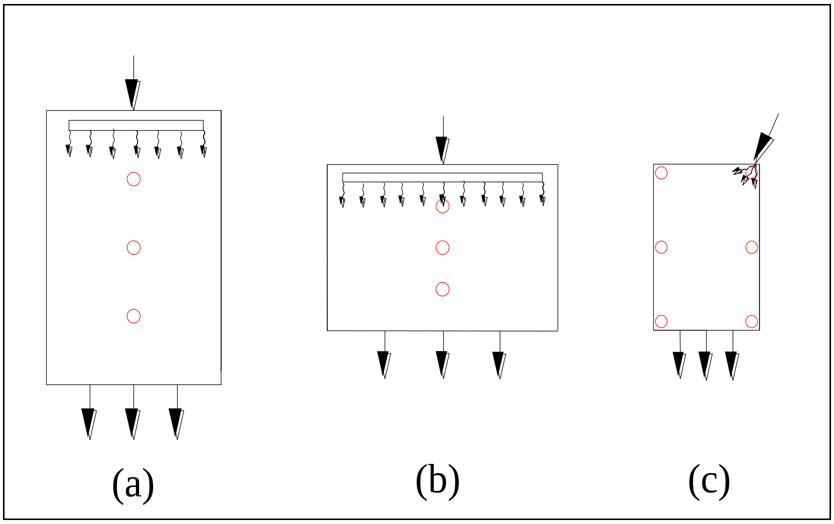


Figure 4.2 Sampling points in the representative wetlands of Alpens (a), Sant Boi de Lluçanès (b) and Seva (c). Note that in Alpens and Sant Boi de Lluçanès the sludge is discharged to the beds by means a channel with a weir, while in Seva by a pipe. Exit arrows represent the outlet of water drainages.

Sludge characterization

Samples collected during sampling campaigns were analysed (generally in triplicate) using conventional methods following the procedures indicated in the Standard Methods (APHA-AWWA-WPCF, 2001). The parameters analysed were those recommended in the literature for characterization of sludge quality and sludge treatment process efficiency (Mujeriego and Carbó, 1994; Obarska-Pempkowiak et al., 2003; Soliva, 2001): pH, Electrical Conductivity (EC), Total and Volatile Solids (TS and VS), Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD_5 as easily biodegradable organic matter and BOD_{21} as easily as well as slowly biodegradable organic matter), Total Kjehldahl Nitrogen (TKN), Total Phosphorous (TP), heavy metals and faecal bacteria indicators (*Salmonella* spp. and *E. coli*). pH and EC were analysed on dried samples diluted in distilled water with a 1:5 ratio. Analyses of COD, BOD, TKN, TP and heavy metals were conducted on sludges previously dried (at room temperature until a constant weight was obtained), and therefore the results are expressed on dry matter bases (kg TS). In leachate samples, in addition to most of the parameters measured in the sludges, nitrite (NO_2^-) and nitrate (NO_3^-) were also analysed.

Figure 4.3 Details of the Eijkelkamp soil coring kit and of one collected sample.

While solids and organic matter concentration of the sludge is needed to follow the evolution of the process and were measured in all three campaigns, other parameters like the concentration of nutrients, heavy metals and faecal bacteria indicators are useful to determine the quality of the final product for its use on land as an organic fertilizer. In practise, these parameters must only be analysed in the end of the cycle, once the sludge is going to be removed and disposed. Nevertheless, a characterisation was carried out especially during the first campaign, in order to obtain some initial figures.

ANOVA tests were conducted on TS, VS and COD data in order to study the statistical significance of the differences found between layers. Tests were carried out using Minitab 15.0.

Results and discussion

The results obtained in the 3 wetlands are shown and discussed together for each parameter considered.

Sludge height

The values of sludge height measured in each campaign are shown in Table 4.3. The variation of the sludge height in Seva is due to the slope of the bottom of the bed, which was not levelled during construction. Examination of the sludge cores in Alpens and Sant Boi de Lluçanès showed a somewhat blackish upper layer and a brownish bottom layer, suggesting higher mineralization degree in the bottom layer. Each of these two layers extended to the half of the total sludge height. In Seva the upper layer (10-45 cm) was brown and the bottom layer (35-70 cm) somewhat black. Note that samples taken at different depths (surface and bottom) matched quite well the two differently couloured layers detected in the three beds.

Table 4.3 Sludge height measured in each sampling campaign in the three systems.

	Sludge height (cm)		
	Campaign I	Campaign II	Campaign III
Alpens	7	12.5	7.5
Sant Boi de Lluçanès	30	30	15
Seva	45-100	60-105	40-85

The measure of sludge thickness in the beds enables to estimate the sludge height increasing rate, which dictates the lifespan of each cycle of filling and emptying. This rate is a key factor in order to evaluate and improve the performance of STW. Nielsen (2003, 2004) recommends a maximum rate of 10 cm/year, and from the three studied beds, only the bed of Alpens has a lower rate (7 cm/year). Considering this, and the nominal height available for sludge accumulation (Table 4.2), it is estimated that some additional 8 years of operation could be expected in this facility before emptying the beds. In Sant Boi de Lluçanès the height increasing rate (30 cm/year) is some 200% higher than the maximum recommended value, despite of having approximately the same sludge loading rates and patterns than Alpens. This high value of the rate is probably a result of a non-uniform distribution of the sludge between all the beds of this facility during the start-up period, due to operating problems of the feeding system. This resulted in some excessively loaded beds, in special the representative evaluated in the present study. Taking into account that those operating problems have already been solved, this trend must decrease in the years to come.

In the case of Seva, the observed high sludge height increasing rates (33 cm/year) are well related with the greater loading rates applied in this bed (125 kg TS/m^2·year). In this facility the beds filled up rapidly during the first operating cycle, and the sludge was removed after only 4 years of operation instead of the 8-12 years usually reported in the literature (Nielsen, 2003). Altogether these findings highlight the importance of the sludge loading rate, in order to maximize the lifespan of the operation cycles.

From the second (spring) to the third campaign (summer) it could be noticed that the conspicuous level of evapotranspiration in summer enhanced sludge drying and consequently volume reduction. Sludge layers were halved in Sant Boi de Lluçanès where sludge feeding was not regular due to operation problems. In the others systems the decrease in sludge height was lower but in any case important (around 1/3 sludge layer reduction). These patterns indicate that, due to environmental conditions, the sludge height changes within annual cycles increasing in winter and decreasing in summer. Therefore, for calculating the height increasing rate, it must be necessary at least consider a complete annual cycle.

pH and electrical conductivity

pH values were fairly similar in all sampling points in all field campaigns, ranging from 6.7 to 8 (Table 4.4).

Generally, the values of EC of the influent were higher than those observed in the sludge accumulated into the beds (especially in the second campaign where influent values of 3.8-6.7 dS/m decreased to 1.0-2.3 dS/m within the beds).

Table 4.4 pH and Electrical Conductivity (EC) of the sludge samples obtained in the three campaigns. Analyses were conducted on one replica. Note that EC data of all beds in the first campaign and influent data of Sant Boi the Lluçanès in the second campaign are not available.

		Campaign I	Campaign II		Campaign III	
		pH	pH	EC 1:5 (dS/m)	pH	EC 1:5 (dS/m)
Alpens	Influent	7.5	7.2	3.8	7.3	1.4
	Surface	7.9	7.4	2.3	7.5	0.8
	Bottom	7.6	7.3	1.9	6.7	0.6
Sant Boi de Lluçanès	Influent	7.4	-	-	-	-
	Surface	8.0	7.5	2	6.6	1.7
	Bottom	7.8	7.6	2.2	6.9	1.8
Seva	Influent	7.3	7.3	6.7	7.3	2.1
	Surface	7.3	7.2	1.1	6.9	0.5
	Bottom	7.2	8.2	1.0	7.9	0.7

Sludge dewatering

Sludge dewatering is evaluated by means the increase in TS. The results indicate that sludge dewatering follows the same trend in the beds of Alpens and Sant Boi de Lluçanès (Figure 4.4): there is a rise in the TS concentration from the influent sludge (1-3%) to the upper layer (7-26%) and to bottom layer (20-30%). In Seva a different pattern was observed because the TS decreased from the upper layer (18-23%) to the bottom layer (11-13%). In the end, the bed in Seva has a sludge with a lower moisture content than the other two beds. Bearing in mind that the bed in Seva has the highest sludge thickness and sludge loading rate, sludge dewatering is worsened by the higher values of both factors. Statistical analyses between upper and bottom layers in all beds indicate significant differences in all campaigns ($p<0.005$). In the same way, TS values between beds result to be significantly different.

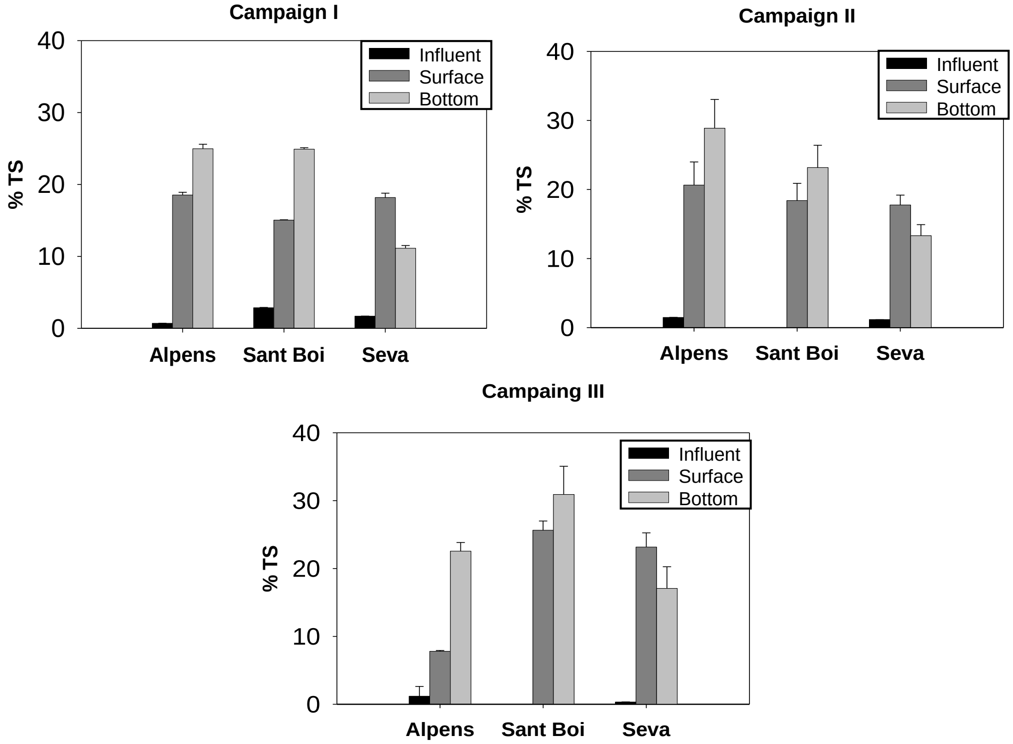


Figure 4.4 Average Total Solids (TS) and standard deviation of the sludge samples taken in three sampling campaigns. Note that influent data of Sant Boi the Lluçanès in the second and third campaigns are not available.

In the beds of Seva and Sant Boi de Lluçanès the higher TS were those of the third campaign (summer, when there is a greater evapotranspiration that enhances the drying capacity). Note that in Sant Boi de Lluçanès the TS values in this third campaign were probably increased due to operational problems that avoided regular feeding in the previous 3 months. It is interesting to see that the TS observed in the bed of Alpens in the third campaign were lower than in the other campaigns, in opposition to the trend observed for the other two systems. This is due to the fact that the system received a recent feeding before the samples were taken out, and as a result the moisture increased, especially in the surface layer.

Differences in TS content between bed layers are due to dewatering processes. After each feeding event, most of water content seems to be removed rather quickly from the surface layer due to evapotranspiration and percolation processes (water content decrease from 97-99% in the inlet to 82-85% within the surface layer). After that, the sludge seems to have a slow progressive dewatering with the time. Actually, the comparison of the upper and bottom sludge layers in the beds of Alpens and Sant Boi de Lluçanès suggests that remaining water is progressively eliminated during storage, as in depth sludge is accumulated for longer period. Thus, the final product is sludge with moisture contents around 70-80%, which fills in the range obtained with conventional technologies, like centrifuges and belt-filter presses (Metcalf and Eddy, 2003).

The lower TS concentration in the bottom layer with respect to the surface layer observed in the bed of Seva suggests a low percolation rate through the draining filter layer. In these type of systems, a deficient functioning in terms of hydraulic characteristics and excess of load may result in clogging of the granular media by accumulation of organic solids which are retained in the pores (Platzer and Mauch, 1997).

Organic matter removal

Together with sludge dewatering, sludge mineralization takes place during sludge storage, as indicated by lower VS, COD and BOD values observed in the beds than in the influent in the two first campaigns (Figures 4.5 and 4.6, Table 4.5).

In the two first sampling campaigns in the beds of Alpens and Sant Boi de Lluçanès the VS concentration decreased from 52-67% in the influent to 36-45% and 32-39% in the surface and the bottom layers, respectively (Figure 4.5). In Seva the VS concentration decreased from around 58-59% in the influent to 46-51% in the wetland (both layers). Note that only in the bed of Alpens the VS concentration was statistically different between layers ($p<0.05$). Considering the VS of the bottom layer, the VS percentage removal rates in these two first sampling campaigns were 13 to 31% in Alpens, 17% in Sant Boi de Lluçanès and 10 to 12% in Seva. Lower removal rates observed in Seva are in accordance with the lower dewatering capacity observed in this facility. Pempkowiak and Obarska-Pempkowiak (2002), and Obarska-Pempkowiak et al. (2003) reported similar organic matter decreasing trends between surface and bottom layers (about 7%).

Furthermore, from TS and VS concentration behaviour, as colour of the sludge layers, it seems that the high sludge height increasing rates observed in Seva have effects on process performance. In fact, in this facility the higher loading rates than those applied in the others systems and than those suggested in literature (Nielsen, 2003) seem to be related to its lower efficiency in terms of dewatering as well as stabilization.

The VS removal rates generally observed in this study are lower (up to 30%) than typical values reported for stabilization treatments like aerobic and anaerobic digestion (50 - 65%) or composting (Metcalf and Eddy, 2003; Bertrán et al., 2004). Nevertheless, the VS concentration in the sludge accumulated in the bottom layer of the beds in Alpens and Sant Boi de Lluçanès (near 40%) is rather similar to that obtained in anaerobically digested sludge, clearly indicating that STW promote sludge stabilisation, as well as sludge dewatering. This lower removal rates in comparison to conventional technologies could be related with the type of biological wastewater treatments used in the facilities studied, which consists of extended aeration or contact-stabilisation systems, leading to the production of a partially stabilised waste activated sludge (i.e. with already a low VS content, from 50 to 60% in our case). STW could be considered as passive form of composting because oxygen transfer from the plant rhizosphere to the sludge and surface aeration of the sludge related with the movement of the stems promote biological stabilization and mineralization of the sludge (Metcalf and Eddy, 2003).

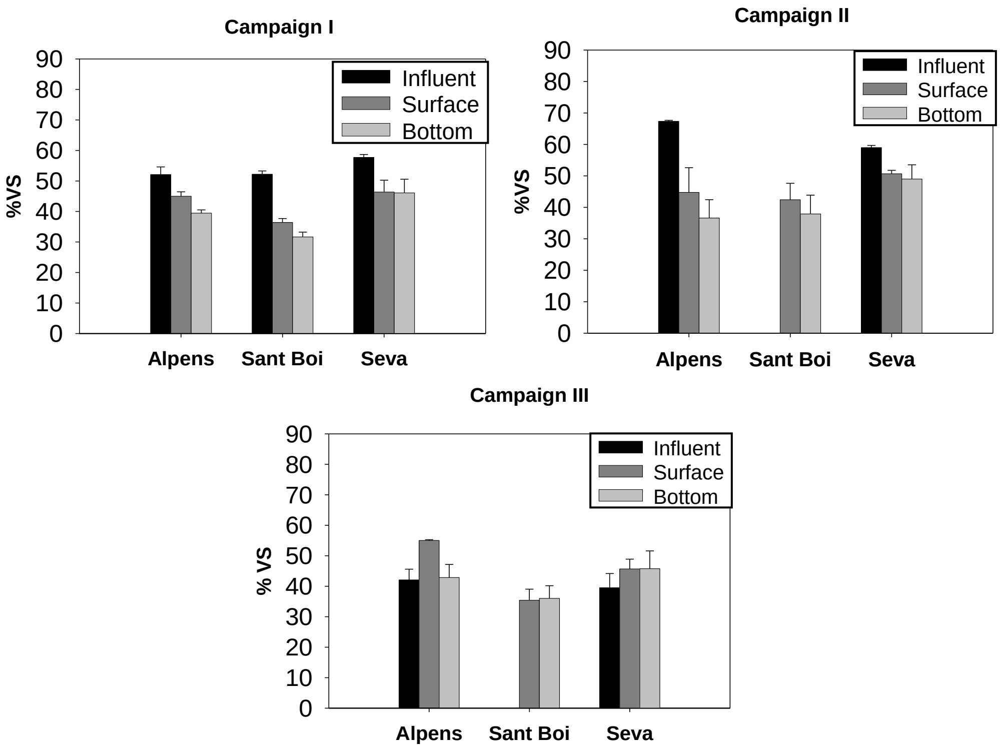


Figure 4.5 Average Volatile Solids (VS) and standard deviation of the sludge samples taken in three sampling campaigns. Note that influent data of Sant Boi the Lluçanès in the second and third campaigns are not available.

Similarly to the results observed for VS in Alpens and Sant Boi de Lluçanès, COD was slightly lower in all wetlands in the bottom layer than in the surface layer (Figure 4.6). Differences in COD between layers were statistically significant in all layers in the first campaign ($p<0.05$), but not in the second. The COD percentage removal rates were approximately 47% in Alpens and 30% in Seva in the second campaign (in Sant Boi de Lluçanès could not be calculated). Note that the COD percentage removal rates observed in Alpens and Seva were higher than the VS percentage removal rates in the same beds.

VS and COD values found in third campaign were generally in agreement with those of two first campaigns, except for the influent. In this case, organic matter content was lower in the influent than in the sludge stored in the beds. This could be due to the fact that the quality of the influent can change sometime depending on certain environmental and operational factors (i.e. the rain or an excessive purge causes dilution and therefore decrease of concentration).

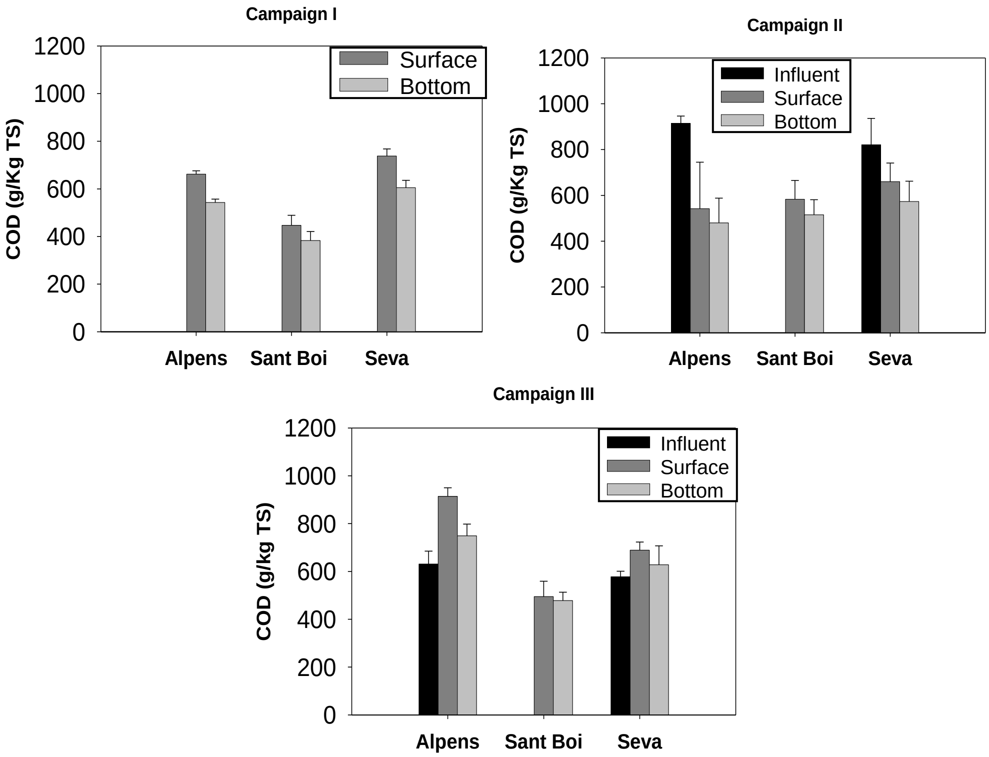


Figure 4.6 Average Chemical Oxygen Demand (COD) and standard deviation of the sludge samples staken in the three sampling campaigns. Note that influent data of the first campaign and influent data of Sant Boi de Lluçanès in the second and the third campaigns are not available.

BOD was measured in the two first campaigns and the results had the same general pattern observed for VS and COD in all beds (Table 4.5). BOD_5 values of all samples were around 100 times (or more) lower than COD. BOD_{21} values were also very low in comparison to COD, approximately from 50 to 100 times lower. Altogether these results indicate that the organic matter is very refractory.

Table 4.5 Biochemical oxygen demand (BOD_5, BOD_{21}) of the sludge samples obtained during the first and second campaigns. Note that influent data of Sant Boi the Lluçanès in the second campaign are not available. Results are expressed in gO_2/kg TS.

		Campaign I		Campaign II	
		BOD_5	BOD_{21}	BOD_5	BOD_{21}
Alpens	Influent	6.8	20.1	3.7	10.9
	Surface	5.5	13.6	1.6	5.1
	Bottom	3.3	8.5	1.0	3.2
Sant Boi de Lluçanès	Influent	6.9	17.4	-	-
	Surface	5.5	14.4	3.6	6.2
	Bottom	3.7	10.0	2.7	4.6
Seva	Influent	3.4	9.7	3.4	10.9
	Surface	2.9	3.4	1.3	5
	Bottom	3.3	4.5	1.9	5.7

Nutrient concentration

TKN decreased from the influent to the sludge accumulated in the all beds (Table 4.6). In general, little differences in TKN are observed between the bottom and upper layers. In Alpens and Sant Boi de Lluçanès the TKN concentration was higher in the surface layer than in the bottom layer, while in Seva the opposite trend was observed. In Alpens the TP concentration in the influent (0.72%) was much higher than in the bed (0.07-0.11%). The TP concentration in the influent of Seva (0.07%) was much lower than that of Alpens.

Stabilized sludges can be used as organic fertilizers and soil conditioners, which constitute major disposal routes for this type of wastes. In general, the high organic content in sludges, together with a certain amount of nutrients, makes them suitable for agricultural application, provided that heavy metals concentrations fulfil current legislation (Council Directive 86/278/EEC).

Nutrient concentrations are not regulated, unless the sludge is to be sold as organic fertilizer following, for example, a composting treatment (Regulation (CE) No. 2003/2003). However, the concentration of nutrients is needed to ensure appropriate dosages of the sludge, and therefore TP and TKN were analysed in this study. In general TKN seems to be transformed in the systems. In fact, high TKN nitrification rates are confirmed by the very high nitrate concentration in the leachate (up to 2500 mg/L, see below). All this makes evident the necessity for sludge characterisation in the end of the operating cycle, prior to land application of the sludge. For instance, phosphorus doses required for agriculture application are strictly dependent on the fertilizer and soil characteristics (Pomares and Canet, 2001).

Table 4.6 Concentration of Total Kjehldahl Nitrogen (TKN), Total Phosphorus (TP)) of the sludge samples obtained in the first and second campaigns. In the first campaign TP was not analysed. In the second campaign TKN was analysed from one replica. Note that influent data of Sant Boi the Lluçanès in the second campaign are not available.

		Campaign I	Campaign II	
		TKN (%ST)	TKN (%ST)	TP (%ST)
Alpens	Influent	5.85 ± 0.018	6.83	-
	Surface	3.77 ± 0.012	3.77	0.07 ± 0.05
	Bottom	2.72 ± 0.007	2.72	0.11 ± 0.11
Sant Boi de Lluçanès	Influent	4.66 ± 0.05	-	-
	Surface	3.08 ± 0.008	3.48	0.04 ± 0.01
	Bottom	2.13 ± 0.014	2.67	0.10 ± 0.08
Seva	Influent	5.19±0.002	5.08	0.07 ± 0.01
	Surface	3.75 ± 0.012	3.37	0.18 ± 0.11
	Bottom	4.99 ± 0.019	4.33	0.10 ± 0.03

Heavy metals concentration

Since heavy metals content may limit or forbid land application of an organic fertilizer, in this study they were analysed in order to verify potential future agriculture application of final sludge. The results obtained for the heavy metals analyses are shown in Table 4.7. In the beds of Alpens and Seva, there seems to be an accumulation tendency, since the values are in general higher in samples of the bed compared to those of the influent sludge. In the case of Sant Boi de Lluçanès, however, there doesn't seem to be a clear trend.

Thus, some metals like Cr and Hg seem to accumulate in the bed, while others seem to decrease during sludge depot. It is clear, though, that heavy metals concentrations of all samples are well below the thresholds set by European Sewage Sludge Directive (Council Directive 86/278/EEC), as well as the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000) for land application. At the same time, the observed values are in the same range found in previous studies on STW applied in urban wastewater treatment plants (Kim and Smith, 1997).

Table 4.7 Concentration of heavy metals of the sludge samples obtained in the first sampling campaign. Limit values proposed in the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000) are also shown.

		Cr (ppm)	Ni (ppm)	Cu (ppm)	Zn (ppm)	Cd (ppm)	Hg (ppm)	Pb (ppm)
Alpens	Influent	35.8	27.9	227	348	0.41	4.29	30.3
	Surface	54.8	29.2	392	537	0.61	5.67	49.3
	Bottom	56.0	29.6	392	565	0.65	4.87	54.3
Sant Boi de Lluçanès	Influent	36.4	50.2	183	609	0.66	1.99	51.0
	Surface	45.6	36.0	174	568	0.73	1.44	46.0
	Bottom	42.4	29.0	120	425	0.54	3.34	33.2
Seva	Influent	52.1	25	232	897	0.76	0.95	60.1
	Surface	62.0	44.7	245	789.7	0.9	3.3	80.9
	Bottom	54.3	37.1	229.7	622.2	1.0	3.2	79.3
Limit values		800	200	800	2000	5	5	500

Although some heavy metals accumulation might be expected in the long term operation of the systems, no such pattern was observed during the field campaigns, hence exceeding the legal limits is not forecasted for the facilities studied.

Faecal bacteria indicators concentration

Faecal bacteria indicators were analysed in particular to assess the quality of the final product for land application. According to the results (Table 4.8), *Salmonella* spp. was not detected in all samples. *E. coli* was present in all cases, decreasing from the influent sludge to the sludge accumulated in the bed of Alpens, while the opposite trend was observed in Sant Boi de Lluçanès. Very low numbers of *E. coli* were found in the bed of Seva.

Sludge hygienisation prior to land application is not yet regulated, but it is likely to be in the near future, as proposed for example in the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000). The results of hygienisation show some differing trends between the studied facilities. In general, though it might be speculated that sludge from the wetlands in Alpens and Seva would fulfil the limits, further analysis would be required to evaluate hygienisation effects in the long term.

Table 4.8 Concentration of faecal bacteria indicators of the sludge samples obtained in the first sampling campaign. Analyses were conducted on one replica.

		Salmonella* spp. (presence/absence in 25g)	***E. coli* (MPN/g)***
Alpens	Inlet	Absence	1100
	Surface	Absence	460
	Bottom	Absence	460
Sant Boi de Lluçanès	Inlet	Absence	1100
	Surface	Absence	1100
	Bottom	Absence	>2400
Seva	Inlet	Absence	4
	Surface	Absence	<3
	Bottom	Absence	<3

*3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000) proposes that sludge shall not contain *Salmonella* spp. in 50 g and that *E.coli* concentration has to be less that 500 MPN/g.

Leachate quality

Grab leachate samples were analysed in all facilities during the second campaign and the results are shown in Table 4.9. EC values were rather similar to those found in sludge accumulated in the beds; only in Seva the value was clearly higher. Total Suspended Solids (TSS) and COD values were very low, in fact TSS were not detected. TKN and TP also had rather low values, while nitrate had a very high concentration.

With respect the organic matter results of the leachate, the undetected values of solids and the very low COD values indicate a good quality of water released by the systems. In spite the high nitrate concentration in the leachate, the recycling to the WWTP does not constitute an especial problem because of its small flow, which has been estimated by the plant managers in around 1/6 of beds influent water. Moreover, the performance of the WWTP is adequate in terms of solids and organic matter removal.

In addition, nitrification detected by decrease of TKN values in the sludge and by the high nitrate concentration within the leachate, confirm the hypothesis that the studied wetlands operate under aerobic conditions.

Table 4.9 Physico-chemical properties of the leachate of each facility. All samples collected during the second sampling campaign. nd means not detected.

	Alpens	Sant Boi de Lluçanès	Seva
pH	7.0	7.9	8.3
EC 1:5 (dS/m)	3.0	1.0	7.5
TSS (mg/L)	nd	nd	nd
COD (mgO_2/L)	72 ± 5.5	94 ± 3.7	59 ± 3.7
TP (mg P/L)	16	25	7.38
TKN (mg N/L)	<10	<10	<10
NO_2^- (mg N/L)	0.112 ± 0.007	0.228 ± 0.007	0.203 ±0.002
NO_3^- (mg N/L)	710 ± 57	2500 ± 22	280 ± 17

Conclusions

In this work it was studied the performance of three STW by analysing samples from the influent sludge and sludge accumulated on the top and bottom layers of the beds. Important changes from the influent sludge to the accumulated sludge in most of the studied parameters were observed

Moisture content of the influent sludge was reduced by 20-27% in Alpens, 26-30% in Sant Boi de Lluçanès and 16-23% in Seva. Therefore, in terms of sludge dewatering, which is one of the main interests of STW technology, all studied systems are capable of achieving efficiencies similar to those of conventional technologies.

Progressive organic matter decrease was observed from the influent sludge to the upper and bottom sludge layers in the wetlands. Considering the first two campaigns, organic matter, expressed as VS, was reduced from 52-67% to 30-49% TS, and similarly occurred with BOD and COD. The results suggest progressive sludge stabilization and mineralization with time, and might be considered as additional advantages of this technology.

The results of nutrients, heavy metals and faecal bacteria suggest that the final product from the treatment, because of its stabilization, may be used as a fertilizer in agriculture.

On the whole, the studied systems demonstrate the good efficiency of STW technology and its potential applicability in the context of small and remote WWTP in Mediterranean regions.

5. Sludge dewatering and mineralisation in a pilot plant with different design configurations

This chapter is based on the article:

E. Uggetti, I. Ferrer, J. Carretero, J. García (2011). Sludge dewatering and mineralisation in a sludge treatment wetlands pilot plant with different design configurations. In preparation

Among the design factors of sludge treatment wetlands (STW), plant species and granular medium play an important role in the treatment process. The objective of this study was to evaluate, along 2 years of experimentation in a pilot scale experiment, the efficiency of three system's configurations with different plant species (*Phragmites australis* and *Typha* sp.) and filter media (gravel or wood shavings). Sludge dewatering and stabilisation were monitored along the sludge treatment; moreover agricultural suitability of biosolids was investigated. At the end of the treatment sludge volume was reduced of about 80%, TS concentration increased up to 16-24%TS and VS was generally reduced to ~50%VS/TS. Agricultural applicability of biosolids was suggested by the absence of phytotoxicity (Germination index >100%) and the low heavy metals and pathogens concentrations (absence of *Salmonella* an *E.coli*<240MNP/g. No significant differences in treatment efficiency were found between treatment configurations, suggesting the suitability of all the configurations tested.

Introduction

Sludge treatment wetlands (STW) consist of shallow tanks (beds) filled with a gravel layer and planted with emergent rooted wetland plants. In these systems, secondary sludge is usually pumped and spread on the wetland's surface. Here, part of the sludge water content is rapidly drained by gravity through the gravel layer; while another part is evapotranspirated by plants. In this way, a concentrated sludge residue remains on the surface of the bed where, after some days without feeding (resting time), thickened sludge is anew spread, starting the following feeding cycle. During feeding periods, the sludge layer height increases at a certain rate (around 10 cm/year). When the layer approaches to the top of the tank, feeding is stopped during a final resting period (from 1–2 months to 1 year), aimed at improving sludge dryness and mineralisation. The final product is subsequently withdrawn, starting the following operating cycle.

Changes of sludge composition in time are a result from dewatering processes (draining and evapotranspiration) as well as the degradation of organic matter (Nielsen, 2003b). Thus, besides dewatering, STW also allow for a certain degree of sludge mineralization. The resulting final product is suitable for direct land application (Nielsen and Willoughby, 2005), or might be further treated to improve sludge hygienisation (Zwara and Obarska-Pempkowiak, 2000), promoting in either case sludge reuse as opposite to sludge disposal in landfill or sludge incineration.

Among design factors, plant species and granular medium play an important role in the treatment process, due to their contribution to water evapotranspiration and water percolation, respectively.

In fact, plants are a key element of STW, since they assist sludge dewatering and mineralisation. According to Edwards et al. (2001) planted beds showed higher TS concentration (20-21%) than unplanted (18%) and higher slugde height reduction (84-86% in planted beds and 81% in unplanted). Plant species used in treatment wetlands have to be able to grow in watery, muddy, anaerobic conditions and at the same time they must be able to tolerate changes in water level, high salinity and variations between high and low pH (De Maeseneer et al., 1997). The most widely used species in treatment wetlands for wastewater as well as sludge treatment is the common reed (*Phragmites australis*) (Puigagut et al., 2007). Hardej and Ozimek (2002) evaluated the effect of sewage sludge on growth and morphometric parameters of Phragmites australis and demonstrated the high adaptation capacity of the common reed to the sewage sludge environment, observing that the shoot density was over two times greater than that commonly found in natural systems. In spite of the extensive use of cattail (*Typha* sp.) in wastewater treatment wetlands; only few experiences with this plant species are reported in literature (Koottatep et al., 2001; Magri et al., 2010).

On the other hand, the granular medium constitutes a filter for water percolation with a total height ranging from 30 cm to 50–60 cm (Uggetti et al., 2010). The filter has several layers of granular media set in increasing size from top to bottom, through which water percolates. Leachate is collected by means of draining pipes, which are located at the bottom of the granular medium. While stones (diameter of around 5 cm) at the bottom protect draining pipes, gravel (diameter from 2 to 10 mm) and sand (diameter from 0.5 to 1 mm) in the upper layers provide a primary physical filtration and rooting medium for plants at the beginning of system operation (Begg et al., 2001; Edwards et al., 2001).

Most studies on full-scale STW show excellent results in many European countries and in US (Uggetti et al., 2009). Several pilot plant trials have been carried out in Palestine (Nassar et al., 2006), Cameroon (Kengne Noumsi et al., 2006) and, more recently, in Greece (Stefanakis et al., 2009; Melidis et al., 2010), France (Vincent et al., 2010), China (Yubo et al., 2008), Thailand (Koottatep et al., 2005; Panuvatvanich et al., 2009) and Brazil (Magri et al., 2010). However, despite of the favourable climates of the Mediterranean region, only few pilot scale studies have been recently carried out in Greece and Italy (Stefanakis et al., 2009; Bianchi et al., 2010).

The objective of this study was to evaluate, in a pilot scale experiment, the effect of two design factors (plant species and type of granular medium) on STW performance in terms of sludge dewatering, mineralisation and hygienisation. This study aimed at gaining knowledge on the implementation and performance of STW in Mediterranean Region.

Materials and methods

Experimental set up

The experiments took place outdoors at the experimental facility located on the roof of the Department of Hydraulic, Maritime and Environmental Engineering of the Technical University of Catalonia, Barcelona, Spain.

The pilot plant was set up in winter 2008 and consists of three PVC containers with surface area of $1m^2$ each and height 1m. During the first 3 months (commissioning phase) STW were fed with wastewater in order to promote plants growth. Afterwards, from May 2009 to March 2011, thickened activated sludge produced at the WWTP of the municipality of Vilanova del Valles, Barcelona, Spain, was manually fed one per week. The WWTP treats wastewater from 10,000 PE in an extended aeration system. The sludge loading rate was around 20 $kgTS/m^2 \cdot year$ during the first 8 months of operation, afterwardrd it was set to approximately to 40 $kgTS/m^2 \cdot year$ for each STW (corresponding to 0.025 m^3/week).

Three different STW configurations were investigated in this study (Figure 5.1):

- STW 1 was planted with common reeds (*Phragmites australis*). The drainage filter was constituted, starting from the bottom, by a 10 cm layer of stones (d_{50}= 250 mm), 30 cm of gravel (d_{50}= 5 mm) and 10 cm of sand (d_{50}= 1 mm).
- STW 2 was planted with cattails (*Typha* sp.). The drainage filter was the same as in STW 1.
- STW 3 was planted with cattails (*Typha* sp.). The drainage filter was constituted, starting from the bottom, by a 10 cm layer of stones (d_{50}= 250 mm), 30 cm of wood shavings and 10 cm of sand (d_{50}=1 mm).

At the bottom of each STW, three perforated PVC pipes were positioned in order to collect leachate and to assess filter oxygenation.

Figure 5.1 Detail of the pilot plant (March 2011)

Experimental procedure

According to the sludge feeding, plant operation can be divided into two periods or operation stages: the sludge feeding period, corresponding to the 22 months during which sludge was weekly fed, and the final resting period corresponding the 2 month after the last feeding. As described below, for each treatment period different parameters were analysed in order to study sludge and biosolids characteristics.

Moreover, since March 2010, sludge height, or thickness, inside the beds was also measured weekly before each loading. Meteorological data were gathered by a municipal meteorological station located near the University (www.meteocat.com).

Wetlands' sludge (sludge feeding period)

Five sampling campaigns were carried out, from November 2009 to March 2011, every 4 months. Each campaign was carried out one week after loading. In order to obtain representative composite samples, sludge was collected from 6 points in each bed and subsequently mixed. Feeding sludge was also characterised during the whole experimentation period.

During this period only few characterisation parameters were analysed, in order to monitor sludge dewatering and stabilisation processes while sludge treatment is taking place. In each sample pH, Electrical Conductivity (EC), Total and Volatile Solids (TS and VS), Chemical Oxygen Demand (COD) were measured in all campaigns in order to follow the evolution of the dewatering and mineralization processes within the beds.

Biosolids (final resting period)

After the last sludge loading, biosolids resulting from the sludge treatment were sampled once per month from March to May 2011 (3 campaigns in total corresponding to 3 months of final resting period). Moreover, biosolids were characterised for pH, Electrical Conductivity (EC), Total and Volatile Solids (TS and VS), Chemical Oxygen Demand (COD), Total Kjehldahl Nitrogen (TKN) and Total Phosphorous (TP). Moreover germination index (GI), faecal bacteria indicators (*Salmonella* spp. and *Escherichia coli*) and heavy metals were determined at the beginning and at the end of the final resting period (March and May 2011).

The monitoring of these parameters during the final resting period (without loading) was useful for the determination of the most appropriate resting period in order to obtain a dry and stabilised product suitable for agricultural application.

Analytical methods

Samples, collected using Eijkelkamp soil coring kit, were analysed (generally in triplicate) using conventional methods following the procedures indicated in the Standard Methods (APHA-AWWA-WPCF, 2001). pH, Electrical Conductivity (EC) of wetlands' sludge were analysed on dried samples diluted in distilled water with a 1:5 ratio. Analyses of COD, TKN, TP and heavy metals were conducted on sludge previously dried (at room temperature until a constant weight was obtained), and therefore the results are expressed on dry matter bases (kg TS).

For the determination of the Germination index (GI) cucumber (*Cucumis sativus*) and lettuce (*Lactuca sativa*) seed were chosen in accordance with Barrena et al. (2009b). Aqueous extracts of the wetlands' sludge were prepared by shaking the fresh sample with distilled water 1:10 w/v and then filtered. After 7 days of incubation at 20°C in the dark, the seed germination, root elongation, and germination index (GI) were determined according to Tiquia et al. (1996).

ANOVA tests were conducted on TS, VS, COD and GI data in order to study the statistical significance of the differences found between the different configurations of the pilot plant. Tests were carried out using Minitab 15.0.

Results

Results are presented in different sections; note that results from sampling campaigns during the system's feeding are described separately from the biosolids' characterisation after the resting period.

Changes in sludge height

Figure 5.2 shows the variations of the sludge layer height during the entire experimentation period. Moreover, rainfalls, air temperature, solar radiation of the period corresponding to the sludge feeding (from March 2010 to March 2011) are illustrated here.

Sludge height of the three beds follows approximately the same pattern. Greatest heights (around 30 cm), corresponding to a lower dewatering degree, were recorded in winter (between January and March) as a consequence of the reduced contribution of the evapotranspiration to the sludge dewatering. In spring, in correspondence with the increasing of the air temperature, sludge height decrease rapidly reaching the lowest values (17 cm) in July. This behaviour if the sludge layer height highlights the importance of the climate conditions in sludge dewatering. In fact, Figure 5.2 shows that sludge height evolution is clearly opposite to the temperature and solar radiation trend, which influence the evapotranspiration rate. Thus, this results confirm the importance of the evapotranspiration rate in sludge dewatering and consequently in volume reduction.

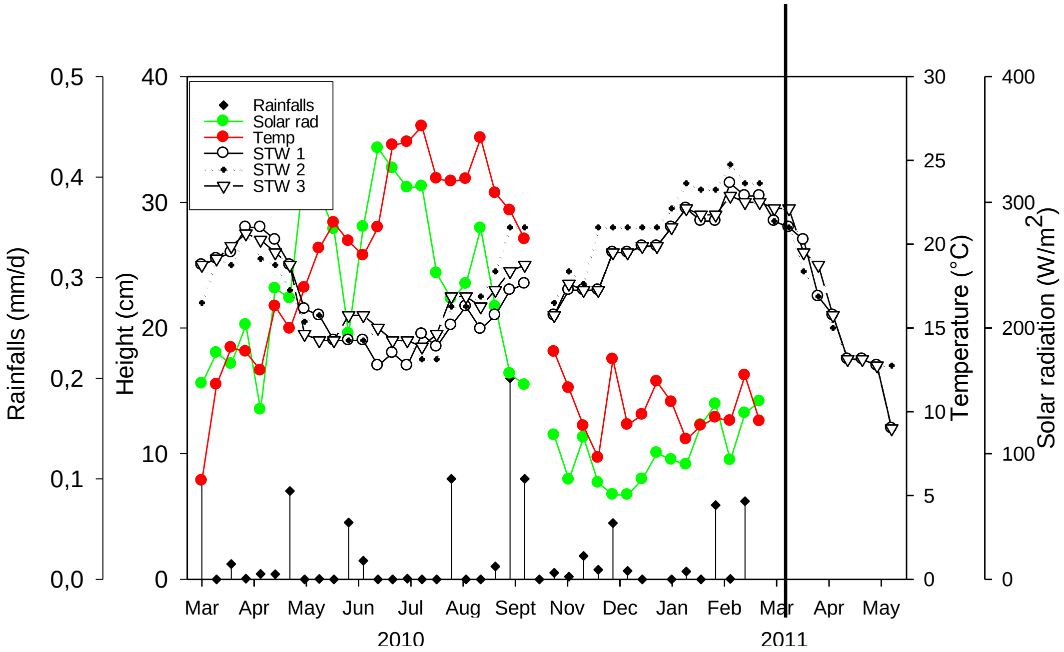


Figure 5.2. Sludge layer height together with rainfalls, air temperature and solar radiation during sludge feeding and resting period. The vertical line corresponds to the last sludge loading.

Observing the differences between years, the sludge height in March 2010 was around 25 cm in all the beds, while in March 2011 sludge height was comparatively increased of about 5 cm, reaching 30 cm in all the beds. According to Nielsen (2003), sludge loading rate of 60 kgTS/m^2·y should correspond to a sludge layer height increase of approximately 10 cm/year.

In general during the feeding period the no significant differences were detected in terms of sludge height between the STW. However, STW 2 (planted with *Typha* sp.) was characterized by slightly higher heights in winter, respect to the other beds. The same behaviour was observed during the final resting period.

Sludge feeding period

As shown in Table 5.1 pH values are rather constant. STW did not influence significantly pH, which varied in the neutral range from 6.8 to 8.5 in the wetland sludge. Influent EC values changed around 5,0 dS/m, while EC within wetlands values were clearly lower ranging between 1000 and 2,0 dS/m (Table 5.1). This is related to the dilution of wetland's sludge (1:5) which is responsible of the lower values found in these samples. Only in autumn, wetlands' EC values were higher (around 6000 µS/cm) and similar to the influent values. No differences between STW were detected.

Table 5.1. pH and EC results from the sampling campaigns carried out during sludge feeding. EC within wetlands were analysed on dried samples diluted in distilled water with a 1:5 ratio.

		pH (1:5)	EC (dS/m)
Campaign I November 2009	Feeding	7.2	6,0
	STW 1	7.7	1,4
	STW 2	7.9	1,5
	STW 3	7.7	1,4
Campaign II March 2010	Feeding	6.5	5,5
	STW 1	7.0	1,1
	STW 2	7.5	1,2
	STW 3	7.1	1,1
Campaign III July 2010	Feeding	7.4	6,1
	STW 1	6.8	6,0
	STW 2	6.9	5,9
	STW 3	7.0	6,7
Campaign IV November 2010	Feeding	7.0	4,6
	STW 1	8.1	2,1
	STW 2	8.3	2,1
	STW 3	8.5	2,2
Campaign V March 2011	Feeding	-	-
	STW 1	7.6	1,3
	STW 2	7.7	1,7
	STW 3	7.6	1,7

Concerning sludge dewatering along the feeding periods, the 5 campaigns carried out highlight the seasonal pattern in terms of TS (Figure 5.3). Influent TS were almost constant along the experimentation with most of the concentrations around 2%. During the first campaign, carried out in autumn 2009, TS values were around 11-13% within all the STW. Similar values were found during the second campaign, in spring 2010. Such low dewatering performances are probably due to the short time of operation elapsed since the first sludge feeding (May 2009).

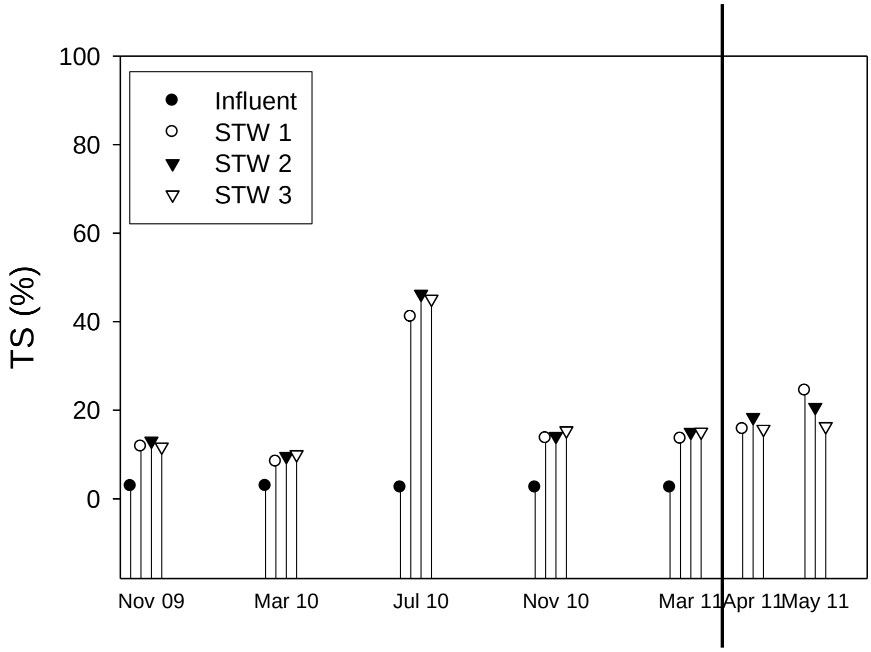


Figure 5.3. Total solids concentration measured within the influent sludge and the three wetlands during the feeding and resting period. The vertical line corresponds to the last sludge loading.

Much better results were found in summer, when TS concentration increased up to 41-46%. In this campaign, higher standards deviations were found due to the higher difference in TS concentration within the sludge layer. The high solar radiation of this period (Figure 5.3) enhanced dewatering of the upper layer which results significantly dryer than the bottom.

In the last two campaigns (November 2010 and March 2011), dewatering performance were quite similar, TS concentration within wetlands vary between 13-15% in autumn and spring. Results are in accordance to the sludge height recorded (around 30 cm in winter 2010). The decreasing in TS concentration should be caused by the lower temperature (Figure 5.3) and the limited evapotranspiration typical of this season.

With regards to the organic matter, both VS and COD were measured during the 5 campaigns. VS concentration in the influent ranged between 60 and 72% VS/TS (Figure 5.4), a certain VS reduction is observable in all campaigns. In some campaigns (November 2009 and July 2010) VS decrease is more evident. In November 2009, the relative low VS concentration (around 50%VS/TS) can be due to the lower sludge loading rate applied during the first 8 months of operation. Moreover, in July the high organic matter mineralization should be related with the high temperature, which enhances microoganisms' activity. In fact, in this campaign, a VS decrease of about 10 % was detected (from 65 to 52%VS/TS). On the other hand, in winter (March and November 2010), only 2% of VS reduction was observed and consequently VS concentration was higher than 55%VS/TS. As for TS, small variations were detected between wetlands' configuration in all campaigns.

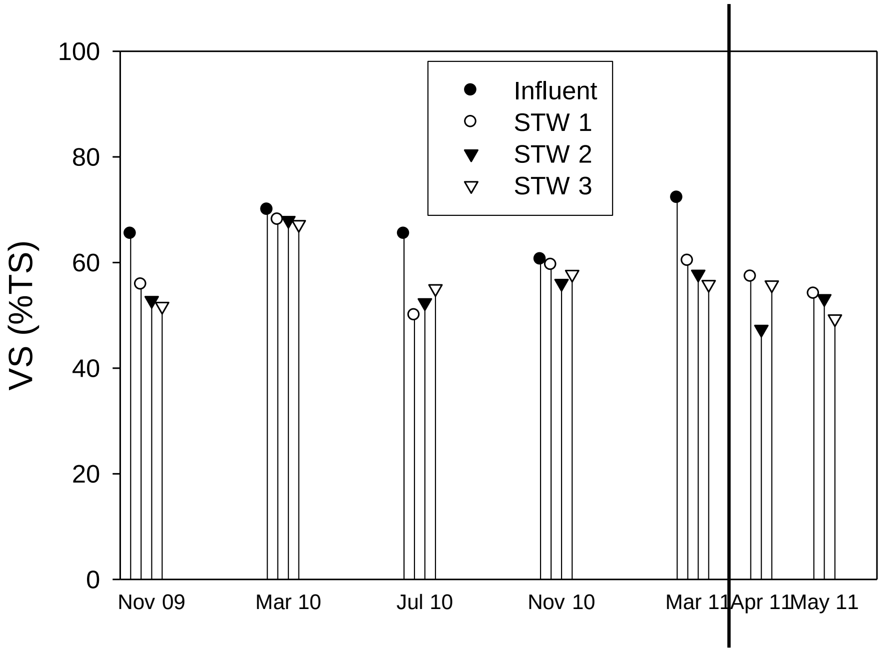


Figure 5.4. Volatile solids concentration measured within the influent sludge and the four wetlands during the feeding and resting period. The vertical line corresponds to the last sludge loading.

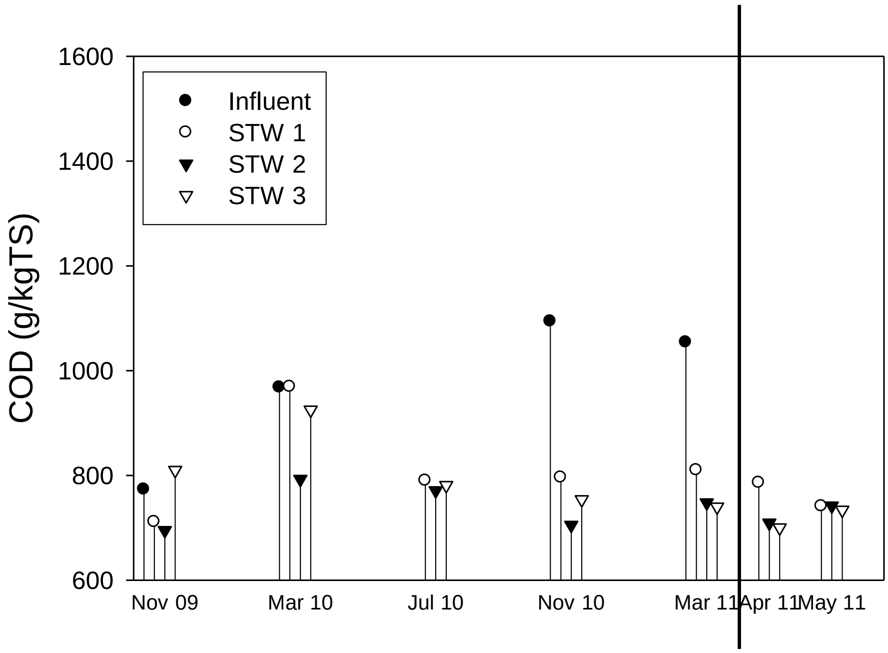


Figure 5. Chemical oxygen demand measured within the influent sludge and the four wetlands during the feeding and resting period. The vertical line corresponds to the last sludge loading.

Similar to VS, COD is a measure of the organic matter concentration, which indicates the sludge mineralisation during sludge treatment. A certain COD reduction was detected in each campaign (Figure 5.5), mainly in the two last ones, where higher influent values (around 1000 g/kgTS) decreased up to 705 g/kgTS. Here as well, no significant differences between STW configurations were observed. In the whole, it seems that higher and more variable COD values (from 711 up to 969 g/kgTS) within STW during the first 2 campaigns tend to stabilise around 770 g/kgTS in the last three campaigns.

Final resting period

After the last loading, sampling campaigns were carried out monthly in order to monitor dewatering and mineralisation processes during this period. Moreover parameters relevant for agricultural reuse of biosolids were investigated.

After the last sludge loading (in March 2011) all the beds had a sharp decreasing in sludge height (Figure 5.2). After only 2 months without loading, sludge height was reduced of 11 cm in the STW 2 and 16-17 cm in STW 1 and 3.

Table 5.2. pH and EC results from the sampling campaigns carried out during the final resting time. EC within wetlands were analysed on dried samples diluted in distilled water with a 1:5 ratio.

		pH (1:5)	EC (µS/cm)
March 2011	STW 1	7.6	1,335
	STW 2	7.7	1,685
	STW 3	7.6	1,665
April 2011	STW 1	7.1	2,480
	STW 2	8.1	2,510
	STW 3	7.2	2,030
May 2011	STW 1	7.2	3,720
	STW 2	7.2	3,360
	STW 3	7.4	2,290

pH and CE values are shown in Table 5.2; pH values during the resting period are in accordance with those found during feeding period, varying between 7.2 and 8.1. Over the resting period, EC values increased from 1,3-1,7 dS/m after the last loading up to 2,290-3,720 µS/cm after 2 months. This is related to water loss. In fact, even higher increase of EC was found in July 2010, in correspondence with really high TS concentrations. No differences between STW are detected.

Concerning sludge dewatering, the decreasing in sludge height during the resting period is confirmed by the increasing in TS concentration (Figure 5.3). In April, after only 1 month of resting, TS concentration was enhanced only by around 2%. This can be attributed to the time elapsed from the last loading, and partially to the higher temperature of May with respect to April. Higher increase in TS concentration was detected in May 2011, two months after the last loading, when TS were varying from 24% in STW 1, to 20% and 16% in STW 2 and 3, respectively. On the whole, TS concentrations from March to May were increased 11% in STW 1, 5% in STW 2 and 1% STW 3.

As expected, the resting time was characterised by a certain sludge mineralisation as indicated by the volatile solids reduction (Figure 5.4). In March, VS concentration was varying between 69 %VS/TS in STW 1, 57 %VS/TS in STW 2 and 55 %VS/TS in STW 3. After 2 resting months, only 4 to 6 % of the VS were reduced in all the beds. Thus, from our results it seems that the most important part of the organic matter stabilisation take place already during the normal operation of the system, and not during the last resting months. This is confirmed by the COD concentration (Figure 5.5). In fact, COD was reduced only about 3-5% during the first resting months. After two months of resting, COD concentrations is around 740g/kgTS, without large differences between beds.

Table 5.3. Nutrients (TKN and TP) concentration in sludge during the resting period.

		TKN	TP
March 2011	STW 1	6.19	0.57
	STW 2	4.26	0.29
	STW 3	4.35	0.25
April 2011	STW 1	4.85	0.01
	STW 2	3.65	0.01
	STW 3	4.45	0.01
May 2011	STW 1	4.31	0.00
	STW 2	3.84	0.01
	STW 3	4.45	0.01

Concerning nutrients (Table 5.3), TKN concentration after the last sludge loading (March 2011) varies between 6.19 %TKN/TS in STW 1 and 4.35 %TKN/TS in STW 2. In the following campaigns, after 2 month without feeding, the nitrogen concentration within sludge was slightly lower, ranging between 3.84 %TKN/TS and 4.31%TKN/TS. However, differences are not significant. On the other hand, TP concentrations were significantly lower (0.25-0.57 %TP/TS) already in March, even lower values were detected in April and May, when concentrations decrease to 0.01 %TP/TS.

The results of the germination tests indicated that there was no phytotoxicity in the sludge of the SWT after the last loading. The same conclusion was reached with samples of the sludge after two months of resting. The relative seed germination, after 7 days of incubation, was almost always higher than 100%, suggesting that sludge has no toxic effects limiting seeds germination (Table 5.4). In a general way lettuce showed higher values (from 110% to 162%) than cucumber (between 83% and 146%). Regarding at the root growth, significantly (p=0.03) higher values were found for lettuce (98% to 187%) than for cucumber (95% to 127%), however the root elongation was in all the samples near to 100% or ever higher, overall during the second campaign. Consequently, the germination index results in values ranging between 94 and 224% for lettuce and from 102 to 146% for cucumber. As for many other parameters, no significant differences were found comparing the results obtained from the three different wetlands (p>0.05).

Table 5.4. Values of the Germination index in the two campaign during the resting period.

		Relative seed germination (%)		Relative root growth (%)		Germination Index (%)	
		Lettuce	Cucumber	Lettuce	Cucumber	Lettuce	Cucumber
March 2011	STW 1	140	108	98	95	138	102
	STW 2	162	140	124	95	201	133
	STW 3	153	146	103	98	158	144
May 2011	STW 1	126	100	152	94	193	94
	STW 2	120	93	187	108	224	100
	STW 3	110	83	164	127	181	106

Finally, regarding at the heavy metals, the concentrations were clearly below the law limits already after the last loading. This is related to the fact that the influent, coming from an urban WWTP, probably had low concentration of heavy metals (Table 5.5). Regarding pathogens (Table 5.5), no *Salmonella* was detected within the samples. In spite of the rather high concentrations (from 3,500 to 3,900 MPN/g) of *E.coli* found in all the beds after the last feeding, concetrations after two resting months decreased below the the limits proposed by the 3rd Draft Working Document (Environment DG, EU, 2000) still not in force (E.coli<500 MNP/g).

Table 5. Heavy metals (mg/kgTS) and Salmonella (absence/presence in 50 g) E.coli (MPN/g).

		Cd	Cu	Hg	Ni	Pb	Zn	*Salmonella*	*E.coli*
March 2011	STW 1	<0.5	138	<0.5	99	49	829	Absence	6,600
	STW 2	<0.5	75	<0.5	50	31	643	Absence	4,500
	STW 3	<0.5	103	<0.5	71	26	517	Absence	3,900
May 2011	STW 1	-	-	-	-	-	-	Absence	93
	STW 2	-	-	-	-	-	-	Absence	240
	STW 3	-	-	-	-	-	-	Absence	150
Directive 86/278/EEC		20-40	1,000-1,750	16-25	300-400	750-1,200	2,500-4,000	-	-
3rd Draft Working Document		10	1,000	10	300	750	2,500	absence	<500

Discussion

Over the two years of operation, each wetland was fed with approximately 1.675 m^3 of sludge. The volume of sludge stored in each STW after the last sludge loading was around 0.35 m^3 in STW 1, 0.30 m^3 in STW 2 and 0.31 m^3 in STW 3, corresponding to a sludge volume reduction of 79%, 82% and 81%, respectively. This results indicate lack of differences in dewatering performances between the tested design factors at least during the two first years of operation: type of plant (*Pragmites australis* was plinted in STW 1 while *Typha* sp. in STW 2 and STW 3) and filter layer configuration (in STW 2 gravel filter of STW a and STW 2 was substituted by wood shavings).

In general results are satisfactory even if higher volume reduction were found in Greece where after 12 years of operation volume reduction was 99% with TS concentration around 50% (Melidis et al., 2010). The weather conditions and the period of operation might have influenced this parameter. In fact, looking at the sludge layer increasing rate recorded in this study (Figure 5.2), after almost 1 year of operation (March 2010), sludge layer was around 22 cm height and TS lower than 10%. However, during the second year of operation (until March 2011) sludge increasing rate was reduced to 5 cm/year (TS 13-15%). This could be due to the initial phase of STW operation, during which plants are growing and their effects are limited. For instance, the scarce root system development might contribute significantly to the reduced sludge dewatering during this phase. Thus, dewatering performances and volume reduction might be enhanced after more years of operation.

Generally, TS concentrations found in this study, ranging between less than 20% in winter to more than 40% in summer, are in accordance with values found in other pilot plants located in Greece and France (Stefanakis et al., 2009; Troesch et al., 2009). Moreover, similar efficiencies are reported in literature from many full-scale systems (Uggetti et al., 2010). According to the one way ANOVA, no significant differences were found on TS concentration between wetlands ($p>0.05$). Thus all the configurations proposed in this study are suitable for sludge dewatering in STW.

Organic matter stabilisation is the other objective of STW together with sludge dewatering. VS and COD are here analysed as a measure of the organic matter reduction during the treatment. VS concentration found in this study (between 50 and 60 %VS/TS) is slightly higher than the range found in literature (36-42 %VS/TS) (Uggetti et al., 2010), probably due to the quite high VS concentration in the influent (up to 70%). As for sludge dewatering, no significant differences were found between wetlands ($p>0.05$). According to the VS results (Figure 5.4), the organic matter mineralisation takes place already over the feeding period, mostly during summer. In fact the lowest VS concentrations (around 50 %VS/TS) were found in July 2010 (10-15% VS reduction), while during winter values increased to approximately 60% VS/TS. A similar pattern was already found from Stefanakis et al. (2009) and can be caused by the reduced microbial at low temperatures. Nevertheless, a higher stabilisation degree was expected over the final resting period as a consequence of the sludge feeding interruption. During this period, results showed VS reduction around 10%, however VS concentration were still rather high after 2 resting months (around 52 %VS/TS in May 2011). This behaviour could be attributed to the important reduction of water content causing the inhibition of the microorganisms responsible for the organic matter mineralisation.

After the experimentation during the feeding period, some parameter related with the agricultural reuse of biosolids were monitored during three months. This aspect to the treatment is still controversy and often depends on the country legislation, in Denmark or France biosolids are directly spread to agricultural fields, while biosolids are post-treated in composting plants in Spain or Italy. This part of the study aims at the determination of the biosolids properties as fertiliser in accordance with the resting period elapsed form the last feeding. The resting time needed to obtain a suitable product for agriculture is strictly dependant on the sludge and biosolids characteristics and the climate conditions such as temperature or rains. On the other hand this parameter is important for its influence on the system operation. In fact, when one or more beds are resting, the surface of the system is limited, thus the surface organic load increases and the resting time between feeding is reduced. All these factors may have consequences on the economic aspects and on the treatment performances. To this end, biosolids were characterised monthly after the last sludge loading (March 2011).

The concentration of nutrients is needed to ensure the appropriate dosages of the sludge for land application. The required agricultural doses are frequently dependent on the fertilizer and soil characteristics (Pomares and Canet, 2001; Andreoli et al., 2007). Although they are essential for plant growth, nutrients (particularly N and P) can be harmful when excessively applied. Thus it is important nutrients application at agronomic rate in order to reduce risks. In general, the sludge is characterized by a considerable variability in nutrient's content, depending on the wastewater source and treatment process, thus the monitoring of these parameters is essential at least at the end of the process. In this study TKN concentration were quite low (around 5 %TKN/TS), but in accordance with previous studies (Stefanakis et al., 2009; Melidis et al., 2010). A certain nitrogen decrease was observed over the last two campaigns probably due to sludge mineralisation, ammonification and plant uptake (Peruzzi et al., 2009). Even lower concentrations were found in TP, especially within the last two campaigns (0.01 %TP/TS) probably due to the phosphate immobilisation in the microbial cells (Elvira et al., 1996) or plant up take (Peruzzi et al., 2009).

To complete biosolids characterisation, phytotoxicity, heavy metals and faecal bacteria indicators were analysed. Phytotoxicity was tested by means of the germination index which had been proven to be one of the more sensitive parameters able to account for both low toxicity affecting root growth, and heavy toxicity affecting germination (Zucconi et al., 1981). The test conducted demonstrates that both lettuce and cucumber seed were able to germinate and root grown. The germination index resultant from seed germination and root growth in sludge was higher than in water samples (values higher than 100%), thus no toxic effects were detected in samples already after 1 resting month.

As mentioned above, the heavy metals concentrations were clearly below the limits set by the current European legislation (Council of the European Union, 1986) already after the last loading. Heavy metals concentration were almost constant during the two months of resting period, suggesting that, in spite of the water content reduction, no heavy metals accumulation was taking place within the stored sludge. Regarding pathogens, Salmonella was absent within all the samples. A significant reduction in *E.coli* concentration was observed after two resting months, when values (lower than 240 MPN/g) matched the limits proposed by the 3rd Draft Working Document (Environment DG, EU, 2000) still not in force (E.coli<500 MPN/g). This is one of the critical points of STW biosolids land application. In fact within wetlands the high temperatures needed for hygienisation are not reached, for this reason post-treatments are required in many countries.

On the whole, biosolids characteristics may significantly vary depending on the sludge source and the treatment conditions. In this case biosolids properties fit quite well with the agricultural application requirements. However, longer resting period will probably enhance sludge mineralisation. As a general recommendation we suggest monthly analysis of mineralisation, nutrients, phytotoxicity, heavy metals and pathogens concentrations in order to determine when the resting period is sufficient to obtain a product suitable as fertiliser.

Conclusions

In this study STW performaces of different configuration were tested in a pilot scale experiment with differen plant specied and filetr medium. Thus, treatment efficiency of *Phragmites australis* was compared to *Typha* sp., while wood shavings were tested as new material for the filter layer.

On the whole, the three configurations studied showed similar efficiency in term of sludge dewatering, mineralisation and hygienisation. No significant differences were found between beds, neither in sludge during the feeding period nor in biosolids during the final resting period.

Concerning treatment performance at the end of the treatment, sludge volume was reduced of about 80% and TS concentration increased up to 16-24%, confirming STW dewatering efficiency found in previous studies. On the other hand, VS reduction to approximately 50%VS/TS indicates rather low mineralisation, indicating that probably further resting period will be needed in order to improve this aspect.

Agricultural applicability of biosolids was suggested by the absence of phytotoxicity (high germination index) and the heavy metals and pathogens concentrations below the legal thresholds.

As a general recommendation this study indicates the suitability of all the configurations tested for sludge treatment in the Mediterranean Region. With regard to the resting period duration, monthly analysis of mineralisation, nutrients, phytotoxicity, heavy metals and pathogens concentrations are suggested in order to determine the optimum resting to obtain a product suitable as fertiliser.

6. Dewatering model for optimal operation of sludge treatment wetlands

This chapter is based on the article:

E. Uggetti, I. Ferrer, A. Argilaga, J. García (2011). Dewatering model for optimal operation of sludge treatment wetlands. In preparation

Sludge treatment wetlands are a technology mainly aiming at sludge dewatering. In spite of many studies on sludge dewatering and mineralisation efficiency, a lack of knowledge concerning operational aspects is still present. The aim of this study was to develop a dewatering model for the determination of the most effective feeding frequency in order to enhance sludge dewatering and reduce sludge layer's increasing rate. The model performed was calibrated with moisture data from a pilot plant. The validation, performed with data from two full-scale systems, suggests model reliability in different climate conditions. The case studies indicate that the optimum feeding frequency (T) is a function of the sludge layer height (H). In spite of the evapotranspiration's contribution to the dewatering performances, it does not influence significantly the feeding time. Besides, the sludge loading rate is determined as a function of evapotranspiration, time between feedings and sludge height.

Introduction

Sludge treatment wetlands (STW), also called drying reed beds, were developed at the end of the 80's with the main objective of sludge dewatering. Besides, a certain degree of sludge mineralization is also reached during the sludge treatment process. STW consists of sealed basins in which sewage sludge is spread onto the surface of a gravel filter planted with wetland plants. Wetlands are fed during some days (from 1 up to 10), during which part of its water content is rapidly drained by gravity through the sludge residue and the filter; while another part is successively evapotranspirated by the plants. In this way, a concentrated sludge residue remains on the surface of the bed where, after some days or weeks without feeding (resting time), sludge is again spread, starting the following feeding cycle. During feeding periods, the sludge layer height increases at a certain rate (around 10 cm/year). When the layer approaches the top of the bed, feeding is stopped during a final resting period (from 1–2 months to 1 year), aimed at improving sludge dryness and mineralisation. The final product is subsequently withdrawn, starting the following operating cycle (Uggetti et al., 2010).

During the last decades, many studies have been carried out both in full-scale and in pilot plants. Sludge dewatering and mineralisation efficiency were investigated under different climate conditions and design factors (sludge influent, plant species, organic loading rate) (Bianchi et al., 2010; Magri et al., 2010; Stefanakis et al., 2009; Uggetti et al., 2010; Vincent et al., 2010). By far, less experience has been developed on operation factors. Feeding and resting timing is actually a key parameter, which can determine the sludge layer increasing rate within the beds and, consequently, the number of the emptying operations needed. Besides, emptying procedure and biosolids transport affect significantly the treatment cost, being the most expensive STW operation (Uggetti et al., 2011). According to Giraldi et al. (2009), the optimisation of resting time might reduce system restoration cost by about 25%.

In spite of the importance of this parameter, currently this aspect of the system's operation is not standardised. In literature, various feeding/resting patterns are reported. While some Danish systems were fed for 7–8 days and rested for 55–56 days, others were fed for 2–3 days and rested for 14–21 days (Nielsen, 2005; Nielsen, 2007). Similarly, 2 weeks of feeding were followed by 14 weeks of rest in a full-scale system in France (Troesch et al., 2008). There are even studies on systems that were loaded only 3–8 times per year (Summerfelt et al., 1998; Obarska-Pempkowiak et al., 2003).

A dewatering model able to simulate both water percolation and evapotranspiration processes could predict the water loss of the sludge layer within wetlands. In order to study dewatering process in STW, the sludge layer can be considered to have similar properties than a soil. Consequently, water drainage in STW can be seen as the result of the pressure exercised by the residual sludge layer. According to Terzaghi and Peck (1967), consolidation is the process in which reduction in volume takes place by the expulsion of water under long term static loads. It occurs when stress is applied to a soil, causing bulk volume reduction and water losing. Consolidation theory refers to any process that involves decrease in water content of a saturated soil without replacement of water by air. This principle is commonly applied to mechanical sludge dewatering (Chu and Lee, 1999). On the other hand, evapotranspiration (ET) is known as the major component of the water balance of many different types of wetlands ecosystems (Zhou and Zhou, 2009). Many efforts have been done in different world's regions in order to measure and simulate ET process of wetlands plants (Baird and Maddock, 2005; Dexler et al., 2008; Borin et al., 2010; Zhongping et al., 2010).

This work aims at developing a model for sludge dewatering in STW. Thus, Terzaghi's theory and ET will be combined and solved with a numerical model. Model calibration and validation will be carried out with analytical data. Moreover, case studies will be provided in order to forecast the optimum feeding frequency and sludge loading rate in different climates. This model will represent a useful tool for the determination of the most effective feeding frequency in order to enhance sludge dewatering and reduce sludge layer's increasing rate.

Materials and methods

Model implementation

Consolidation model

Terzaghi's theory is based on the diffusion equation (Eq. 6.1), where C_v is the consolidation coefficient (m^2/s), u the interstitial pressure ($N/m \cdot s^2$), z the distance (m) and t the time (s).

$$c_v \frac{\partial^2 u}{\partial z^2} = \frac{\partial u}{\partial t} \qquad \text{(Eq. 6.1)}$$

Equation 1 can be written as dimensionless equation

$$\frac{\partial^2 (\Delta u)}{\partial Z^2} = \frac{\partial (\Delta u)}{\partial T} \qquad \text{(Eq. 6.2)}$$

Where $Z = \frac{z}{H}$ $\qquad T = \frac{c_v t}{H^2}$

This equation can be mathematically solved in accordance with the boundary conditions.

Boundary conditions are sets as follow:

1. At t= 0 and at any distance z from the impervious surface, the excess hydrostatic pressure is equal to Δp

2. At any time t at the drainage surface z= H, the excess hydrostatic pressure is zero

3. At any time t at the impervious surface z= 0, the hydrostatic gradient is zero

4. After a very great time, at any value of z, the excess hydrostatic pressure is zero

Evapotranspiration and precipitation term

Due to its relevance in the water balance of natural ecosystems, evapotranspiration needs to be introduced in the dewatering model. ET_0 referred to the standardised reference crop evapotranspiration (mm/d) was calculated using the Penman-Monteith equation (Eq. 6.3) (ASCE-EWRI, 2005):

$$ET_0 = \frac{0.408 \cdot \Delta \cdot (R_n - G) + \gamma \dfrac{C_n}{T+273} u_2 \cdot (e_s - e_a)}{\Delta + \gamma \cdot (1 + C_d \cdot u_2)} \qquad \text{(Eq. 6.3)}$$

Where 0.408=1/2.45 converts the unit from $MJ/m^2 \cdot d$ to mm/d, R_n is the calculated net radiation at the crop surface (MJ/m^2), G the soil heat flux density at the soil surface ($MJ/m^2 \cdot d$, approximated to zero), T is the mean daily air temperature at 1.5 to 2.5 m height (°C), u_2 is the daily wind speed at 2 m height (m/s), e_s is the saturation vapour pressure at 1.5 to 2.5 m height (kPa) calculated for daily steps as the average of saturation vapour pressure at maximum and minimum air temperature. e_a is the mean actual vapour pressure at 1.5 to 2.5 m height (kPa), Δ the slope of the saturation vapour pressure-temperature curve (kPa/°C), γ the psychometric constant (kPa/°C). C_n and C_d for short vegetation (0.12 m) correspond respectively to 900 and 0.34 (mm/d), while for tall vegetation (0.50 m) values are set to 1600 and 0.38 respectively.

Both ET and precipitation values need to be converted to pressures in order to allow ET introduction into the Terzaghi's model (Eq. 6.4 and Eq. 6.5):

$$\frac{\partial^2 U}{\partial Z^2} = \frac{\partial U}{\partial T} + Q \cdot \frac{\tau}{u_0} \qquad \text{(Eq.6.4)}$$

$$Q = q \cdot \frac{u_0}{S_\infty} \quad \text{(Eq.6.5)}$$

Negative values of q indicate predominance of evapotranspiration, while positives ones indicate predominance of precipitation. u_0 is the initial pressure and S_∞ represent the consolidation settlement after infinite time (Eq. 6.6):

$$S_\infty = \int_0^H \varepsilon(z,\infty)dz = \frac{a_v}{1+e_0} \cdot \Delta\sigma \cdot H \quad \text{(Eq.6.6)}$$

Where $E_m = \frac{a_v}{1+e_0}$

E_m and C_v are usually determined by consolidation tests, which defines the compressibility curve and the consolidation coefficient of a soil sample subjected to a one-dimensional compression, using the consolidation theory.

Finally, Eq. 6.7 allows making S_∞ dimensionless by the introduction of the porosity n_o.

$$S_\infty = \frac{S_\infty}{n_0 \cdot H} \quad \text{(Eq. 6.7)}$$

Model solution

Terzaghi's theory can be solved by means of analytical and tabulated solutions. However, in the present study, the integration of ET makes the Terzaghi's model more complicated, and a finite element solution will be required in order to solve Eq. 6.4. For this purpose the software Matlab R2010 was used to create a mesh representing the wetland section. Afterwards, the implicit method of the finite elements was used to solve the differential equation at every time step.

Considering that in STW the pressure is caused by the sludge layer; the initial and the boundary conditions are sets as follow: initial condition f = x; boundary conditions g = h = 0. This means that the initial pressure corresponds to the sludge layer weight, and the surface and the bottom pressure is zero.

Model calibration and validation

Data collection

Moisture data were collected by means of soil moisture probes SM 200 (Delta-T Devices Ltd) connected to a data logger GP 1 (Delta-T Devices Ltd). These soil probes measure soil moisture content at a single location of about 0.5 dm^3 of sludge. Probes consist of a sealed plastic body attached to two sensing rods which transmit an electromagnetic field into the soil. Permittivity variation between soil and water indicates then the sludge water content within the sludge. The probes, calibrated for organic soils, were located *in situ* under the sludge layer. Thus, moisture measurements were recorded every hour and generally refer to a layer 5 cm height, located 10 cm above the sludge surface.

Meteorological data for ET calculation were gathered from the municipal meteorological stations of Barcelona Zona Universitaria and Perafita, Spain, located near the pilot plant and the full-scale systems, respectively (www.meteocat.com).

Facilities considered for calibration and validation

Data for the model calibration were collected from a pilot plant located outdoors on the roof of the Department of Hydraulic, Maritime and Environmental Engineering of the Technical University of Catalonia, Barcelona, Spain. The pilot plant was set up in winter 2008 and consists of three PVC containers with surface area of 1m^2 each and height 1m with different design configuration (Chapter 5). Since May 2009, thickened activated sludge produced at the WWTP near Barcelona, was manually fed one per week. The sludge loading rate was set to 0.025 m^3/week for each STW, corresponding approximately to 40 kgTS/m^2·year.

One pf the bed constituting the pilot plant was used for this study. The drainage filter of the bed was constituted, starting from the bottom, by a 10 cm layer of stones (d_{50}=250 mm), 30 cm of gravel (d_{50}=5 mm) and 10 cm of sand (d_{50}=1 mm). At the bottom of the STW, three perforated PVC pipes were positioned in order to collect leachate and to assess filter oxygenation. Bed was planted with *Phragmites australis*.

Data for the model validation were collected from two full scale systems located in Alpens (400 PE) and Sant Boi de Lluçanès (1,500 PE), province of Barcelona. Main characteristic of the two facilities are summarised in Table 6.1, more details can be found in Uggetti et al. (2009).

Table 1. Main characteristic of the wastewater treatment plants at Alpens and Sant Boi de Lluçanès.

	Alpens	**Sant Boi de Lluçanès**
Population equivalent	400 (800 design)	600 (1500 design)
Type of treatment	Extended aeration	Extended aeration
Sludge production (kg TS/d)	30	45
Sludge flow (m^3/day)	2	3
Total surface area (m^2)	198	324
Sludge loading rate (kg TS/m^2·year)	55	51

Both pilot plant and full-scale systems were fed one day per week, thus the model calibration and simulations were performed for 5 or 8 days, in accordance with the moisture data available.

Model calibration and validation

According to Eq. 6.4 and Eq. 6.6, two parameters need to be calibrated in order to determine sludge dewatering: consolidation and oedometer coefficients (C_v and E_m). Both parameters are related according to Eq. 6.8. C_v defines the dissipation velocity of the pressure exercised by the sludge moisture, thus the slope of the curve moisture vs. time. While E_m influences the consolidation settlement.

$$C_v = \frac{K \cdot E_m}{\gamma_w} \qquad \text{(Eq.6.8)}$$

Where K is the permeability (m/s) and γ_w the water weight (N/m^3)

Calibration was made by setting the sludge height (H), the index of porous (e_0), the porosity (n_0) and the evapotranspiration (q). H was measured *in situ* and n_0 deduced from the moisture values obtained by the soil probes located *in situ*. Thus, C_v and E_m values were determined by matching the model simulation curve with moisture curves deduced from data collected in May from the pilot plant. According to the data availability, validations were performed by using Alpens' data from May; and Sant Boi de Lluçanès records from February, May and June.

Case studies

After the model calibration and validation, different case studies were performed in order to determine the optimum feeding frequency in different climates. For this reason, three climate zones corresponding to different evapotranspiration rates were selected from the Mediterranean Region. The evapotranspiration of the zones selected was calculated for summer and winter seasons according to the Penman-Monteith equation. Moreover, the lowest, the highest and the average ET values, resulting in 14.5 mm/d, 8.6 mm/d, and 2.5 mm/d where selected as representative of the dry continental, the coastal and the Pyrenean climate, respectively.

Moreover, for each climate condition, three scenarios were studied, representing STW in different years of operations. Taking into account the typical sludge height increasing rate of 10 cm/year (Nielsen 2003); the simulation was performed for a 20, 40 and 80 cm sludge layer, corresponding approximately to 2, 4 and 8 years of operation of the system.

Results and discussion

Model calibration and validation

An example of the calibration carried out is shown in Figure 6.1, where t represents each time step after feeding (t=0), x is the sludge height. The surface represented here indicates the water content along the sludge height in every time step, higher values (in red) immediately after sludge loading (t=0) are decreasing in every successive time step. According to this calibration C_v = $3e^{-8}$ m^2/s and E_m= $4e^4$ (kN/m^2). Results are consistent with values found in literature for different soil, where C_v ranges from $3e^{-10}$ to $3.5e^{-8}$ m^2/s depending on permeability and compressibility (Lambe and Withman, 1979). On the other hand, E_m may vary from $2e^2$ kN/m^2 and $2e^3$ kN/m^2 in clay (Gonzáles, 2001).

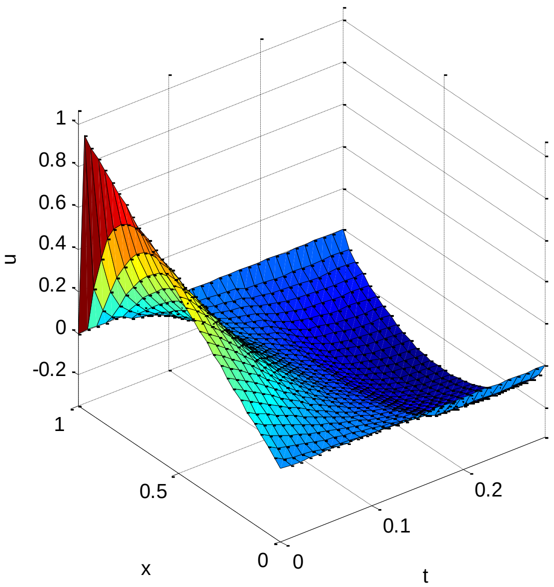


Figure 6.1. Output from the model calibration representing the water loss in STW during the time. t = 0 represents the feeding event.

Values suggested from the calibration of the model were successively tested by means of some validation test performed with moisture data collected from full-scale systems. Figure 6.2 illustrate de validation of the model, carried out with data from two facilities and from different seasons. In general the validations performed in this study show a good response of the model. Although in some cases the curves simulated do not reproduce accurately the trend of the curves recorded by the moisture probes, the model is able to estimate the water loss into the sludge layer estimating rather well the sludge moisture value at the end of the resting period (in this case around 5 days). Note that, corresponding to the data collected by the moisture probe, the model output was set to a 5 cm layer located 10 cm above the sludge surface.

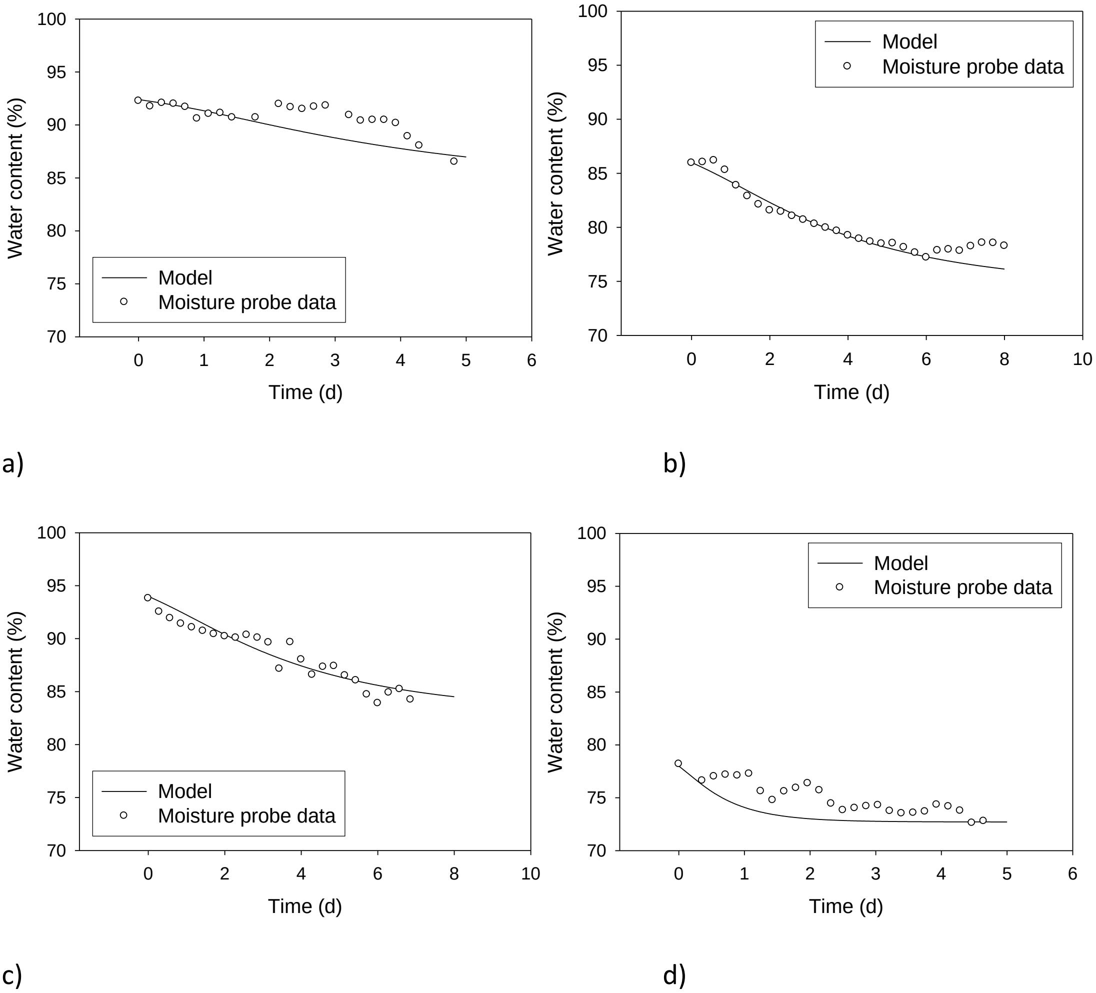


Figure 6.2. Model validations performed in February (a), May (b) and (d) and June (c) with data from Sant Boi de Lluçanès (a, b and c) and Alpens (d).

It is interesting to see that the model is valid for different seasons due to its strict dependence on the evapotranspiration, which reflects the climate conditions. In fact, the final moisture value forecasted by the model correspond to the value recorded by the probe both in winter (Figure 6.2a), when sludge moisture is reduced of about 5%, and in spring (Figure 6.2b, 6.2c and 6.2d), when more than 10% of the water is lost in 8 days. The comparison of the curves measured in February and in May (Figure 6.2a and 6.2b) highlights the different pattern followed by the water loss in different seasons. In fact, in winter (Figure 2a) the initial humidity is higher (around 92%) and the water loss is almost constant during 5 days, this is mainly due to the low evapotranspiration rate which does not enhance sludge drying during the cold season. On the other hand, in spring (Figure 6.2b, 6.2c, 6.2d) the initial sludge moisture is generally lower (around 85%) and the moisture curve is characterised by a higher reduction during the first days followed by lower constant water loss.

Case studies

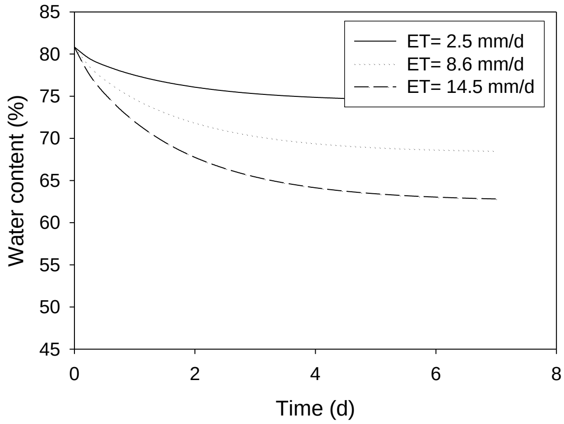
The main objective of the development of this model was to standardise STW operation. Thus, in order to determine the optimum feeding pattern, different case studies were performed by evaluating the influence of two variables, evapotranspiration rate and sludge layer height. Note that here the water content refers to the mean value of the entire sludge layer and not only to a definite depth. As expected, the evapotranspiration rate significantly influences the dewatering performances of the systems. In fact, considering the case study for a sludge layer of 20 cm (Figure 6.3a), 5% of moisture reduction, corresponding to 2.5 mm/d of ET, can be enhanced up to 20% in climate characterised by ET of 14.5 mm/d. As mentioned above, evapotranspiration is responsible for the increasing of the slope of the moisture curve, which is evidently proportional to the ET rate. Thus, the model confirms the relevance of ET in sludge dewatering.

This behaviour is even more evident with the increasing of the sludge layer height, when performances are significantly higher (Figure 6.3b and 6.3c). In the model performed, water loss is due to the consolidation of the sludge layer caused by its pressure, thus sludge layer height is the main parameter influencing the dewatering performances of the treatment. This means that during the first years of operation the dewatering degree achieved is lower than in the following years, when the sludge layer reaches higher elevation, which means higher pressure.

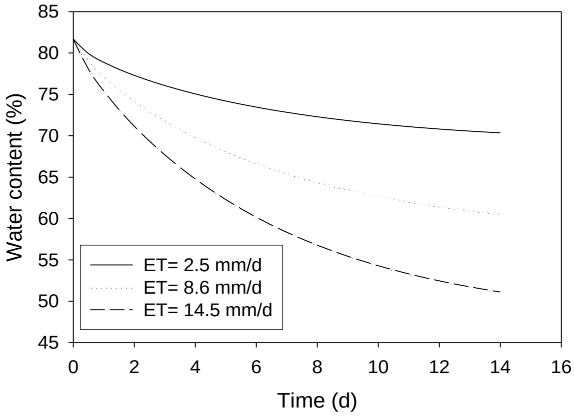
It is important to note that the hypothesis of this model are only valid for saturated conditions, the dewatering degree implying a consolidation degree higher than the consolidation settlement after infinite time (S_∞) will invalidate the model. The model hypothesis should be reformulated for such high dewatering degree, which can be reached after months or years without fresh sludge loading. However, this model can be used to estimate water loss within some weeks of resting between feedings.

The dewatering curves obtained from the case study are useful for the determination of the optimal feeding timing, which is the main practical application of the model. According to the results shown in Figure 6.3, feeding pattern should be varied according to the sludge layer height, which normally corresponds to the years of operation of the system. On the other hand, ET rates improve the dewatering performances but do not influence significantly the dewatering time.

a)



b)



c)

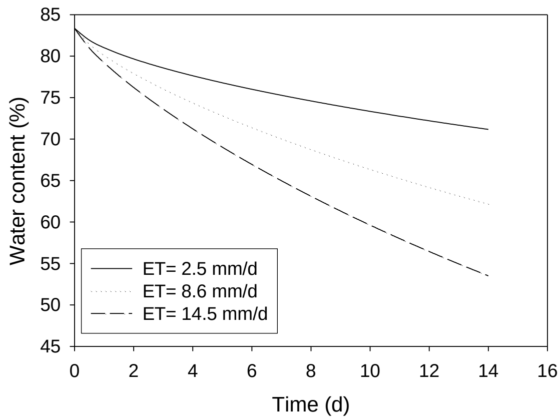
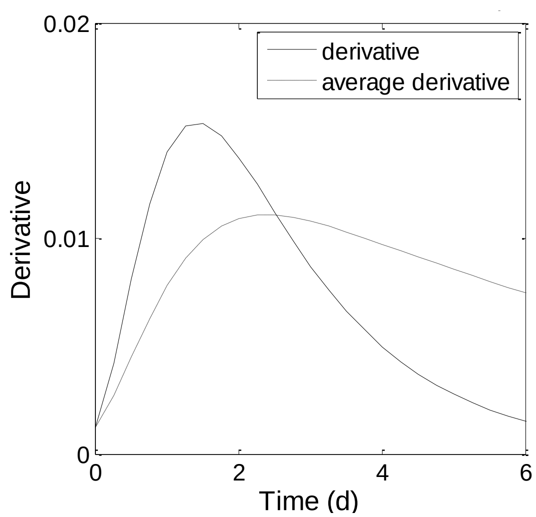


Figure 6.3. Case study considering sludge height of 20 cm (a), 40 cm (b) and 80 cm (c)

In order to establish the optimum feeding timing according to the sludge height, the derivative of the dewatering curve can be useful to determine the point of maximum dewatering. However, in order to enhance sludge dewatering the optimum feeding frequency should include the whole period of sludge dewatering, and not only the maximum instant. Thus, we deduced the optimum feeding frequency by determining the peak of the cumulative moving average of the derivative distribution (Figure 6.4), which corresponds to the time interval in which most of the water loss takes place. This calculation was made considering ET=0, as we observed that evapotranspiration do not influence the dewatering time. According to this calculation, a system with a layer 0.20 cm high, corresponding to 2 years of operation, should be fed every 2.5 days, the resting between feeding increases according to the sludge height, thus for a 0.40 cm layer 10 days is the optimum feeding frequency, while 40 days are required for a 0.80 cm layer. This patters in confirmed by the curves simulated by the model (Figure 6.3), in fact for 0.20 cm of sludge water content is reduced from 10 to 20% during the first 3 days after feeding, depending on the ET rate. Furthermore, for sludge height around 40 cm, period between feeding should be increased up to 10 days in order to reach the same dewatering performances (10-20% of water loss).

According to the optimum feeding periods found in the case studies, Eq. 6.9 determines the optimum feeding time as a function of the sludge height stored within wetlands. It indicates the interval of time corresponding to the maximum dewatering performance. By decreasing this time, the dewatering degree achieved between feedings is reduced, affecting the performances of the treatment, the sludge increasing rate and consequently the treatment costs.

$$T(d) = 0.65 \cdot H(m) \qquad \text{(Eq.6.9)}$$



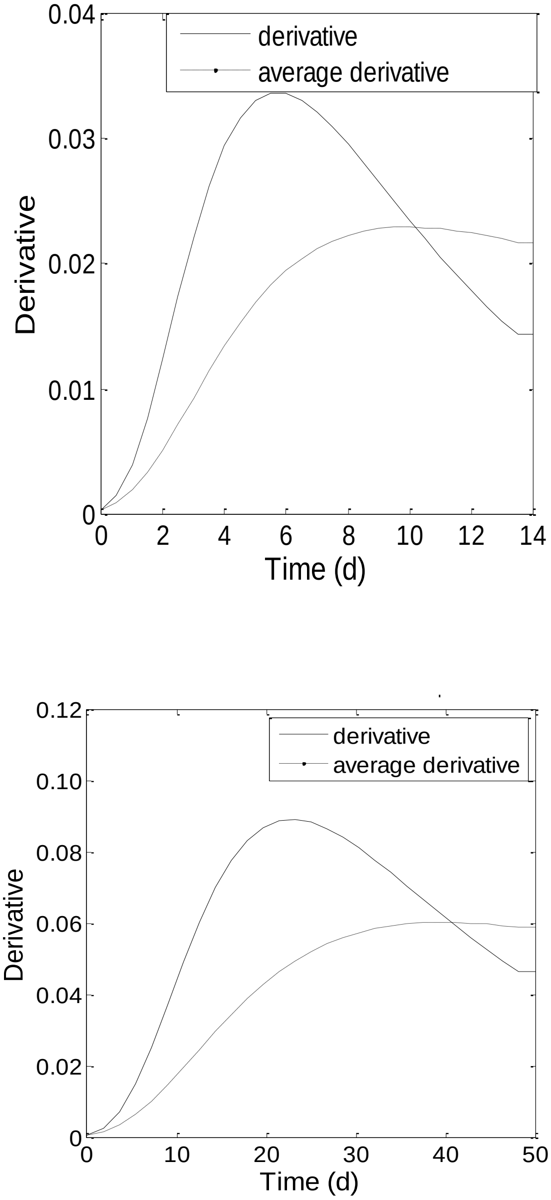


Figure 6.4. Intersection between the derivative of the dewatering curve and the mean of the derivative in each step time.

Table 6.2. Sludge dewatering (mm/d) as function of sludge height (H) and evapotranspiration (ET).

Time between feedings (d)	H (m)	ET(mm/d)		
		2.5	8.6	14.5
3.5	0.15	2.147	4.286	6.347
	0.2	3.291	6.309	9.229
	0.25	4.336	8.086	11.721
	0.5	7.986	13.543	18.929
7	0.3	3.536	6.587	9.540
	0.4	4.994	9.074	13.023
	0.5	6.179	10.964	15.593
	0.8	8.983	14.869	20.560
14	0.4	3.234	5.977	8.629
	0.6	5.331	9.587	13.697
	0.8	6.926	12.069	17.040
	1	8.264	13.957	19.886

Successively, the sludge dewatering corresponding to each period between feedings was calculated from the derivate of the dewatering curve (Table 6.2). By means of the last squares regression, we found a relationship between ET and the feeding time (R^2>0.996). Thus, the sludge loading rate (V), which corresponds to the sludge volume reduction caused by dewatering, can be determined as a function of ET, time between feedings and sludge height (Eq. 6.10).

$$V = \left(0{,}3778 \cdot T^{1{,}1396} \cdot ET + 4{,}0662 \cdot T\right) \cdot \ln H + 1{,}5409 \cdot T^{0{,}8236} \cdot ET + \left(-0{,}2408 \cdot T + 9{,}0975\right) \cdot T$$

(Eq. 6.10)

Where V is the height of sludge loading (mm), H is the sludge layer height (m), T is the time between feedings (d) and ET is the evapotranspiration (mm/d). In this way the model can be used for the determination of operational criteria aimed at the improvement of STW operation. In fact, the standardisation of parameters such as feeding pattern and loading rate can be helpful for the better operation of STW and the enhancement of treatment performances, which leads to a reduction of STW operation costs.

Conclusions

A dewatering model able to simulate sludge dewatering in STW was performed and solved by means of a finite element method. The model combined Terzaghi’s consolidation theory representing water loss by percolation and ET, which increases water loss by plants respiration.

This model was calibrated with moisture data from a pilot plant. According to the validation test, the model implemented is able to forecast the percentage of water loss within the time, giving reliable information about the moisture reduction within the sludge layer.

The application of the model to different case studies results into the determination of the most appropriate feeding frequency as a function of the sludge height stored within the wetland. In the same way, the sludge loading rate is determined as a function of evapotranspiration, feeding frequency and sludge height.

The model implemented is thus able to forecast the percentage of water loss within the time, giving reliable information about the moisture reduction within the sludge layer in different seasons. The importance of the climate variables like temperature, wind speed and pressure into the model suggests its validity for different climate regions. However, further test would be needed in order validate or to adapt the model to a larger range of conditions. On the whole, this model is a useful tool for the establishment of standardised criteria for STW operation.

7. Characteristics of biosolids from sludge treatment wetlands for agricultural reuse

This chapter is based on the articles:

E. Uggetti, I. Ferrer, S. Nielsen, C.Arias, H. Brix, J. García (2011). Characteristics of biosolids from sludge treatment wetlands for agriculture reuse. Waste Management, submitted.

E. Uggetti, I. Ferrer, E. Llorens, D. Güell, J. García (2011) Properties of biosolids from sludge treatment wetlands for land appplication. Chapter 2 In: water and nutrient management in natural and constructed wetlands. Ed. Vymazal, J. Springer, 9-21. ISBN: 978-90-481-9584-8.

Sludge treatment wetlands (STW) consist of constructed wetlands systems specifically developed for sludge treatment over the last decades. Sludge dewatering and stabilisation are the main features of this technology, leading to a final product which may be recycled as an organic fertiliser or soil conditioner. In this study, biosolids from full-scale STW were characterised in order to evaluate the quality of the final product for land application, even without further post-treatment such as composting. Samples of influent and treated sludge were analysed for pH, Electrical Conductivity, Total Solids (TS), Volatile Solids (VS), Chemical Oxygen Demand (COD), Dynamic Respiration Index (DRI), nutrients (Total Kjeldahl Nitrogen (TKN), Total Phosphorus (TP), Potasium (K), heavy metals and faecal bacteria indicators (*E. coli* and *Salmonella* spp.). According to the results, the sludge water content is reduced from 1.5-5.0 %TS to 75-82 %TS. Organic matter biodegradation leads to VS around 43-46 %TS and COD around 500-700 g/kgTS. The values of DRI24h (0.5-1.4 $mgO_2/gTS{\cdot}h$) indicate that treated sludge is an almost stabilised final product. Besides, the concentration of nutrients is quite low (TKN, TP and K all around 0.1 %TS). Both heavy metals and faecal bacteria indicators meet current legal limits for land application of the sludge. Our results suggest that biosolids from the studied sludge treatment wetlands could be valorised in agriculture, especially as soil conditioners.

Introduction

Sewage sludge is the waste generated by wastewater treatment processes, after solid and liquid separation units. The amount of sludge produced and its composition depend on the influent's characteristics and the wastewater treatment used. Sludge production in conventional activated sludge processes ranges from 60 to 80 g of total solids per person per day (Von Sperling and Gonçalves, 2007). In Europe, the Urban Wastewater Treatment Directive 91/271/EEC (Council of the European Union, 1991) promoted the implementation of wastewater treatment plants (WWTP) with secondary wastewater treatment in municipalities above 2,000 Population Equivalent (PE); and the Water Framework Directive (Council of the European Union, 2000) encouraged wastewater treatment even in municipalities below 500 PE. As a result sludge production has increased in the European Union by 50 % since 2005 (Fytili and Zabaniotou, 2008). For instance, in Catalonia (north-est Spain) around 50 % of the WWTP (170) were constructed between 2000 and 2006 (Agencia Catalana del Agua, 2007).

In order to manage the increasing amount of sludge produced in Spain the following hierarchy was proposed (Consejo de Ministros, 2001): 1) valorisation in agriculture, 2) energetic valorisation, and 3) landfilling. Agricultural valorisation is nowadays preferred to landfilling, since sludge recycling ensures the return of organic constituents, nutrients and microelements to crop fields, which eases the substitution of chemical fertilizers (Oleszkiewicz and Mavinic, 2002). Sludge disposal onto agricultural land is regulated by the European Sludge Directive (Council of the European Union, 1986), which controls land application of sewage sludge according to heavy metals concentrations. Recent regulation proposals are more restrictive in terms of heavy metals, and also consider emerging pollutants and microbial faecal indicators (Environment DG, EU, 2000).

In practice, sludge treatment systems should provide a final product suitable for land application (fulfilling legislation requirements), with reasonable investment as well as operational and maintenance costs. In this sense, sludge treatment wetlands (STW) are regarded as a suitable technology for sludge management from both an economical and environmental point of view (Uggetti et al, 2011). Treatment wetlands reproduce self-cleaning processes occurring in natural wetlands and are being used for wastewater treatment in many regions of the world (Brix et al., 2007, Puigagut et al., 2007). Since the late 1980s, treatment wetlands have been adapted for sludge treatment, a technology that is nowadays used in most European countries and in North America (Uggetti et al., 2010).

Sludge treatment wetlands consist of sealed basins with a filter consisting of successive layers of stone and gravel fractions, in which wetland plants like *Phragmites australis* (common reed) are planted. The sludge is discharged and spread evenly on the wetland's surface. Here, part of the sludge water content is rapidly drained by gravity through the sludge residue and the filter; while another part is evapotranspirated by the plants. In this way, a concentrated sludge residue remains on the surface of the bed where, after some days without feeding (resting time), sludge is again spread, starting the following feeding cycle. During feeding periods, the sludge layer height increases at a certain rate (around 10 cm/year). When the layer approaches the top of the banks or walls surrounding the STW (usually after 8 to 12 years), feeding is stopped. The sludge remains for a final resting period (from 1–2 months to 1 year), aimed at improving sludge dryness and mineralisation. The final sludge product is subsequently withdrawn, leaving only a thin layer at the bottom of the bed, and the following operating cycle can start.

The quality of the final sludge product is the result of both dewatering processes (draining and evapotranspiration) and organic matter biodegradation (Nielsen, 2003). According to Nielsen and Willoughby (2005) it is suitable for land application; although further post-treatments might be required to improve sludge hygienisation (Zwara and Obarska-Pempkowiak, 2000). Nevertheless, detailed studies on the properties of biosolids from STW are still lacking in the literature.

The aim of this study was to evaluate the efficiency of full-scale STW in term of sludge dewatering, mineralization and hygienisation and to characterise the quality of the final product for land application and agricultural reuse. To this end, physico-chemical and microbiological parameters, as well as stability indexes, were analysed, as proposed in the European Sludge Directive (Council of the European Union (1986)), the 3rd Draft EU Working Document on Sludge (Environment DG, EU, 2000) and the 2nd Draft EU Working Document on Biological Treatment of Biowaste (Environment DG, EU, 2001).

Materials and Methods

Systems' description

Three full-scale STW of different size were selected for this study (Figure 7.1); one located in Spain (Seva, 1,500 PE) and two in Denmark (Greve, 50,000 PE and Hadsten, 12,000 PE). The main characteristics of the facilities are summarised in Table 7.1.

In Seva the STWs were set-up in 2000 by transforming conventional drying beds. They were planted with *P. australis.* The surface area of the 7 basins is 175 m^2 and the sludge loading rate around 125 kg $TS/m^2 \cdot year$, much higher than the recommended value of 50-60 $kgTS/m^2 \cdot year$ (Burgoon *et al.*, 1997; Edwards *et al.*, 2001; Nielsen, 2003). Other details on the design and operation of the wetlands may be found in Uggetti *et al.* (2009a). The first operating cycle lasted about 5 years; the sludge was then removed and the process re-started (without replanting) between 2004 and 2005. The second operating cycle was finished by the end of 2008 for the wetland beds used in this study (basin 1 and basin 2).

After a resting period of 4 months, these basins were emptied with a power shovel and the final product was transported to a composting plant.

Table 7.1. Characteristics of the WWTP and STW in Seva (Spain), Greve and Hadsten (Denmark).

	Seva	Greve	Hadsten
Treated population equivalent	1,500	50,000 (60,000 design)	12,000 (20,000 design)
Sludge source	Municipal	Municipal	Municipal
Type of treatment	Contact-stabilisation	Surplus activated System (MBKDN)	Surplus activated System (MBKDN)
Wastewater flow rate (m^3/d)	180 (summer) 400 (winter)	22,500	-
Sludge production (kg TS/d)	60	3,300	820
Number of treatment Basins	7	10	6 (8 design)
Total surface area (m^2)	175	16,700	3,810
Nominal height for sludge accumulation (m)	~0.8 (capacity 1)	1.30-1.50 (capacity 1.70)	~1.30 (capacity 1.70)
Sludge loading rate (kg TS/m^2·year)	125	50-80 (60 design)	85 (60 design)

Greve's system has been in operation since 1999, 10 beds (16,700 m^2) were established to treat the sludge from a near WWTP (3.5 km). They were planted with *P. australis.* The design loading rate was 60 kgTS/m^2·year, as suggested in the literature. However, the actual loading rate range from 50 to 80 kgTS/m^2·year, indicating the overloading during some periods. In 2010, after 11 years of operation, three basins were emptied for the first time and two of these (basin 3 and basin 4), were sampled after a resting period of 8 and 18 months, respectively.

Hadsten's system was set-up in 1999. Here 8 basins were constructed 3 km from the WWTP with a total surface of 5,080 m^2 and planted with *P. australis*. This system was loaded with 55-60 kgTS/m^2·year until 2006. Between 2007 and 2008 four beds were emptied, but only two of them were re-established. Thus, since 2008 the 6 beds (3,810 m^2) are loaded at a rate of 85 kgTS/m^2·year, reducing the resting period and causing a certain overloading, which might affect biosolids quality. In August 2010, during the emptying process, two basins (basin 5 and basin 6) were sampled after a resting period of 4 months. In Denmark, biosolids that match legal requirements are immediately spread on crop fields if emptying occur after harvest (from August to October).

Figure 7.1 Emptying operation in Seva (Spain), Greve and Hadsten (Denmark) (form the top to the bottom).

Sludge sampling and characterisation

When the basins were emptied (in 2008 in Seva, and in 2010 in Greve and Hadsten), the layer height was about 80 cm in Seva and about 130 cm in Greve and Hadsten. In each system, composite samples were prepared by mixing subsamples from each basin; while an integrated influent sample was obtained in Seva from subsamples collected during a whole week.

The sludge quality was characterised in terms of: pH, Electrical Conductivity (EC), Total and Volatile Solids (TS and VS), Chemical Oxygen Demand (COD), Total Kjehldahl Nitrogen (TKN), Potassium (K), Total Phosphorous (TP), heavy metals and faecal bacteria indicators (*Salmonella* spp*.* and *Escherichia coli*). Additionally, the Dynamic Respiration Index (DRI) was determined according to Adani *et al.* (2000) and Barrena *et al.* (2009a).

Sludge samples were analysed following Standard Methods (APHA-AWWA-WPCF, 2001). Samples for COD, TKN, TP, K and heavy metals' analyses were air dried at room temperature (until constant weight) before analysis and the results are expressed on a dry matter basis (per kg or %TS). Air dried samples were subsequently equilibrated in distilled water (1:5) for pH and EC measurements.

Results and discussion

Tables 2 and 4 show the main characteristics of influent and treated sludge from the full-scale STW, and the properties of biosolids from different sludge treatments are compared in Table 7.3. Moreover, a comparison with compost is proposed to assess the requirement of composting post-treatment following STW.

Sludge dryness

Secondary sludge is spread on the beds with very high water content (typically around 99 %) (Table 7.2). Sludge moisture is significantly reduced to 75-82 % during the treatment and final resting period (ranging from 4 months in Seva to more than 1 year in Greve). This is the main goal of the dewatering processes.

The dryness of the final product (TS around 18-25 %) is lower than the observed in other facilities after a resting period of approximately one year (TS around 30-40 %) (Nielsen, 2003). A previous study also indicated a poor dewatering efficiency in Seva's system compared to other Catalan facilities (Uggetti *et al*., 2009a). A possible reason for this is that the sludge loading rate (125 $kgTS/m^2 \cdot year$) is twofold the recommended value of 50-60 $kgTS/m^2 \cdot year$ (Burgoon *et al.*, 1997; Edwards *et al.*, 2001; Nielsen, 2003). Also in Hadsten a certain overloading resulting from the reduced number of basins might have reduced the dewatering efficiency of the system. Moreover, samples from Denmark were collected in August, when the wet climate might influence sludge dryness. This fact suggests that the dryness of the final product could be enhanced with a better system management (i.e. reducing the sludge loading rate, applying the correct resting period and emptying beds during the dry season).

Table 7.2. Physico-chemical properties of influent sludge and biosolids from the studied STWs (mean ± standard deviation).

Parameter	Seva			Greve			Hadsten		
Physical properties	**Influent**	**Basin 1**	**Basin 2**	**Influent**	**Basin 3**	**Basin 4**	**Influent**	**Basin 5**	**Basin 6**
pH	6.75	6.21	6.27	6.50	6.85	6.65	7.30	5.09	5.78
EC 1:5 (dS/m)	0.30	1.51	1.88	-	0.81	1.68	-	1.93	1.23
TS (%)	1.1 ± 0.0	24.2 ± 0.6	25.8 ± 2.1	1.42 ± 0.4	26.3 ± 3.7	19.5 ± 0.5	0.5 ± 0.1	20.2 ± 2	18.2 ± 0.1
Organic matter									
VS (%TS)	51.5 ± 0.8	42.9 ± 1.8	44.6 ± 3.0	65.8 ± 4.6	44.6 ± 0.5	46.6 ± 5.2	55 ± 0.1	46.5 ± 2.6	45.4 ± 0.5
COD (g/kgTS)	709 ± 11	554 ± 32	494 ± 55	-	713 ± 95	693 ± 10	-	771 ± 48	664 ± 8
DRI_{24h} (mgO_2/gTS·h)	-	1.4 ± 0.3	1.1 ± 0.2	-	0.5 ± 0.0	0.5 ± 0.1	-	0.7 ± 0.1	0.5 ± 0.1
Nutrients									
TKN (%TS)	0.02	0.03	0.25	4.96	0.20	0.11	-	0.12	0.12
TP (%TS)	2.68	0.13	0.39	3.03	0.08	0.07	-	0.27	0.16
K (%TS)	0.27	0.18	0.62	-	0.01	0.01	-	0.01	0.01

Organic matter and stability

The concentration of VS in the influent is generally low (51-66 %VS/TS) as a result of the high solids retention time in the wastewater treatment process. In spite of the variability in influent composition (51-65% VS/TS), the organic content of the biosolids is quite uniform in all the basins analysed (43-46 %VS/TS), falling into the range obtained after conventional sludge stabilisation techniques, such as anaerobic digestion (Ferrer, 2010) (Table 7.3). Again, the results are in accordance with previous studies (Uggetti el al., 2009a and 2009b), which highlights the mineralisation process through the treatment. Moreover, mineralisation processes in STW were also assessed by Peruzzi et al. (2009).

Table 7.3. Physico-chemical properties of the final product from different sludge treatments systems.

Parameter	Mechanical dewatered sludge	Composted sludge	Anaerobically digested sludge	Biosolids from STW
pH	7.5	6.6-7.5	7.5 (1:5)	6.1 (1:5)
EC (dS/m)	1.4	2.9-4.1	1.9 (1:10)	9.0 (1:5)
TS (%)	-	56-83	18	22
VS (%TS)	54	62-71	50	45
TNK (%TS)	3.3	2.3-2.6	6.9	0.15
TP (%TS)	2.56	2.33	2.04	0.15
K (%TS)	0.32	0.65	-	0.14
Reference	Sánchez et al., 2010	Bertrán et al., 2004;Sánchez et al., 2010	Fang and Wong, 1999, Ferrer et al., 2010	This study

In STW, plants are a key element for sludge mineralisation, contributing through the transport of oxygen from the aerial parts to the belowground biomass. This oxygen is released in the rhizosphere, which creates aerobic microsites in the bulk sludge layer and thus ensures appropriate conditions for aerobic degradation processes and other oxygen-dependent reactions like nitrification (Vymazal, 2005). Plants also indirectly contribute to aerobic mineralisation through stems, which as a result of their movement (by the wind) crack the surface of dry sludge and prompt aeration of the upper sludge layers. In addition, the effect of the movement of the stems and the complex root system support pore maintenance within the sludge layer and preserve drainage efficiency through the gravel filter (Nielsen, 2003).

The organic content of compost is usually much higher due to humic-like substances produced during the composting process. Values from the literature range between 71 % VS/TS for compost of sewage sludge (Ruggieri et al., 2008) and 62 % VS/TS for compost of sewage sludge mixed with vegetable wastes (Bertan et al., 2004). Similar values (around 50 % VS/TS) are described for dewatered sewage sludge (Sánchez et al., 2010) (Table 7.3). However, comparisons with other systems are not straightforward, since the biodegradability of the sludge depends on a number of parameters, including its nature and composition.

Organic matter in soil amendments can improve the properties and quality of soils, which is essential to guarantee long-term soil fertility (Draeger et al., 1999). In particular, an increase in organic matter content can improve physical properties (water retention, soil structure, water infiltration, bulk density, porosity), chemical properties (cation exchange capacity, pH) and, in some cases, biological properties (Moss et al., 2002, Andreoli et al., 2007). Such a response depends on the sludge:soil ratio (Singh and Agrawal, 2008) and can be useful for soil restoration.

On the other hand, higher biological stability implies lower environmental impacts (like odour generation, biogas production, leaching and pathogen's re-growth) during land application of the product (Muller et al., 1998). Despite the significance of biosolids biological stability upon land application, current legislation does not set stability values, due to the high dependence on final destination. Lasaridi et al. (1998) define biological stability as a characteristic that determines the extent to which readily biodegradable organic matter has been decomposed. Referring to compost, the stability is a quality parameter related to the microbial decomposition or microbial respiration activity of composted matter (Komilis et al., 2009).

The DRI is based on the rate of oxygen consumption and is a useful indicator of the biological stability of a sample, the lower the DRI value, the higher the stability. In this study, the DRI24h from samples ranged between 1.1 and 1.4 mgO_2/gTS·h in Seva and between 0.7 and 0.5 mgO_2/gTS·h in the Danish's systems (Table 7.2). Such a stability degree is much higher than the values reported in the literature for a mixture of primary and activated sludge (6.7 mgO_2/gTS·h) and for anaerobically digested sludge (3.7 mgO_2/gTS·h) (Pagans et al., 2006).

In a recent study, Ponsá et al. (2008) analysed the DRI of the organic fraction of municipal solid wastes at different stages of a mechanical biological treatment. These authors observed DRI values above 7.0 mgO_2/gTS·h for the input material, a decrease to around 1.5 mgO2/gTS·h for digested material and near 1.0 mgO_2/gTS·h for composted material, with a value of 1.0 mgO_2/gTS·h for the output material. Similarly, Scaglia and Adani (2008) found values around 2.5 mgO_2/gTS·h for input samples, around 1.1 mgO_2/gTS·h for intermediate samples and between 0.3-0.6 mgO2/gTS·h for the final product of the stabilisation process.

The comparison of the results suggests that biosolids from the studied STW achieve almost the same stabilization degree as compost; it is thus suitable for land application. Especially in Denmark, already after 4 months of resting, values are consistent with stabilised material. In fact, the final product from the Danish systems is valorised in agriculture without post-treatment in a composting plant (Figure 7.2), which reduces sludge treatment costs compared to systems needing composting (Uggetti et al., 2011).

Figure 7.2 Biosolids spreading in an agricultural field in Hadsten (Denmark).

Nutrients

Sewage sludge provides essential nutrients for plant growth. Biosolids are able to restore N, P, sulphur and other nutrients in soils. The concentration of nutrients in biosolids depends on sewage composition and treatment, and on subsequent sludge management. N, P and K concentrations resulting from in this this study are shown in Table 7.2.

Nitrogen comes from microbial biomass present in the sludge and from wastewater residues. In this study, TKN values in biosolids range between 0.25 and 0.03 % TKN/TS (Table 7.2), being significantly lower than in activated sludge (Andreoli et al., 2007). Significantly higher TKN values (2.30-2.53 %TNK/TS) are found in compost of sewage sludge (Bertan et al., 2004; Ruggieri et al, 2008; Sánchez et al., 2010) (Table 7.3). In Greve, we detected a certain nitrogen reduction, which takes place in STW due to sludge mineralisation, ammonification and plant uptake (Peruzzi et al., 2009). On the other hand, in Seva, the absence of TN reduction may be due to the fact that the nitrogen in the influent is already in its stable from, which is normally more than 50% of the total nitrogen (Peruzzi et al., 2009).

Phosphorus in sludge comes from biomass formed during wastewater treatment, residues and phosphate-containing detergents and soaps. Biosolids can be seen as a P source assuring a slow and continued release to plants (Andreoli et al., 2007). In this study, TP values show a clear decrease from the influent (2.68-3.03 %TP/TS) to the treated sludge (0.07-0.39 %TP/TS), probably due to the phosphate immobilisation in the microbial cells (Elvira et al., 1996). The same patter was already detected by Peruzzi et al. (2009), which also found certain P retention in plants causing low TP concentration in treated sludge. As shown in Table 7.3, the values of the final product are quite lower than in composted or mechanical dewatered sewage sludge (Bertrán et al., 2004; Sánchez et al., 2010).

In a previous study, Yubo et al. (2008), detected a certain decreasing in nutrient concentration (TN and TP) along the vertical profile of sludge treated in STW, probably due to the plants adsorption during the growing season. The same patter was detected by Pempkowiak and Obarsza-Pempkowiak (2002). On the other hand, the concentration of K found in this study does not seem to vary after the treatment, with values ranging between 0.18 and 0.62 %K/TS in Seva. These values are in accordance with sludge compost (Bertrán et al., 2004) (Table 7.3). However, very low concentrations were detected in Danish facilities (0.01 %K/TS).

In general, the sludge is characterized by a considerable variability in nutrient's content, depending on the wastewater source and treatment process (Moss et al., 2002). The concentration of nutrients is needed to ensure appropriate dosages of the sludge for land application. The required agricultural doses are frequently dependent on the fertilizer and soil characteristics (Pomares and Canet, 2001; Andreoli et al., 2007).

Although they are essential for plant growth, nutrients (particularly N and P) can be harmful when excessively applied. Different studies have shown that both N (Walter et al., 2000, Hernandez et al., 1990) and P accumulation in sludge-amended soils (Hernandez et al., 1990). It is well known that over application of N can lead to nitrate contamination of groundwater; although such a risk is reduced if nutrients are applied at agronomic rates (Moss et al., 2002).

Table 7.4. Concentrations of heavy metals in influent sludge and biosolids from the studied STWs.

Parameter	Seva			Greve			Hadsten			Limit values
Heavy metals	Influent	Basin 1	Basin 2	Influent	Basin 3	Basin 4	Influent	Basin 5	Basin 6	
Ni (mg/kgTS)	39	30	32	65	76	-	-	-	-	300-400
Cu (mg/kgTS)	252	318	213	290	305	440	-	232	197	1000-1750
Zn (mg/kgTS)	719	588	641	670	874	1189	-	1269	1085	2500-4000
Cd (mg/kgTS)	1.7	0.8	0.8	0.83	1	1	-	1	1	20-40
Hg (mg/kgTS)	<1.5	<1.5	<1.5	1	1.5	5	-	1	2	16-25
Pb (mg/kgTS)	53	73	76	42	57	72	-	64	55	750-1200

Note: Limit values according to the current European legislation (Council of the European Union, 1986)

Heavy metals and faecal bacteria indicators

The main hazard associated with sludge application on agricultural soils is the potential long term accumulation of toxic elements (Singh and Agrawal, 2008), which may be taken up by crops. Such elements include both inorganic pollutants, like heavy metals, and organic micropollutants. Currently, however, only heavy metals concentrations are regulated for land application of sewage sludge in the European Union (Council of the European Union, 1986), although Denmark and some other European countries has regulations that include organic micropollutants. Table 7.4 summarises the concentration of heavy metals in the STW together with the limits set by the current European legislation (Council of the European Union, 1986). According to the results, there are only little differences between influent sludge and the final product, suggesting that heavy metals accumulation in sludge is negligible. In all samples heavy metals concentration was clearly below the law thresholds; this is also probably attributed to the low concentration in the influent. In fact, the facilities considered treat urban water, thus poor in heavy metals.

Even if this work did not analyse this aspect, heavy metal uptake by plants is likely to be the main biological removal mechanism (Sheiran and Sheoran, 2006). According to Ye et al. (2001) and Hallberg and Johnson (2005), another important biological uptake process in wetlands is bacterial metabolism. However, De Maeseneer et al. (1997) reported that the amount of heavy metals uptaken by *Phragmites australis* is lower than in the case of *Salix fragilis* and *Salix trandra*. Also Peruzzi et al. (2007) detected a slight increase in the heavy metal concentration in *Phragmites australis* shoots after 400 days of systems' operation; being significantly lower than heavy metals concentration in sludge. Furthermore, heavy metal concentrations in sludge and in *Phragmites australis* are generally below disposal standards and do not pose a problem for sludge and reed disposal or recycling (Begg et al., 2001).

The bioavailability of heavy metals in soil and plants depends on soil pH, plant species and their cultivars, growth stage, biosolids source, soil condition and the chemistry of the element (Warman and Termeer, 2005). According to these authors, it is important to monitor the Cu and Zn concentrations in plant tissues after few years of sludge application to verify the tolerance levels for animal feed or human food. In this study only in Greve, Zn concentration in biosolids is significantly higher than in the influent. Furthermore, in all cases the concentrations are clearly below the consent limits.

Since treated sludge may contain high numbers of pathogens, depending on the treatment processes used, limit values for faecal bacteria indicators have also been suggested. In fact, the Environment DG, EU, 2000 proposes limits values for Salmonella spp. (absence in 50 g) and *E.coli* (6 $\log_{10}$ reduction to less than $5 \cdot 10^2$ MPN/g). The faecal indicators Salmonella spp. and E. coli were analysed in Seva; Salmonella spp. was not detected in 25 gTS, while *E.coli* was present in small quantities in all cases (less than 3 MPN/gTS). Both faecal bacteria indicators are well below the limits proposed (absence of Salmonella in 50 gTS and *E.coli* <500 MPN/gST) (Environment DG, EU, 2000). Moreover, a study carried out in Denmark indicates that Salmonella is reduced to <2/100g and *E.coli* to <200MPN/100g between 1 and 4 months after loading (Nielsen, 2007).

The 3rd Draft European Working Document on Sludge (Environment DG, EU, 2000) also suggests limit values for the concentrations of organic compounds and dioxins in sludge, which are not considered in this work. Nevertheless, previous studies demonstrate that toxic organic compounds like nonylphenolethoxylates (NPE) and linear alkylobenzene sulphonates (LAS) are mineralised within 5 and 6 months of treatment (Nielsen, 2005 and Peruzzi et al., 2010) leading to concentrations below the proposed threshold. Moreover, Nielsen (2005) showed that STW were able to achieve NPE and LAS mineralisation degrees similar to composting or mechanical aeration.

On the whole, the characteristics of the biosolids from the STWs (dryness, organic matter contents and nutrients) show that the biosolids are suitable for land application, especially as an organic amendment or soil conditioner. Moreover, biosolids from STWs fulfil the requirements for agricultural application concerning the concentration of heavy metals and faecal bacteria indicators. However, in Spain, biosolids from STW are post-treated in composting plants before agricultural reuse, while they are directly spread on fields in countries like Denmark and France (Nielsen and Willoughby, 2005; Liénard et al., 2008).

Conclusions

This study characterised the properties of biosolids from sludge treatment wetlands and compared them with other stabilised products, like anaerobic digested sludge and compost. Focus was put on the quality of biosolids for agricultural reuse as organic fertilisers and soil conditioners. From this work, the following conclusions can be drawn.

In the full-scale STW studied, sludge moisture is reduced by 18-25 % TS, from 99 to 82-75 %. The moisture content could be improved further by optimising the loading rate, the resting period and the emptying procedures. Organic matter biodegradation leads to VS contents around 45 %VS/TS and COD concentrations around 500-700 g/kgTS in the final product, which is comparable to digested sludge. Besides, DRI values (1.4 -0.5 $mgO_2/gTS \cdot h$) indicate a partly stabilised product, with nearly the stabilisation degree of compost. This suggests that composting post-treatments would not be needed if sufficient resting time is left at the end of each operating cycle. Monitoring the stabilisation degree as the heavy metals and pathogens concentration during the final resting period would help optimising the duration of such a period. Hence, sludge treatment wetlands can produce a final product which can be used on land without further post-treatment.

8. Quantification of greenhouse gas emissions from sludge treatment wetlands

This chapter is based on the article:

E. Uggetti, J. García, S.E. Lind, P.J. Martikainen, I. Ferrer (submitted). Quantification of greenhouse gas emissions from sludge treatment wetlands. Water Research, submitted.

Constructed wetlands (CW) are nowadays successfully employed as an alternative technology for wastewater and sewage sludge treatment. In these systems organic matter and nutrients are transformed and removed by a variety of microbial reaction and gaseous compounds such as methane (CH_4) and nitrous oxide (N_2O) are released to the atmosphere. The aim of this work is to introduce a method to determine greenhouse gas emissions from sludge treatment wetlands (STW) and use the method in a full scale system. Sampling and analysing techniques used to determine greenhouse gas emissions from croplands and natural wetlands were successfully adapted to the quantification of CH_4 and N_2O emissions from a STW. Gas emissions were measured using the static chamber technique in 9 points of the STW during 13 days. Spatial variation in the emission along the wetland did not follow some specific pattern found for the temporal variation in the fluxes. Emissions ranged from 10 to 5,400 mg $CH_4/m^2 \cdot d$ and from 20 to 950 mg $N_2O/m^2 \cdot d$ depending on the feeding events. The comparison between the CH_4 and N_2O emissions of different sludge management options shows that the STW has the lowest atmospheric impact in terms of CO_2 equivalent emissions (Global warming potential with time horizon of 100 years): 17 $kgCO_2eq/PE \cdot y$ for STW, 36 $kgCO_2eq/PE \cdot y$ for centrifuge and 162 kgCO2eq/PE·y for untreated sludge transport, PE means Population Equivalent).

Introduction

Constructed wetlands (CWs) constitute an alternative to conventional wastewater treatment systems to reduce pollutant discharge to water bodies. The application of this technique has increased during the last decades especially in the treatment of different effluents such as municipal and industrial wastewater, agricultural water, landfill leachate and runoff from managed peatlands (Vymazal, 2009). More recently, CWs have been adapted to sewage sludge treatment, as a low cost and low energy demand process, to enhance sludge dewatering and stabilization needed for agricultural uses., Such systems have demonstrated a good applicability as an alternative sludge treatment for small (<2,000 Population Equivalent, PE) and remote communities (Uggetti et al., 2011).

Removal of organic matter in wetlands is mediated by several microbial reactions such as aerobic respiration, denitrification, sulphate reduction, fermentation processes, and methanogenesis (García et al., 2005). By means of such reactions inorganic gaseous compounds such as methane (CH_4) and nitrous oxide (N_2O) are released to the atmosphere (Kadlec and Wallace, 2009). These compounds are known as greenhouse gases due to their contribution to the radiative forcing of the atmosphere and consequently to the Climate Change. Thus, their production by CWs is a matter of concern, which needs to be clarified before the massive implementation of this technique.

It is well known that CH_4 is produced in anoxic soils and sediments, while drained soils act as a sink for atmospheric CH_4 due to methane oxidation through methanotrophs (Hanson and Hanson, 2006). Methane production is regulated by numerous factors, including oxygenation, water table, plant species and temperature (Grünfeld and Brix, 1999).

On the other hand, nitrogen is removed in agricultural soils, riparian buffer zones and natural or constructed wetlands by biological processes, mainly nitrification and denitrification (Brodrick et al., 1988; Bastviken et al., 2005; Maljanen et al., 2004; Groffman et al., 2000). Both nitrification and denitrification can lead to the emission of nitrogen oxides (Kampschreur et al., 2009). Formation of N_2O depends on several environmental conditions such as availability of oxygen, carbon and nitrogen, and hydraulic loading rate (Knowles, 1982).

The gas dynamics are strongly affected by climatic factors, especially temperature and humidity (Martikainen et al., 1993). Humidity conditions within the wetlands determine the location and extension of aerobic and anoxic processes. As a result, the gas fluxes in the systems have a strong seasonal and temporal variability (Liikanen et al., 2006).

Until now, greenhouse gas emissions (GHG) have been measured from agricultural soils (Gregorich et al., 2005), rice fields (Ahmad et al., 2009), riparian buffer zones (Teiter and

Mander, 2005), peatlands (Alm et al., 1999), municipal wastewater treatment plants (Sümer et al., 1995) and constructed wetlands (Søvik et al., 2006; Søvik and Klove, 2007). To our knowledge, GHG emissions from sludge treatment systems (STW) have not been quantified, nor have been developed techniques to determine their gas emissions.

With the increasing application of STW, it is relevant to study their GHG emissions and their possible contribution to the radiative forcing of the atmosphere. Thus, the establishment of a simple and reliable technique to quantify gas emissions from STW is needed to determine and compare their potential environmental impact with alternative systems. The aim of this study is to establish a sampling and analysing method to determine GHG emissions from STW. The methodology is then applied to measure the spatial and temporal variation of methane and nitrous oxide emissions in a full-scale system. Finally, the atmospheric impact of STW is compared to that of conventional sludge management adopted in small communities.

Materials and methods

Site description and sludge characterisation

La Guixa is a small wastewater treatment plant (1,000 Population Equivalent, PE) located in the province of Barcelona (Spain) which treats 100 m^3/d of urban wastewater in an activated sludge with extended aeration system. In this facility, 5 wetlands with a total surface of 210 m^2 were established in 2007 to treat wasted activated sludge. The wetlands were planted with Phragmites australis (common reed) with a density of 4 plants/m^2. STW are fed with thickened sludge from a pipe located along the shorter side with and annual loading rate of approximately 20 kgTS/m^2·y.

The wetland selected for the study has a surface of 37.1 m^2, with a width of 5.3 m and a length of 7 m. During the experimental period (3 weeks), the wetland was fed once per week with 240 m^3 of sludge.

The properties of influent sludge and sludge stored in the wetland were determined in one sampling campaign carried out according to Uggetti et al. (2009). Core samples were collected from three points along wetland length (inlet, middle and final zone) at two depths, corresponding to the surface and bottom layers.

Samples were analysed in triplicate according to the Standard Methods (APHA-AWWA-WPCF, 2001). In order to evaluate sludge quality, the following parameters were determined: Total Solids (TS), Volatile Solids (VS) and Chemical Oxygen Demand (COD). COD analysis were conducted on sludge dried at room temperature until a constant weight was obtained, therefore results are expressed on a dry matter bases (kg TS).

Measurements of gas emissions

The methodology applied to measure gas emission is based on the static chamber technique used for natural ecosystems containing emergent macrophytes (Crill et al., 1991, Duchemin et al., 1999). In this technique, chambers are used to close the sampling area to the atmosphere to collect the gas emitted during a period established.

Sampling campaigns were conducted in the selected wetland in July 2010. Samples were collected always between 10 and 14 am. Two different experiments were carried out in order to see the possible spatial and the temporal variation in the GHG emissions.

In order to measure the spatial variability, a mesh of nine points was selected along three transects corresponding to the inlet, middle and final zone. Samples were collected in every point at the same time my means of nine chambers. The temporal variation of GHG emission was studied during three loading periods, by sampling the nine points before loading, immediately after loading and every 24 hours for four days, as specified in Table 8.1.

Table 8.1. Sampling timing.

Time (day)	Sample number	Note
1	1	Before loading
	2	Immediately after the first loading (15,2 kgTS)
2	3	1 day after the first loading
3	4	2 days after the first loading
4	5	3 days after the first loading
5		No sampling
6	6	Before the second loading
	7	Immediately after the second loading (15.2 kgTS)
7	8	1 day after the second loading
8	9	2 days after the second loading
9	10	3 days after the second loading
10	11	Immediately after the third loading (15.2 kgTS)
11	12	1 day after the third loading
12	13	2 days after the third loading
13	14	3 days after the third loading

PVC chambers with a diameter of 50 cm, height of 80 cm and a volume around 130 L were applied. A removable lid provided with a rubber septum allowed gas sampling using syringes; while a thermometer positioned into the septum recorded the chamber temperature (Figure 8.1). Each chamber was equipped with a fan to ensure thorough gas mixing. Additionally, a thin tube (inner diameter of 0.3 cm) placed in the septum prevented development of underpressure in the chamber during gas sampling.

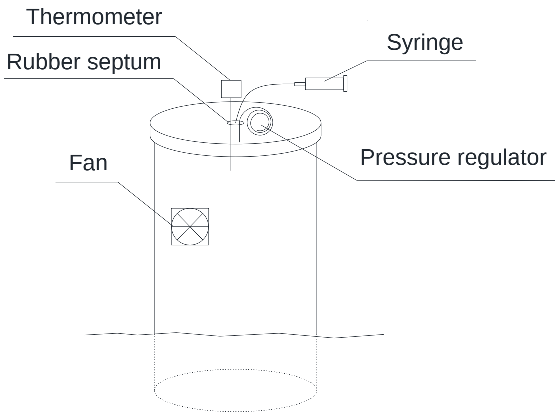


Figure 8.1. Gas sampling system.

The chambers were pressed into the sludge to a depth around 10 cm to ensure airtightness. Each chamber remained positioned without lid in the same sampling point during the whole experimental period, in order to prevent sludge and vegetation disturbance. The fans were running during all the time, and the lid was closed only during the sample collection.

Before starting measurements, each chamber was flushed by means of an additional fan, in order to remove the possible gas accumulated in the open chamber. The lid was then fit, bending accurately the highest plants inside the chambers. Gas samples (30 ml) were collected 0, 5, 10, 20 and 40 minutes after chamber closure by means of polypropylene syringes. It is important to consider that sampling time might vary depending on the chambers' volume and emissions estimated. Thus, pre-tests were run in order to determine the optimal sampling time.

Samples, stored in evacuated glass vials (EXETAINER, Labco Ltd., UK), were subsequently analysed with GC (6890N, Agilent Technologies, USA) equipped with flame ionization detector (FID) for CH_4 and electron capture detector (ECD) for N_2O. The gas concentration was calculated in accordance with three standards containing (I) 1.98 ppm CH_4 and 0.389 ppm N_2O, (II) 15 ppm CH_4 and 3 ppm N_2O, and (III) 50 ppm N_2O, respectively.

Gas fluxes were calculated from the linear increase of gas concentration in the headspace of the chamber within the measuring time of 40 minutes. The increase rate of gas concentrations in each chamber was calculated from the slope of the linear regression for the concentration versus time. The emission rates (Eq. 1) are calculated by taking account the chamber's surface area ($mg/m^2 \cdot d$).

$$Gasemission\left[mg/m^2 \cdot d\right] = \frac{slope[mg/d]}{chambersurface\left[m^2\right]} \qquad \text{Eq. 1}$$

Data analysis

The Minitab 16.0 statistical package was used for the statistical analysis of experimental data. ANOVA tests were conducted on gases emissions from each sampling day, in order to study the statistical significance of the differences between measurements of the same day from different transects. The normality of variable distribution was checked using the Kolmogorov-Smirnov test. If data distributions differed from the normal, Spearman rank-order correlation was performed.

Results and discussion

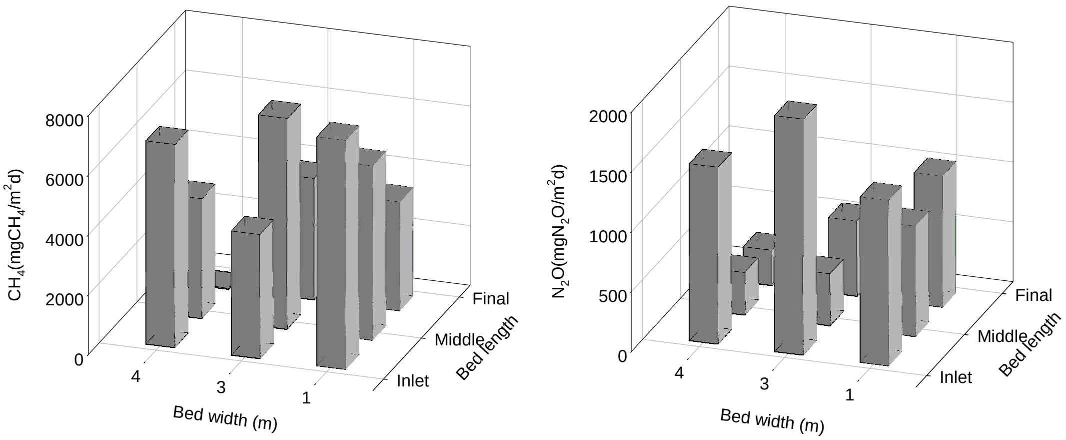
Adaptation of the static chamber method to STW

The static chamber method was mostly successfully adapted to the determination of gas emissions from STW. In some cases, mainly for CH_4, the increase in gas concentration in the chamber was not linear within time. The probable reason for the high methane concentration in the beginning of the measurement was the release of bubbles during lid placement. The bubbles have high methane concentration but low nitrous oxide concentration, therefore bubbling is causing more problems with methane. Hence, results like here with low r^2 (<0.8) were discarded.

In this study methane emissions ranged from 10 to 5,400 $mgCH_4/m^2 \cdot d$, being in the range reported for other systems like wastewater constructed wetlands. According to Søvik and Kløve (2007) emissions from wastewater constructed wetlands located in the North Europe range from -32 to 38,000 $mgCH_4/m^2 \cdot d$. Nitrous oxide fluxes measured in this work (20-950 $mgN_2O/m^2 \cdot d$) also fall in the range from -2 to 1,000 $mgN_2O/m^2 \cdot d$ reported by Søvik and Kløve (2007) for wastewater constructed wetlands. Similar values (560 to 1070 $mgN_2O/m^2 \cdot d$) have been found for agricultural soils (Maljanen et al., 2007) and in some other European nitrogen loaded ecosystems, like riparian zone (1,050 $mgN_2O/m^2 \cdot d$, Marchefelt et al., 2002).

Spatial variation in gas emissions

Figure 8.2a represents the CH_4 emissions after the first feeding event. Even if the values were highly variable, ranging between 3,600 and 7,600 $mgCH_4/m^2 \cdot d$, the distribution was rather uniform over the wetland, due to the uniform spreading of the influent sludge facilitated by the small dimensions of the wetland studied. In such system the sludge fed from one side of the wetlands, is rapidly distributed over the wetland. Only one corner of the final zone (Figure 8.2a) had significantly lower emissions due to the low amount of sludge received. Even sludge spreading can be obstructed by plant cover or roots.



a) b)

Figure 8.2. Examples of spatial distribution of CH_4 (a) and N_2O (b) emissions.

N_2O emissions shown in Figure 8.2b ranged between 300 and 1,900 $mgN_2O/m^2 \cdot d$. Again, the variation between points was quite high. However, as for methane the distribution of nitrous oxide fluxes did not follow any clear pattern along the wetland, both high and low values were detected all over the wetlands. Similarly, high spatial variations were recorded in previous studies on wastewater horizontal flow constructed wetlands (Søvik and Kløve, 2007; Tanner et al., 1997; Picek et al., 2007). However, in these studies, CH_4 emissions were significantly higher near the wastewater inlet. This could be due to the different operational characteristics between the sludge and wastewater CWs. In STWs the sludge is distributed in few minutes along the wetland while in wastewater horizontal CWs wastewater flows slowly from the inlet to the outlet and most of the organic load is degraded near the inlet zone. This hypothesis is supported by the sludge characterisation (Table 8.2), which shows almost constant TS, VS and COD in the different transects of STWs. This suggests homogeneous distribution and degradation of the sludge along the STW.

Table 8.2. Sludge properties (mean±s.d.).

Sample zone	Sample depth	TS (%)	VS (% TS)	COD (g/kgTS)
Influent		0.38 ± 0.01	52.45 ± 0.3	646 ± 20
Inlet	Surface	17.7 ± 1.89	51.18 ± 2.45	700 ± 60
	Bottom	30.19 ± 2.12	43.59 ± 3.28	645 ± 40
Middle	Surface	18.39 ± 0.9	50.34 ± 0.69	590 ± 10
	Bottom	29.51 ± 0.64	43.29 ± 4.75	570 ± 10
Final*		32.05 ± 2.94	49.55 ± 0.3	640 ± 40

*In the final zone only one layer was considered due to the thickness of the sludge

Table 8.3. Analysis of variance comparing CH_4 emissions between transects (T1, T2, T3 refer to inlet, middle and final transects).

Sample number	CH_4 emissions (mean ± s.d.)			P-value
	T1	T2	T3	
1	53 ± 60	19 ± 15	7 ± 6	0.329
2	6197 ± 1831	5621 ± 1533	2573 ± 2219	0.113
3	2897 ± 646	1420 ± 431	1278 ± 1254	0.107
4	3766 ± 926	2752 ± 1018	3054 ± 2979	0.804
5	3861 ± 1805	2825 ± 560	1713 ± 2483	0.416
6	5506 ± 4871	442 ± 373	4442 ± 6517	0.432
7	1224 ± 452	100 ± 403	1951 ± 1164	0.339
8	3452 ± 2778	644 ± 273	2398 ± 1791	0.269
9	6417 ± 6012	1222 ± 782	6024 ± 5230	0.371
10	14093 ± 12953	3233 ± 1798	12733 ± 11112	0.401
11	1868 ± 487	2057 ± 563	3183 ± 2085	0.446
12	5843 ± 4254	4674 ± 1779	8643 ± 9205	0.716
13	5703 ± 1984	3556 ± 313	4265 ± 4887	0.696
14	5999 ± 2644	5787 ± 1653	9940 ± 8156	0.557

Sludge characterisation (Table 8.2) indicates that the studied STWs are efficient for sludge dewatering and stabilization. Influent sludge had a low TS content (0.4 %) and organic matter concentration (52 %VS/TS and 664 gCOD/kgTS) as a result of the high retention time of solids in the wastewater treatment process. Sludge dewatering during the treatment is demonstrated by the increase in TS within the wetland (up to 30% in the bottom layer), while organic matter mineralization was observed from the decrease in VS and COD. Indeed, VS around 40% in the bottom layer indicate a good degree of stabilization. Values of TS, VS and COD observed in the sludge stored in the wetland are in accordance with values reported for conventional dewatering and stabilization treatments like centrifugation and digestion (Uggetti et al., 2010).

The homogeneity between transects suggested by the sludge analysis, was confirmed by the gas emissions. The one way ANOVA was performed on the data collected in each day of sampling. The differences between mean emissions from transects were tested (Table 8.3 and 8.4). In spite of the variability between sampling points, there were no statistical significant differences in the CH_4 or N_2O emissions between transects. As commented above this is probably attributed to the small dimension of the wetland studied (37.1 m^2).

Table 8.4. Analysis of variance comparing N_2O emissions between transects (T1, T2, T3 refers to inlet, middle and final transects).

Sample number	N_2O emissions (mean ± s.d.)			P-value
	T1	T2	T3	
1	1608 ± 316	568 ± 308	671 ± 403	0.019
2	683 ± 550	515 ± 174	609 ± 476	0.926
3	540 ± 502	878 ± 529	367 ± 240	0.457
4	394 ± 204	696 ± 389	432 ± 164	0.391
5	465 ± 261	1004 ± 527	778 ± 369	0.324
6	2083 ± 1910	2691 ± 1353	1832 ± 1902	0.829
7	612 ± 516	675 ± 666	4738 ± 8094	0.508
8	485 ± 1146	1092 ± 1212	2221 ± 3775	0.681
9	262 ± 637	566 ± 820	1858 ± 3209	0.594
10	510 ± 626	943 ± 1291	1920 ± 3318	0.721
11	102 ± 164	121 ± 48	56 ± 88	0.766
12	6 ± 71	129 ± 153	127 ± 198	0.55
13	14 ± 39	106 ± 163	116 ± 194	0.76
14	1 ± 49	30 ± 30	6 ± 4	0.532

Temporal evolution in gas emissions

In the first feeding event the CH_4 fluxes (Figure 8.3a) were low before feeding (around 10 $mgCH_4/m^2 \cdot d$) but increased rapidly after sludge loading (up to 5,400 $mgCH_4/m^2 \cdot d$ immediately after the first loading). During the following days (after 24, 48 and 72 hours) emissions decreased, varying between 2,000 and 3,000 $mgCH_4/m^2 \cdot d$. This pattern was slightly different in the other two feedings where emission peaks were detected 48h and 72h after the feeding (samples 9 and 14) with values between 4,000 and 6,000 $mgCH_4/m^2 \cdot d$ after the second and third feedings, respectively.

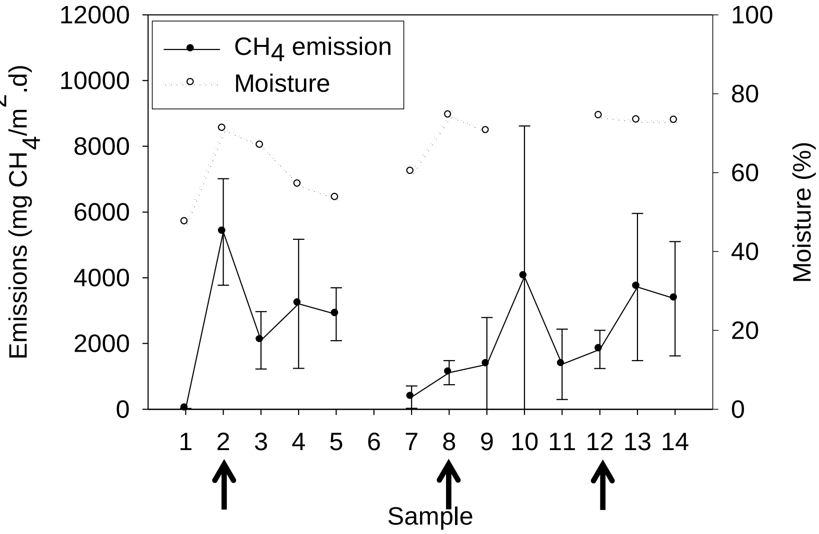
These results highlight the rise in CH_4 fluxes as a consequence of sludge loading which serves fresh organic matter for anaerobic microbial decomposition processes including methanogenesis as a terminal processes (Whiting and Chanton, 1993, Tanner and Sukias, 1995, Nykänen et al., 1998).

In addition, humidity greatly affects methane emissions determining the extent of oxic and anoxic microbial processes. According to Moore and Dalva (1993) and Grünfeld and Brix (1999) high water table favours generally the strict anaerobic CH_4 production in wetlands. However, Grünfeld and Brix (1999) demonstrated that, in CWs for wastewater treatment, a slight decrease in water table has minor effect on methane emissions. This conclusion is supported by the results here. Even if the increase in the methane emissions after the third feeding period (sample 11-14 Figure 8.3a) was associated with the increase in the sludge moisture (above 70%), no statistically significant correlation between the CH_4 fluxes and humidity was observed (Table 8.5). The difference in the methane peak between feeding could be largely due to the change in pH, substrate availability and temperature, as these parameters strongly affect methane production (Neue et al., 1997).

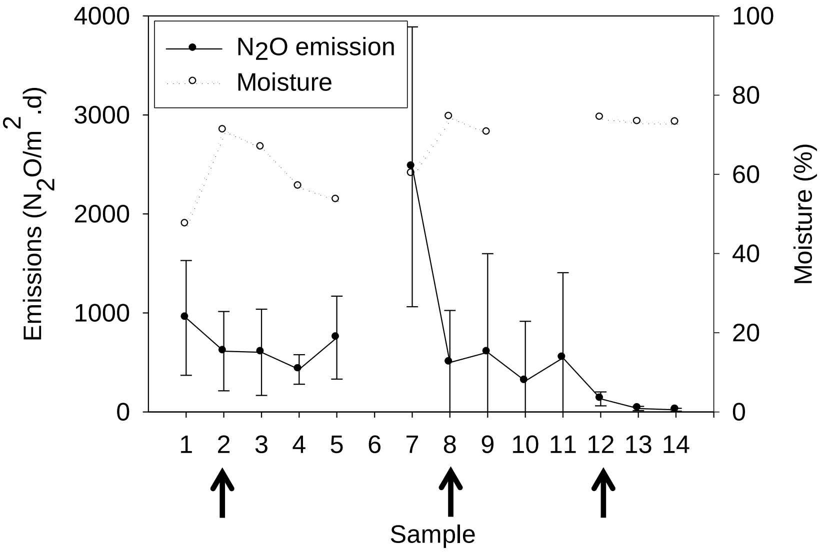
Table 8.5. Analysis of the Spearman correlation.

Parameters	Pearson correlation	P-value
CH_4 vs. N_2O emissions	-0.601	0.039
CH_4 emissions vs. humidity	0.273	0.391
N_2O emissions vs. humidity	-0.678	0.015

The N_2O emissions, (Figure 8.3b) showed an opposite trend than the methane emissions. In the first loading event, the high emissions (950 $mgN_2O/m^2 \cdot d$) decreased after feeding and remained almost constant during the following days (between 430 and 750 $mgN_2O/m^2 \cdot d$). During the second and third feedings, there was a decreasing tendency in the N_2O fluxes in contrast to the CH_4 emissions. There the nitrous oxide emissions were high before feeding (2,500 $mgN_2O/m^2 \cdot d$) and then reduced by sludge loading (to 500 $mgN_2O/m^2 \cdot d$ or even 20 $mgN_2O/m^2 \cdot d$ in the last sample).



a)



b)

Figure 8.3. CH_4 (a) and N_2O (b) emissions together with the moisture variation during the sampling period (mean±s.d.). Notice that outliers were removed for the calculation. Feeding events are indicated by the arrows.

Nitrous oxide production is affected by oxygen availability, as N_2O is produced as an intermediate in denitrification or as a by-product of nitrification. Biological denitrification is an anaerobic heterotrophic microbial process, which is commonly regarded as the dominant process responsible for the N_2O production in constructed wetlands (Kadlec and Knight, 1996). However, nitrous oxide production is highly dependent on nitrification, an aerobic chemoautotrophic process, which produces nitrate from ammonium. In this work, the addition of fresh sludge, which enhances anaerobic conditions and consequently CH_4 emissions, leads to a reduction in the N_2O emissions. There was a negative correlation between methane and nitrous oxide emissions (Table 8.5). The low nitrous oxide production in anaerobic soil is a result of limited nitrate production leading lack of nitrate needed in denitrification (Martikainen et al. 1993, Regina et al., 1996). In water-saturated highly anaerobic conditions a substantial part of nitrous oxide can also be reduced to N_2 before released from the sediment to the atmosphere.

Global warming potential (GWP)

Methane and N_2O emissions (based on mass units) correspond to 25 and 298 CO_2 equivalents (with time horizon of 100 years, IPCC 2007), respectively. By using the mean emissions in this study, the Global warming potential of methane emission results in 0.07 $kgCO_2eq/m^2 \cdot d$, while nitrous oxide results in 0.16 $kgCO_2eq/m^2 \cdot d$. In spite of the lower N_2O emissions, the atmospheric impact of nitrous oxide was twice of that of methane. Based on the total emissions of CH4 and N20 in this work (0.23 $kgCO_2eq/m^2 \cdot d$) and dimensions of the STW (210 m^2 and 1,000 PE), the Global warming potential of the STW was 17 kg $O_2eq/m^2 \cdot d$. The CO_2 emissions were not considered here. Carbon dioxide produced in organic matter decomposition can be considered to be fixed from the atmospheric CO_2 in photosynthesis and thus be atmospherically neutral (IPCC, 2006).

Centrifugation and untreated sludge transport to a wastewater treatment plant (WWTP) provided with sludge treatment line are here considered as potential alternatives for sludge management in small communities (<2,000 PE), and can be compared with STW in terms of CO_2 equivalent emissions. Their potential emissions (Uggetti et al., 2011) are 36 $kgCO_2eq/PE \cdot y$ (centrifuge) and 162 $kgCO_2eq/PE \cdot y$ (sludge transport to a complete WWTP at a distance of 30 km). The Global warming potential of STW is thus from 2 to 9 times lower than those of the alternative options. Thus, the implementation of STW in small and remote communities would reduce the greenhouse gas emissions from sludge management.

Conclusions

This study focused on the establishment of a simple and reliable method for the determination of methane and nitrous oxide emissions from STW. The methodology was then applied to measure greenhouse gas emissions and to determine GWP from STW. The following conclusions can be drawn from this study.

The static chamber method can be successfully adapted to the determination of gas emissions from STW.

In spite of the spatial and temporal variation in the CH_4 and N_2O emissions in STW, there is no specific difference in the emissions along the wetland resulting from homogeneous sludge distribution and biodegradation in STW.

Aerobic conditions before feeding, characterised by low methane emissions (10 $mgCH_4/m^2{\cdot}d$) and high nitrous oxide emissions (950 $mgN_2O/m^2{\cdot}d$), were strongly changed by fresh sludge feeding which enhances CH_4 emissions (5,400 $mgCH_4/m^2{\cdot}d$) and decreases N_2O emissions (20 $mgN_2O/m^2{\cdot}d$).

The Global warming potential of CH_4 and N_2O emissions from STW correspond 17 $kgCO_2eq/PE{\cdot}y$ which is from 2 to 9 times lower than that of sludge centrifugation and transport.

9. Technical, economic and environmental assessment

This chapter is based on the article:

E. Uggetti, I. Ferrer, J. Molist, J. García (2011). Technical, economic and environmental assessment of sludge treatment wetlands, Water Research, 45 (2), 573-582.

Sludge treatment wetlands (STW) emerge as a promising sustainable technology with low energy requirements and operational costs. In this study, technical, economic and environmental aspects of STW are investigated and compared with other alternatives for sludge management in small communities (<2,000 population equivalent). The performance of full-scale STW was characterised during 2 years. Sludge dewatering increased total solids (TS) concentration by 25%, while sludge biodegradation lead to volatile solids around 45% TS and DRI24h between 1.1-1.4 gO_2/kgTS·h, suggesting a partial stabilisation of biosolids. In the economic and environmental assessment, four scenarios were considered for comparison: 1) STW with direct land application of biosolids, 2) STW with compost post-treatment, 3) centrifuge with compost post-treatment and 4) sludge transport to an intensive wastewater treatment plant. According to the results, STW with direct land application is the most cost-effective scenario, which is also characterised by the lowest environmental impact. The life cycle assessment highlights that global warming is a significant impact category in all scenarios, which is attributed to fossil fuel and electricity consumption; while greenhouse gas emissions from STW are insignificant. As a conclusion, STW are the most appropriate alternative for decentralised sludge management in small communities.

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 in the European Union has increased by 50% since 2005. Therefore, optimisation of sludge management becomes a key element in the wastewater treatment sector.

Secondary sludge consists of excess biomass produced during biological wastewater treatment. It is characterised by high organic matter (50-80% TS) and low dry solids (0.5-2% TS) contents (Wang et al., 2008). According to these properties, sludge treatment processes may be separated into stabilisation and dewatering techniques. Sludge stabilisation aims at reducing the biodegradable fraction of organic matter, thus the risk of putrefaction, while diminishing the concentration of pathogens (Luduvice, 2007). On the other hand, the aim of dewatering is to decrease sludge volume, hence disposal costs and environmental risks associated. Besides, sludge dewatering is required prior to composting, incineration or landfilling.

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 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 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, 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.

Life Cycle Assessment (LCA) is a useful tool for investigating the environmental impacts of a product or system over its whole life cycle. As established by the ISO 14040 and 14044 guidelines (ISO 2006a, ISO 2006b), LCA gives overall information on resource consumption and environmental emissions by including extraction of raw materials, processing, manufacture, use and end of life of a product or a process. The LCA method has been previously used to assess the environmental impact of sewage sludge management scenarios (Suh and Rousseaux, 2002, Lundin et al., 2004, Houillon and Jolliet, 2005, Tarantini et al., 2007) and treatment technologies (Svantröm et al., 2004, Hospido et al., 2005, Peregrina et al., 2006). Besides, LCA studies are at times supported by economic analysis (Murray et al., 2008, Hong et al., 2009).

In this study, the performance of STW is investigated by means of a field study carried out over a period of two years in one STW located in Spain. The system's efficiency is then compared to literature results from conventional treatments for sludge management in small communities (<2,000 PE). Data collected from field campaigns and from the literature are the bases for subsequent economic and environmental assessment, assuming design and operation criteria of full-scale systems located in Spain.

Four scenarios are 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 WWTP without previous treatment. To our knowledge, this is the first time that an economic and environmental assessment of STW is conducted and compared with other alternatives for sludge management. Our aim is to demonstrate the suitability of STW for small communities, from a technical, economic and environmental point of view.

Materials and methods

Sludge treatment wetlands' performance

The performance of STW was studied by monitoring a full-scale facility (1,500 PE) located in Seva, province of Barcelona (Spain). The wastewater treatment line consists of a contact-stabilisation unit. Secondary sludge is stored in a tank and pumped to the STW. In this facility, 7 wetlands were set-up in 2000 by transforming conventional drying beds. The total surface area of the STW is 175 m^2 and the sludge loading rate around 125 kg TS/m^2·year. Each bed is fed semi-continuously during alternate days. Other details on the design and operation of these wetlands may be found in Uggetti et al. (2009a). Operating cycles last on average 5 years; after a resting period of some 4 months, the final product is removed with a power shovel and transported to a composting plant.

During 2 years, 6 field campaigns were carried out in one bed to characterise the properties of the influent and sludge from the wetlands. Composite samples were taken from three sampling points located along the bed. The biosolids obtained (final product) were also characterised in two wetlands of the same facility at the end of the operating cycle after a resting period of 4 months.

Sludge dewatering was determined by the TS concentration, while organic matter was analysed in terms of Volatile Solids (VS) and Chemical Oxygen Demand (COD). The stability of biosolids was measured by the Dynamic Respiration Index (DRI), as proposed by Adani et al. (2000) and Barrena et al. (2009a). Nutrients (nitrogen (NTK), phosphorus (TP) and potassium (K)), heavy metals and faecal bacteria indicators (*Salmonella* spp. and *Escherichia coli*) contents in biosolids were also determined. All parameters were analysed in triplicate following Standard Methods (APHA-AWWA-WPCF, 2001).

Methane (CH_4) and odour emissions from wetlands were measured both after feeding and between feedings; corresponding to the maximum emission rate and the average emission rate, respectively. These measurements were carried out as described by Sarkar and Hobbs (2003). Samples were collected from representative STW by positioning a Linvall Hood of 1m^2 surface area. A controlled airflow (0.1 m/s) was passed over the chamber surface and samples of inlet and exhaust air were collected in Nalophan NA sample bags. Odour concentration was determined according to the European Standard EN13725:2003 (Committee for European Normalization, 2003), as a function of the number of required dilutions to be detectable by 50% of the odour panel. According to this method, the odour concentration is expressed as unit of odour per m^3 of air (ou_E/m^3·s). CH_4 was analysed by gas chromatography (Thermo Finnigan Trace, GC 2000).

Economic evaluation

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. Besides, the need for post-treatment after STW is also accounted for. Consequently, the following scenarios are considered: 1) STW with direct land application of biosolids, 2) STW with compost post-treatment, 3) centrifuge with compost post-treatment, 4) transport to an intensive WWTP. 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. The results are expressed in m^3/d of wastewater treated.

Design and operation criteria of STW located in Spain are adopted (Table 9.1). In this sense, 5 year operating cycles are assumed, although longer operating cycles are reported in other countries like Denmark (Nielsen, 2003). Emptying procedures involve biosolids withdrawal with a power shovel and transport to final destination. STW operation is thereafter restarted without replanting. STW investment costs (Table 9.5) include soil occupation and excavation, wetlands construction, pump and pipe installation, gravel placement and plantation.

Table 9.1. Sludge treatment wetlands' design and operation parameters considered in the economic and environmental assessment (scenarios 1 and 2).

	Wastewater treated		
	500 PE	1000 PE	2000 PE
Population equivalent	500	1,000	2,000
Sludge loading rate (kg TS/m^2·year)	50	50	50
Total surface area (m^2)	167	331	662
Number of beds	4	6	12
Wall height (m)	1.6	1.6	1.6
Gravel volume per wetland (m^3)	16	22	22
Sludge storage capacity per wetland (m^3)	45	59	59
Sludge withdrawn (t)	182	361	724
Operating cycle (years)	5	5	5

Table 9.2 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% TS and 20% TS, respectively (Uggetti et al., 2010).

Table 9.2. Sludge flow rates and emissions considered in the economic and environmental assessment.

	Wastewater treated			Scenario
	500 PE	1000 PE	2000 PE	
Waste activated sludge (sludge generation) (m^3/year)	275	550	1100	1-4
Sludge production in STW (m^3/year)	33	66	132	1-2
Sludge production in centrifuge (m^3/year)	41	82	165	3
Pump electricity consumption in STW (kWh/year)	25	50	105	1-2
Pump electricity consumption in centrifuge (kWh/year)	30	60	125	3
Centrifuge electricity consumption (kWh/year)	140	280	560	3
CH_4 emission rate from STW ($mg/m^2 \cdot s$)	< 88	< 88	< 88	1-2
Odour emissions ($ou_E/m^2 \cdot s$)	5.7-7.3	5.7-7.3	5.7-7.3	1-2

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, which is commonly used in small facilities of the zone (Uggetti et al., 2009). For this reason, the functional unit is defined as the management of 1 ton of sewage sludge (wet weight).

Taking into account the functional unit, the system boundaries are as follows:

a) The wastewater treatment line is not included in the model, because it is the same in all scenarios.

b) 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.

c) 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.

d) Final transport and disposal are not included either, bearing in mind that they would be approximately the same in all scenarios.

e) Raw materials required for systems' construction and energy consumption for systems' operation are taken into account.

f) The 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).

g) The end of life is included for the centrifuge, as it should be replaced over the period considered (20 years). This aspect has not been taken into account for STW since their lifespan is longer than the 20 years period considered in this study.

System boundaries and scenarios defined in the model are shown in Figure 9.1. Inventory data on systems' design and operation are the same as for the economic analysis, collected in full-scale facilities from Spain (Tables 9.1 and 9.2). 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). The results are therefore expressed as a quantification of the potential contribution of materials and processes to each impact category.

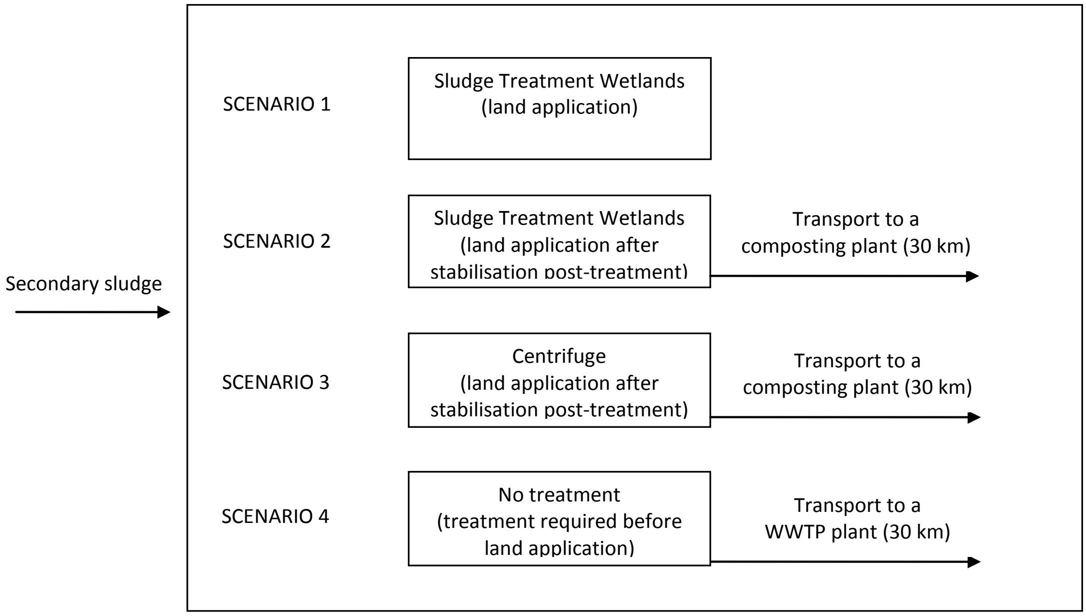


Figure 9.1. System boundaries and scenarios of the Life Cycle Assessment

Results and discussion

Sludge treatment wetlands' performance

The main results from the sampling campaigns of the full-scale STW are summarised in Table 9.3. Notice that campaign VI corresponds to the biosolids obtained at the end of the operating cycle. In general terms, TS increase from 1% in the influent to 15-16% in the wetlands, with a maximum concentration of 25% in the final product after a resting period of 4 months. Dewatering efficiency is generally increased during the summer season, reaching high TS concentration (25%) (campaign III), being generally lower (around 16%) during the rest of the year. Notice that even the lowest dewatering efficiency found in this study (14.8% in autumn) is competitive compared to centrifuges, which are capable of achieving 14-18% TS with conventional waste activated sludge (Gonçalves et al., 2007) and 18% TS with waste activated sludge from extended aeration units (Uggetti et al., 2010).

Organic matter biodegradation is shown by some 10% decrease in VS and COD contents during sludge treatment; VS and COD concentrations being quite stable all over the year. On average, during the treatment VS content is reduced from 55% VS/TS in the influent to 48% VS/TS in the wetlands. The same can be said for COD which is decreased from an average value of 770 g/kgTS to 720 g/kgTS. The lowest VS concentration is reached in summer (45% VS/TS), although in this case the values are also a result of the low influent concentration

(39.5% VS/TS). Besides, the seasonal variability that influences the efficiency of the treatment is minimised by the resting period, which enhances organic matter biodegradation before biosolids removal leading to VS over 40% VS/TS and COD around 500 g/kgTS in biosolids (Table 9.3). Final values are within the range obtained after conventional sludge stabilisation techniques, such as anaerobic digestion (Ferrer et al., 2010). However, in compost samples organic contents are usually higher, around 60% VS/TS for compost of sewage sludge mixed with vegetable wastes (Bertrán et al., 2004), due to humic-like substances produced during composting.

Table 9.3. Sludge characteristics (mean value ± SD) from samples taken during 2 years. Note that Campaigns IV correspond to sludge samples taken during the treatment in wetlands, while Campaign VI refers to the final product after 4 months resting period (biosolids).

		Campaign I (autumn)	Campaign II (spring)	Campaign III (summer)	Campaign IV (winter)	Campaign V (spring)	Campaign VI (autumn)
TS (%)	Influent	1.7±0.01	1.2±0.01	0.3±0.02	1.7±0.06	0.6±0.01	1.1±0.0
	Wetland	16.1±3.3	14.8±2.5	25.7±13.4	14.3±3.03	16.7±4.2	25±1.6
VS (%TS)	Influent	57.7±0.9	59.0±0.7	39.5±4.7	58.7±0.2	45.7±5.5	51.5±0.8
	Wetland	47.7±3.5	50.2±3.2	45.0±4.4	49.5±4.0	48.9±5.8	43.7±7.1
COD (g/kg TS)	Influent	940±180	820±110	580±20	740±20	780±60	709±11
	Wetland	880±440	660±100	670±70	620±200	720±90	520±50

Even if the properties of biosolids in terms of total solids and organic matter suggest their suitability for land application, the need for post-treatments depends on the stability and hygienisation degree of the final product. Biological stability determines the extent to which readily biodegradable organic matter has been decomposed (Lasaridi et al., 1998). For agricultural uses, higher biological stability implies lower environmental impacts (like odour generation, biogas production, leaching and pathogen's re-growth) during land application of the product (Muller et al., 1998). The DRI is based on the rate of oxygen consumption and is a useful indicator of the biological stability of a sample: lower oxygen consumption (DRI value) corresponds to higher biological stability. In this study, the DRI_{24h} from STW biosolids ranged between 1.1 and 1.4 gO_2/kgTS·h. Such a stability degree is much higher than the values reported for a mixture of primary and activated sludge (6.7 gO_2/kgTS·h) and for anaerobically digested sludge (3.7 gO_2/kgTS·h) (Pagans et al., 2006). Values around 1 gO_2/kgTS·h are found in compost (Ponsá et al., 2008) and partially digested material (Scaglia and Adani, 2008). Biosolids from the studied STW achieve almost the same stabilisation degree as compost; therefore by prolonging the resting time to ensure stabilisation, the final product could be valorised in agriculture without post-treatment in a composting plant.

Concerning the main nutrients, a certain amount of nitrogen (4.4% TKN/TS) is found in biosolids, indicating the potential use of the final product as organic fertiliser. However, the concentration of phosphorus (0.26% TP/TS) and potassium (0.15% K/TS) are relatively low. For compost of sewage sludge, Bertrán et al. (2004) give slightly lower nitrogen (2.5%TS) but higher phosphorus (2.3%TS) contents. In general, sludge is characterized by a considerable variability in nutrient's content, depending on the wastewater source and treatment process (Moss et al., 2002). The concentration of nutrients is needed to ensure appropriate dosages of the sludge prior to land application.

On the other hand, the main hazard associated to sludge application on agricultural soils is the potential long term accumulation of toxic elements (Singh and Agrawal, 2008), which may then be uptaken by crops. Such elements include both inorganic pollutants, like heavy metals, and organic micropollutants. Currently, only heavy metals concentrations are regulated for land application of sewage sludge (Council of the European Union, 1986). Since treated sludge may have considerable amounts of pathogens, depending on the treatment processes used, limit values for faecal bacteria indicators have also been proposed (Environment DG, EU, 2000). According to this proposal, sludge shall not contain *Salmonella* spp. in 50 g and that *E.coli* concentration has to be less that 500 MPN/g

The concentration of heavy metals and faecal bacteria indicators in STW biosolids are compared to the limit values for unrestricted land application according to current legislation (Council of the European Union, 1986) and more restrictive values proposed (Environment DG, EU, 2000) (Table 9.4). Notice that there are only little differences between influent sludge and the final product with regards to heavy metals, suggesting that heavy metals accumulation is negligible. Furthermore, in all cases the concentrations are clearly below the limits proposed.

With regards to pathogens, *Salmonella* spp. was not detected, but small quantities *E. coli* were present in all cases (Table 9.4). Both faecal bacteria indicators are well below the limits proposed. On the whole, the characteristics of biosolids analysed (total solids, organic matter contents and nutrients) put forward their suitability for land application especially as organic amendment. Moreover, according to the concentration of heavy metals and faecal bacteria indicators, biosolids from STW studied in this work fulfil the requirements for agricultural application. Nevertheless, biosolids from STW are post-treated in composting plants before agricultural re-use in the case of facilities located in our zone, while they are directly spread on fields in countries like Denmark or France (Nielsen and Willoughby, 2005; Liénard et al., 2008).

Table 9.4. Concentration of heavy metals and faecal bacteria indicators in the influent and biosolids from sludge treatment wetlands (STW).

Parameter	Influent	STW	Council Directive 86/278/EEC limits	Environment DG,EU,2000 proposed limits
Heavy metals				
Cr (ppm)	51	57	-	800
Ni (ppm)	39	31	400	200
Cu (ppm)	252	265	1,750	800
Zn (ppm)	719	615	4,000	2,000
Cd (ppm)	1.7	0.8	-	5
Hg (ppm)	<1.5	<1.5	-	5
Pb (ppm)	53	75	1,200	500
Faecal bacteria indicators				
Salmonella spp. (presence/absence in 25g)	Absence	Absence	-	Absence in 50g
E. coli (MPN/g)	<3	<3	-	<500 MPN·g^{-1}

Economic evaluation

The technical analysis of STW demonstrates that the efficiency of such a technology is comparable to that of conventional treatments in terms of sludge dewatering and stabilisation. Furthermore, the stability index observed suggests that biosolids can reach a high stabilisation degree if sufficient resting time is left at the end on each operating cycle. This means that biosolids' post-treatment is not needed before agricultural application. Nevertheless, it has been included in the economic and environmental assessment to compare the impacts of STW with and without of post-treatment (scenarios 1 and 2), versus conventional treatments (centrifuge) (scenario 3) and transport to an intensive WWTP with sludge treatment line (scenario 4).

The most significant costs of the centrifuge (Table 9.5) include machine assembly and installation, room construction and polyelectrolyte preparation. Notice that STW investment costs increase with the treatment capacity, from 50,000 to 160,000 € for 500 and 2,000 PE systems, respectively. On the other hand, centrifuge costs increase only slightly, from 75,000 to 97,000 €. Therefore, the difference between investment costs is more evident for 2,000 PE facilities; with centrifuges becoming more competitive.

Table 9.5. Investment and operation costs for all scenarios expressed in €/year: (1) sludge treatment wetlands (STW), (2) STW + compost, (3) centrifuge + compost and (4) transport to wastewater treatment plant (WWTP).

		Cost (€/year)		
		500 PE	**1000 PE**	**2000 PE**
Scenario 1	STW investment costs	50.563	83.606	159.442
	Personnel costs	1.125	1.830	2.840
	Materials replacement	502	801	1.304
	Emptying procedure, biosolids transport and agriculture application	1745	3468	6948
	Total investment and operation cost	53.935	89.705	170.534
Scenario 2	STW investment costs	50.563	83.606	159.442
	Personnel costs	1.125	1.830	2.840
	Materials replacement	502	801	1.304
	Emptying procedure, biosolids transport and compost post-treatment	2.581	5.131	10.277
	Total investment and operation cost	54.771	91.368	173.863
Scenario 3	Centrifuge investment costs	74.557	76.007	96.587
	Personnel costs	2.256	4.512	5.716
	Materials replacement	1.160	1.560	2.100
	Sludge treatment, transport and compost post-treatment	2.839	4.786	8.976
	Electricity	427	541	804
	Total investment and operation cost	81.232	87.406	114.183
Scenario 4	Personnel costs	540	1.080	1.350
	Transport	5.700	9.990	17.760
	Sludge treatment in WWTP	3.296	6.592	13.185
	Total cost	9.536	17.662	32.295

Regarding operation costs, there are little differences between STW with direct land application (scenario 1) and with compost post-treatment (scenario 2) (Table 9.5). Note that only one year cost is considered. However, this difference increases with the treatment capacity, from 500 PE (1,000 €) to 2,000 PE (5,000 €); which is attributed to the higher cost of composting (35 €/ton) with respect to the agriculture application canon (12 €/ton) in our zone (Catalonia, Spain). In all cases, centrifuge operation costs are higher, increasing with the treatment capacity (from 7.000 to 17.000 € for 500 and 2,000 PE, respectively). Transport (scenario 4), which does not have investment costs, is characterised by the highest operation cost (from 9.000 € for 500 PE up to 32.000 € for 2000 PE).

The economic analysis considering a life cycle of 20 years is shown in Figure 9.2. 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 or 170 m^3/d (0.28 €/m^3 versus 0.24 €/m^3). Likewise, STW costs are 0.05-0.07 €/m^3 lower than transport. 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 generally 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 depend on local circumstances, including the costs and taxes of energy in each country, 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.

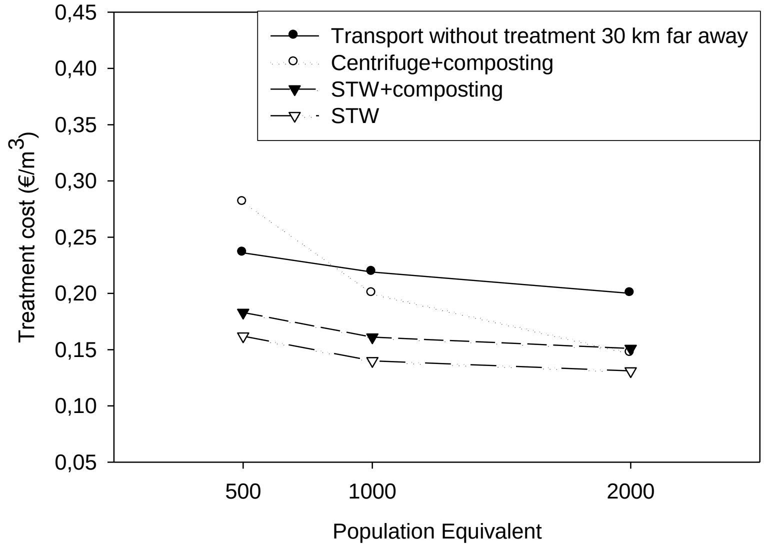


Figure 9.2. Investment and operation costs over a 20 years period of all scenarios: (1) sludge treatment wetlands (STW), (2) STW + compost, (3) centrifuge + compost and (4) transport to wastewater treatment plant.

Life cycle assessment

In LCA analysis the environmental impacts attributed to materials or processes are grouped according to the so-called impact categories. Figure 9.3 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 Figure 9.3 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 scenario 1, the biggest impact is caused by raw materials employed in system's construction; while direct greenhouse gas emissions (Table 9.2), as well as indirect emissions derived from energy consumption and transport, have a smaller contribution. 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.

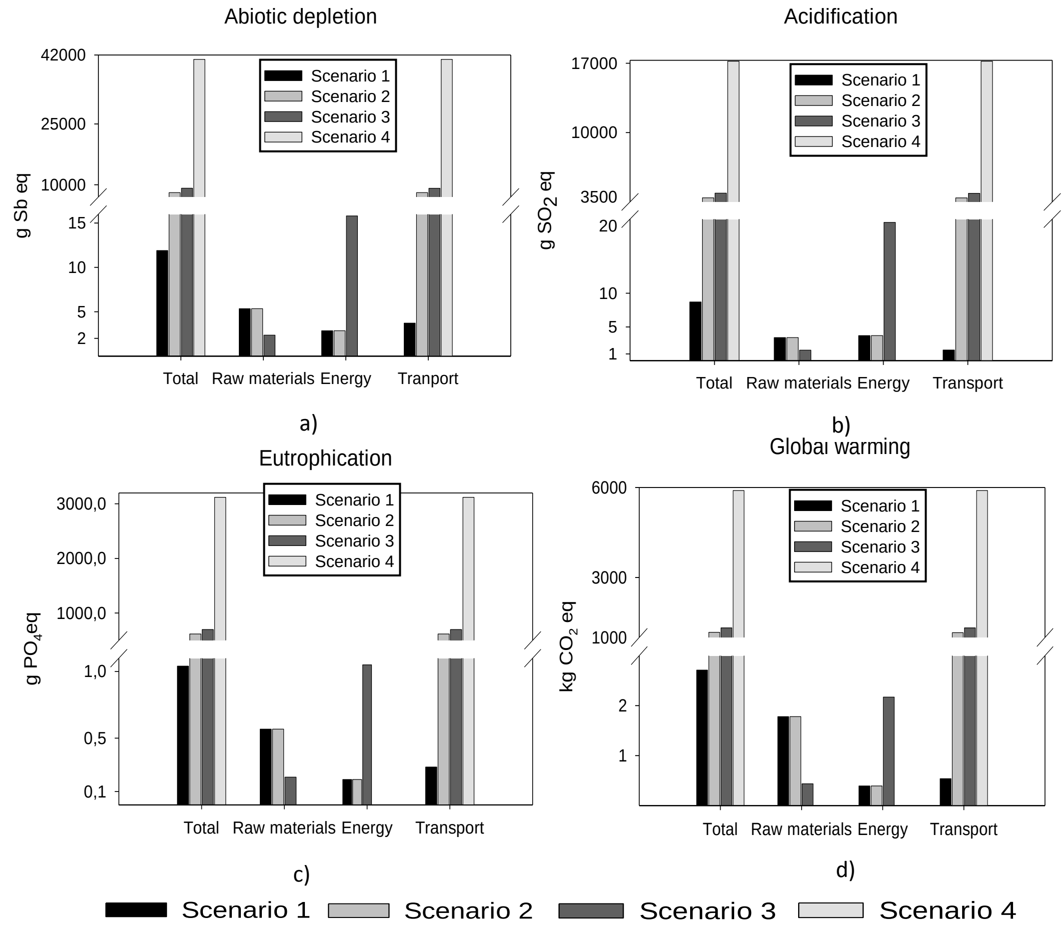


Figure 9.3. Life Cycle Assessment results grouped according to CLM 2 impact categories for all scenarios: (1) sludge treatment wetlands (STW), (2) STW + compost, (3) centrifuge + compost and (4) transport to wastewater treatment plant.

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 9.2).

If we look at individual contributions of raw materials, energy and transport within each scenario (Figure 9.3), 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 2, 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 between sludge management alternatives is that the solution must be appropriated to the local conditions of each site.

Conclusions

This study looked at technical, economic and environmental aspects of sludge treatment wetlands for small communities (500-2,000 PE). The system was then compared with conventional treatments for sludge management. From this evaluation, the following conclusions can be drawn:

In STW, sludge dewatering and stabilisation result in biosolids with around 25% TS and 40-45% VS/TS; with DRI_{24h} between 1.1-1.4 $gO_2/kgTS \cdot h$, indicating a partial stabilisation of the sludge treated and suggesting that with sufficient resting time the final product could be valorised in agriculture without post-treatment in a composting plant.

According to the economic and environmental assessment, STW with direct land application is the most cost-effective scenario, which is also characterised by the lowest environmental impact (almost negligible in comparison with the other options evaluated). If compost post-treatment is required, the costs increase only slightly but environmental impacts increase significantly. Centrifugation costs are higher than STW for systems up to 1,800 PE, but become similar for 2,000 PE systems. However, environmental impacts are higher in all categories compared to STW with direct land application. Sludge transport to external treatment is always the most expensive and environmentally unfriendly scenario.

The LCA highlights that in all scenarios global warming has a significant impact, which is attributed to fossil fuel and electricity consumption; while gases 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.

10. Discussion

In this chapter, results from the different investigations carried out during this thesis are gathered together and discussed. An overview of fundamentals of sludge dewatering and stabilisation processes in STW is given here. Furthermore, design factors and operational criteria deduced from literature and from the results of this thesis are exposed in order to obtain design criteria for STW construction and operation in small Mediterranean WWTP.

Dewatering and stabilisation fundamentals

The interest in studying STW in the Mediterranean region originates from the lack of experience in STW technology in this climatic zone. Thus, as a first point, technical aspects of sludge dewatering and stabilisation in STW were investigated in full-scale systems and in a pilot plant (Chapters 4 and 5).

As presented in the introduction of this thesis, the main sludge treatments can be grouped into dewatering techniques and stabilisation processes. Dewatering aims at decreasing sludge humidity, and consequently its volume and disposal risks and costs. While stabilisation processes were developed with the purpose of reducing the biodegradable fraction of the organic matter as well as the concentration of pathogens. As extensively mentioned in this thesis, both sludge dewatering and stabilisation take place in STW, thus technical aspects of these processes are here recopilated, with special focus on their development in STW.

Sludge dewatering

According to Tsang and Vesilind (1990), moisture in sludge is distributed as: a) free moisture, that is not attached to the solid particles and includes void water not affected by capillary force; b) interstitial moisture, that is trapped within the flocs of solids when sludge is in suspension or is present in the capillaries when a cake is forms; c) surface moisture, that is held on the surface of the solid particles by adsorption and adhesion; d) intercellular moisture, that is chemically bound to the soil particles (Figure 10.1).

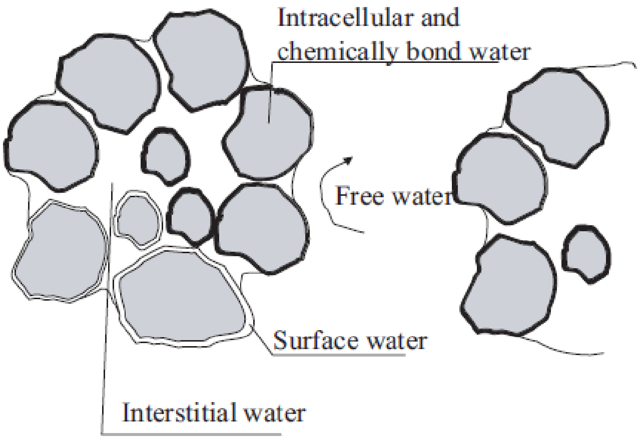


Figure 10. 1 Water distribution on sludge (Chen et al. 2002).

Free water can be removed by simple gravitation action; this is what happens in gravity thickeners, where a 2% TS sludge influent is increased to a 5% TS sludge effluent, leading to a sludge volume reduction of 60% or more. According to Gonçalves et al. (2007), another example of free water removal is the initial stage of sludge dewatering in drying beds, characterised by a rapid water loss due to percolation.

On the other hand, interstitial and surface water demand considerably larger forces to be separated from the solids in sludge. These forces can be either chemical, when flocculants are used, or mechanical, when mechanical dewatering processes such as filter presses or centrifuges are employed. Within such processes solids content higher than 30% may be achieved, resulting in a final semi-solid product known as cake. The removal of free, interstitial and surface water from a sludge originally characterised by 2% TS, may result in 90-95% reduction of the original volume.

Intercellular water, as part of the solid phase, can only be removed through thermal forces that lead to the change in the state of aggregation of the water. Freezing and evaporation are two different possibilities for cellular water separation.

Overall, the amount of the water that can be removed depends on the dewatering process and also the status of the water in the sludge.

In STW, between sludge feeding events, dewatering is carried out basically as the combination of two mechanisms: water percolation through the granular medium and water evapotranspiration (which combines the effect of evaporation and transpiration of plants). Taking into account the water distribution in sludge (Figure 10.1), water percolation might correspond to the free water loss. In fact, immediately after feeding, water is rapidly collected by the drainage pipe located on the bottom of the granular filter. In the pilot plant studied in this thesis (Chapter 5) the major leachate quantity was collected within the first 24h after feeding. Leachate quantity varied significantly according to the season; almost half of the water fed was percolated in winter and spring, while no leachate was collected in summer, when all the water was lost by the high plants evapotranspiration. In WWTP, leachate is normally returned to headwork, thus its quality and quantity should be controlled.

Rapid water percolation is followed by plants evapotranspiration (ET), which provides for additional water loss. In this way, an additional part of free water can be lost, together with some quantity of the bound water. ET is actually known as the major component of the water balance of many different types of wetlands ecosystems (Zhou and Zhou, 2009). Evapotranspiration rate is a function of solar radiation, temperature and wind speed; thus it strongly dependents on the climate. For this reason, ET relevance in sludge dewatering may vary seasonally and even daily. As mentioned above, in the Mediterranean region, during summer the significance of ET rate makes it the most important way of water loss in STW.

Dewatering mechanisms put forwards the relevant role of plants in these systems. Plants contribute mainly enhancing evaporation (from sludge and plant surface) and transpiration (internal plant evaporation). On the other hand, the extended root systems that is developed through the sludge and the granular layer, is important to maintain the capillarity connection within the vertical profile. In fact, roots create a channel system improving water circulation and avoiding hydraulic failures which can negatively affect sludge dewatering (Nielsen 2005). In Chapter 5, the plants relevance was confirmed by the better dewatering performances found in the warmer season and durignthe second year of operation, due to the enhacement of plant activity. In addition, plants' movement cracks the sludge surface, increasing water evaporation and oxygen transfer.

The complexity of sludge percolation mechanisms and evapotranspiration rate determination complicate the understanding of the dewatering process in STW. Moreover, their relative importance in sludge dewatering are strongly climate depending and, to our knowledge, no experimental studies have been carried on this aspect. However, in the present work, efforts have been done in order to perform a model predicting sludge dewatering in STW (Chapter 6). The model reflects the rapid water drainage immediately after sludge feeding, followed by plant evapotranspiration during the next days; pattern that was previously described in a semi-empirical equation proposed by Giraldi et al. (2009b).

Sludge stabilisation

In wastewater treatment, the biological stabilisation of the organic matter is carried out through the action of bacteria in contact with the sludge, which develop in conditions that are favourable for their growth and reproduction. Organic matter digestion can be anaerobic, aerobic or a combination on both.

Anaerobic digestion is a multi-stage biochemical process capable of stabilising different types of organic matter. The process normally occurs in three stages: 1) the enzymes break down complex organic compounds, such as cellulose, proteins and lipids, into soluble compounds, such as fatty acids, alcohol, carbon dioxide and ammonia; 2) microorganisms convert the first-stage products into acetic and propionic acid, hydrogen, carbon dioxide, besides other low-molecular weight organic acids; and 3) two groups of methane-forming organisms take actions: one group produces methane from carbon dioxide and hydrogen, while a second group converts the acetates into methane and bicarbonates.

The organic fraction of the sludge is basically made up of polysaccharides, proteins and fats. As mentioned above, during the anaerobic digestion, colonies of anaerobic microorganisms (hydrolytic, acidogenic and methanogenic organisms) convert the organic matter into cellular mass, methane, carbon dioxide and others macro-constituents. Moreover, sludge digestion significantly reduces the pathogenic organisms, favouring the agricultural use of the sludge.

The efficiency and stability of anaerobic digestion process are variables directly related to the characteristics of the raw sludge and the environment. Normally, the presence of macro and micro nutrients such as nitrogen, sulphur and phosphorus is sufficient for ensuring the development of the anaerobic digestion process.

In aerobic digestion, in absence of substrate supply, microorganism are forced to consume their own protoplasm to obtain energy for cell maintenance reactions. In this way, the biodegradable cell mass is aerobically oxidised to carbon dioxide, ammonia and water. Actually, only about 75-80% of the cell tissue can be oxidised, the remaining 20-25% is composed of inert components and organic compounds that are not easily biodegradable, which will remain in the final product of the digestion. Considering the formula $C_5H_7NO_2$ as representative for cell mass of a microorganism, the biochemical changes in an aerobic digester can be described by the following general equation (10.1):

$$C_5H_7NO_2 + 7O_2 + bacteria \Rightarrow 5CO_2 + NO_3^- + 3H_2O + H^+ \qquad (10.1)$$

As mentioned above, stabilisation processes can combine anaerobic and aerobic digestion. As an example, composting is accomplished mostly under aerobic conditions, even if the process is never completely aerobic. Aerobic composting accelerates material decomposition and results in the higher rise in temperature necessary for pathogens destruction. Moreover, aerobic conditions minimises the potential for nuisance odours. Within the composting process, the biological degradation of the organic matter to a stable end product is carried out by bacteria, actinomycetes and fungi. During the process, approximately 20-30% of the volatile solids are converted to carbon dioxide and water.

Similarly, in wetlands treating wastewater it has been showns that removal of organic matter is mediated by several microbial reactions that can occur at the same time in different locations such as aerobic respiration, denitrification, sulphate reduction and methanogenesis (García et al., 2010). According to greenhouse gasses emissions measured in STW (Chapter 8), organic matter stabilisation in STW is also carried out by different types of microorganisms (anaerobes as well as aerobes). In fact, conditions within STW are widely variable depending on the humidity of the sludge, which is strictly related to the system operation (feeding and resting periods).

The determination of greenhouse gases emissions (Chapter 8) suggests that aerobic conditions before feeding, characterized by low methane emissions (around 10 mg $CH_4/m^2 \cdot d$) and high nitrous oxide fluxes (950 $mgN_2O/m^2 \cdot d$), are strongly affected by fresh sludge feeding. As a consequence of fresh sludge loading, relatively high methane fluxes were detected, as a consequence of the limites oxygen transport caused by the high water content. Moroeverg, processes in anaerobic conditions are enhanced by the addition of nutrients and organic matter enhances decomposition. At the same time, a certain decrease in N_2O flux results from anaerobic conditions promoted by the high water content, which determines the extent of aerobic and anoxic microbial processes. Later on, as the water content decreases, under aerated conditions, methane emissions tend to decrease while nitrous oxide tends to increase due to nitrification.

On the other hand, nitrification is suggested by the decrease of TKN values in the sludge and by the high nitrate concentration within the leachate reported in Chapter 4, confirming the aerobic conditions in STW. Moreover, the low odour emissions reported in Chapter 9, suggest the aerobic condition of the system studied.

The vegetation plays a key role also on this aspect of the treatment; plants contribute to sludge mineralisation through the transport of oxygen from the aerial parts to the belowground biomass. This oxygen is released in the rhizosphere, which creates aerobic microsites in the bulk sludge layer and thus ensures appropriate conditions for aerobic degradation processes and nitrification (Vymazal, 2005). Actually, a direct relation between redox potential and plant density was found by Troesch et al. (2009b) in a STW pilot plant. The development of the root system within the sludge layer is then important to ensure oxygen distribution and to favour conditions for microorganisms' survival along all the vertical sludge profile. Such environment is suitable to enhance organic matter stabilisation within STW.

Plants also contribute indirectly to aerobic mineralisation through stems; which, as a result of their movement enhanced by the wind, crack the surface of dry sludge and prompt aeration of the lower sludge layers. Bottom aeration is, in the same way, improved by the aeration pipes which allow air circulation under the gravel layer.

Design factors

The design factors here exposed are the result of the knowledge acquired during the development of this thesis. Results obtained by the experimental work are here analysed and summarised in order to present some STW design criteria mainly for small Mediterranean communities.

Preliminary statistical analysis

In order to better understand the results obtained from the experimentation on the full-scale systems (Chapter 4), a principal component analysis (PCA) is proposed here.

In PCA, in order to identify possible patterns, a set of raw data is reduced to a number of principal components that retain the most variance within the original data. In this case, the data obtained from the three Catalan full-scale systems were statistically analysed in order to determine the most influent parameters in the determination of the system efficiency.

As shown in Table 10.1, the first two components (PC1 and PC2) summarise almost 90% of the total variance, while the other 3 account for less than 10%. Thus for the interpretation of results only the two first componens will be reained. The first component (67%) is negatively correlated with the total solids (TS) concentration, and positively with volatile solids (VS) and chemical oxygen demand (COD). On the other hand, the second component (22%) is positively correlated with nitrogen (TNK) and negatively with TS.

Table 10.1 Principal component analysis of the three sampling campaigns carried put in the full-scale systems studied.

Eigenvalue	3.3648	1.0936	0.4463	0.0569	0.0384
Proportion	0.673	0.219	0.089	0.011	0.008
Cumulative	0.673	0.892	0.981	0.992	1.000
Variables	**Variable correlation with PC1**	**Variable correlation with PC2**			
TS	-0.439	-0.534			
VS	0.531	0.118			
COD	0.508	0.018			
TNK	-0.262	0.810			
TP	0.445	-0.210			

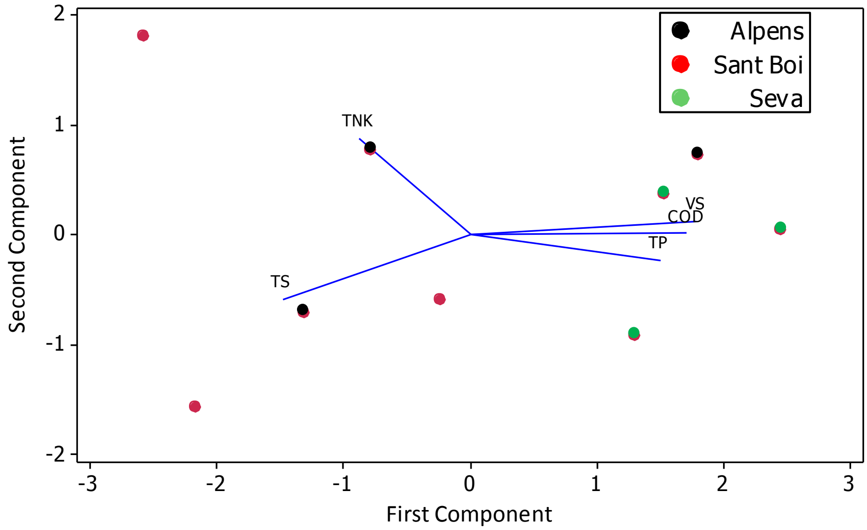


Figure 10.2 Results of PCA relative to the data from the full-scale systems shown in Chapter 4.

The results of the PCA (Figure 10.2) clearly indicate that treatment efficiency is determined by TS, VS, COD and TP. VS, COD and TP follow an opposite trend to TS. The spatial distribution of the facilities studied showed in Figure 10.2 indicates the lower efficiency of the Seva facility in term of both sludge dewatering (low TS concentration) and sludge mineralisation (high VS and COD values). On the other hand, Alpens and Sant Boi de Lluçanès have similar efficiencies, even if the second facility seems to be characterised from better performances (high TS and low VS concentrations in the three campaigns).

PCA analysis confirms the pattern highlighted in Chapter 4. Even if the efficiency of the three systems were satisfactory compared to other technologies, Seva presents lower performances. Taking into account the different configuration and operation of this system with respect to the others, this concern might be considered for the determination of design criteria.

Sludge loading rate

Sludge loading rate is an important parameter for STW design; which determines, as described below, the surface requirement of the systems.

According to the experience found in literature and exposed in Chapter 3, sludge loading rate is usually set from 40 to 60 kgTS/m^2·year, with some exception for warmer climate (Greece, Cameroon, Brazil), where loading up to 200 kgTS/m^2·year were tested. However, the value used for STW design nowadays in Europe is 60 kgTS/m^2·year. Looking at our experience, in the Mediterranean region, the systems studied were characterised by sludge loading rate varying between 50-55 kgTS/m^2·y in Alpens and Sant Boi de Lluçanès and 125 kgTS/m^2·y in Seva (Table 4.2).

The range of loading applied in Catalonia (50-125 kgTS/m^2·y) shows the good performances of STW technology, which was capable of achieving efficiencies similar to those of conventional technologies both in term of sludge dewatering and organic matter stabilisation (Chapter 4). Even thought, PCA analysis highlights the better results obtained in Alpens and Sant Boi de Lluçanès, were the sludge loading rate was significantly lower.

Summarising, 50-60 kgTS/m^2·y seems to be an appropriate sludge loading rate in Mediterranean climate. However, climate conditions allows for a wide variability; in fact, depending on the sludge production and the system requirements, sludge loading rate might be increased up to 125 kgTS/m^2·y with a relatively content reduction of the treatment's efficiency according to the results showed in Chapter 4. Further studies might improve the knowledge on this important aspect of the treatment, enabling the determination of a more reliable sludge loading rate for these specific climate conditions.

Surface required

The range of application of STW reported in the state of art (Chapter 3) is widely variable, from 400 PE in Spain, up to 125,000 PE in Denmark. Indeed, literature suggests that the STW's capacity is not a limiting factor, wetlands might be used if sufficient land is available.

In this sense, it is important to consider the elevated surface required from such systems, which is one of the major disadvantages of this technology.

Actually, land availability could be a major concern in areas highly populated like the coastal regions of the Mediterranean. For this reason, in such circumstances, this technology can be considered appropriate for small (<2,000 PE) and remote communities.

The typical factor used for wastewater constructed wetlands is based on the population equivalent. On the contrary, the surface required for STW implementation (equation 10.2) is function of the sludge production, which is strictly dependent on the wastewater treatment used.

$$Surface = \frac{Sludge\_production\_(kgTS/y)}{Sludge\_loading\_rate\_(kgTS/m^2 \cdot y)} \quad \text{Eq. 10.2}$$

Thus, considering sludge loading rate of 60 $kgTS/m^2 \cdot year$, surface requirements are easily calculated from the sludge production. Table X.2 proposes a general sludge production from different wastewater treatments.

Considering that in most cases sludge production varies between 20 and 40 gTS/inhabitant·d, the STW surface requirement can vary approximately from 0.12 to 0.24 m^2 per inhabitant. However, due to the huge variability of sludge production, loading rate is conventionally used as a reliable parameter to characterise STW systems.

Table 10.2 Sludge production from different wastewater treatments (adapted from Von Sperling and Gonçalves, 2007).

Wastewater Treatment	Sludge production (gTS/inhabitant·d)	Total Solids (%)
Primary treatment (conventional)	35-45	2-6
Primary treatment (septic tanks)	20-30	3-6
Facultative pond	20-25	7-20
Conventional activated sludge		
Primary sludge	35-45	2-6
Secondary sludge	25-35	0.6-1
Mixed sludge	60-80	1-2
Activated sludge-extended aeration	40-45	0.8-1.2

Number and size of wetlands

The number of wetlands (beds or basins) constituting one system may vary depending on the facility size. Literature shows that also beds' dimension may be significantly variable (between 25 and 1000 m^2). According to the experiences acquired during the development of this thesis, and the literature review, the surface of each basin does not influence the systems performance. However, beds' size should be suitable to ease the uniform sludge distribution during feeding and the sludge withdrawn at the end of the treatment without using heavy machinery. There are no specific criteria for the shape of beds, although they are normally rectangular with a variable length-to-width ratio.

On the other hand, bed's number should be accurately evaluated in accordance to the operation of the system. In fact, the number of beds influences the feeding rotation among beds, and hence the resting periods and the consequent sludge dewatering performances.

As shown in the dewatering model performed in Chapter 6, the period elapsed between consecutive feedings is strictly correlated to the dewatering performances. Likewise, sludge dryness affects sludge layer thickness increasing rate and hence the frequency of beds' emptying procedure required.

Therefore, the appropriate number of beds allows for a right feeding/resting pattern and increases treatment performances. In small communities it is considered that at least four wetlands are necessary in order to allow the feeding rotation. In this way, when the sludge layer within the beds approaches the walls height; feeding should be intensified on two beds and then stopped. These beds, now filled, will be rested for some months and then withdrawn; while the other two beds will follow the feeding rotation. More beds will allow changes in feeding procedure and increase the resting periods between feedings, nevertheless construction costs will also raise.

Walls dimensioning

As mentioned in Chapter 3, systems can be excavated or built in concrete. In all cases, the covering with a waterproof membrane is suitable to seal the beds and prevent leaching.

Walls height determines the sludge thickness that can be accumulated within the wetlands. In the same way, bed's depth is related to the filter gravel elevation described later on. The most common depth available for sludge accumulation is around 1.5-1.6 m. Generally, the sludge layer increasing is around 10 cm/year, thus 1 m height for sludge storage would ensure sludge treatment during 8-10 years without require emptying operations.

According to our experience (Chapter 4), in Seva, where sludge thickness was around 1m, treatment efficacy was lower than in other systems. In this facility, the most evident consequence of the sludge layer height was the scarce dewatering degree of the bottom layer (35-70 cm depth) (Figure 4.4). It should be specified that in Seva sludge loading rate was significantly higher than in other systems and the feeding was manual, thus not standardised. Both specifics might be also responsible for the reduced performances of Seva's facility.

To sum up, considering 1 m as the maximum sludge thickness stored within the wetlands, and accounting for 0,50 m of filter medium and 0,20 m of safety margin, walls height should be designed around 1,60-1,70m.

Feeding and drainage/aeration system

Feeding pipe should be well designed in order to allow the excellent sludge distribution along all the bed. Figure 10.3 shows three different configurations of feeding pipe actually employed in different countries. In Denmark, the most used configuration for feeding systems consists of four (or more, depending on the bed's dimension) vertical pipe located in different zones of the bed (Figure 10.3a). On the other side, in Italy feeding from a singular pipe located in a corner of the bed is widely used (Figure 10.3b).

A different configuration is currently in use in Spain, where almost all the systems are fed along the shortest side. The pipe carrying the sludge is located in the middle of the wetland's side, where an horizontal opened pipe facilitates the sludge distribution along the bed' side (Figure 10.3c).

Considering advantage and inconvenience of each feeding system configuration, the vertical pipes seem to allow for a better sludge distribution all over the basin. In fact, in spite of the low water content of the influent sludge, the feeding on one corner or along one side, often do not provide a well sludge distribution.

Figure 10.3 Details of tree different feeding systems. Skovby (Denmark) on the top left, Oratoio (Italy) on the top right and Alpens (Spain) on the bottom.

Actually, a wet zone can be observed in Figure 10.3c in correspondence of the feeding pipe. Moreover, in many cases, plants growth is made difficult by the higher sludge loading in the feeding area. Sometimes, after heavy rains, horizontal distribution of sludge is even avoided by plants reclined on the opened pipe or on the sludge surface.

The perforated pipes located on the bottom of the filter are important for two reasons, in fact water percolated trough the filer is here collected and then returned to the headwork of the WWTP. Moreover, the perforation of the pipes allows a certain air exchange and thus bottom aeration.

Pipes should be located on the bottom of the filter along the entire basin in order to collect the major quantity of leachate; a slope of at least 1% is usually planned to ease water flow to the pipes outlet.

As for feeding system, different configuration of drainage pipes can be designed, in Figure 10.3a can be observed that in Danish systems a large number of pipes are designated to drainage and aeration (around 1 pipe each meter). On the other hand, in Spain normally these pipes are located every 2 or 3 meters depending on the bed width. Even though, as mentioned above, the results of nitrate concentration in the leachate (Chapter 4) and the greenhouse gasses measured (Chapter 8) suggest nitrification, thus the aerobic conditions of the Spanish systems.

The pipe systems suggested is constituted by three horizontal pipes for water percolation collection along the bottom of the wetlands, and four vertical pipes located in the middle of four zones of the bed in order to allow the homogeneous distribution of the influent sludge.

Filter medium

As for many other parameters, no standardisation is provided for the filter medium height or configuration. As mentioned in Chapter 3, filter heights found in literature are variable (between 30 and 60 cm) and do not seem to significantly affect the treatment efficiency.

The main function of the filter is to separate sludge solids from the liquid fraction. Thus, water percolates through the filter, which is normally constituted by several layers of granular media set in increasing size from the top to the bottom. Stones (diameter of around 5 cm) at the bottom protect draining pipes, gravel (diameter from 2 to 10 mm) is the main filter medium and sand (diameter from 0.5 to 1 mm) in the upper layer provides a primary physical filtration and rooting medium for plants.

The appropriate granulometry of the layers ensures the capillarity connection between the sludge and the filter, avoiding hydraulic failures that may cause insufficient percolation and consequently dewatering inefficiency. According to a Canadian guide about vertical flow constructed wetlands (Guide Marais Artificiels, 2010), the optimal hydraulic conductivity is assured by the following gravel granulomentry, which avoids particles migration between layer:

- D_{15} bigger gravel layer < D_{85} smaller gravel layer
- D_{50} bigger gravel layer < D_{50} smaller gravel layer
- D_{75} bigger gravel layer < D_{15} smaller gravel layer

Besides, the clogging of the filter is generally prevented by the sand layer, which offers the first separation between the sludge solids and the granular medium. The sand layer is also useful to protect the main filter layer during the emptying operation of the beds, in this way the gravel layer will not be replaced, reducing costs and materials requirements.

In fact, according to the economic study and the life cycle assessment (LCA) carried out in Chapter 9, the gravel medium is, with concrete and steel, one of the materials affecting significantly the investment costs and the environmental impact of STW. The replacement of the entire granular layer (around 260 m^2 per 2000 PE) would highly increase STW operation costs, which include the new raw material, the empting operation and the material disposal.

For a standard granular medium with a height of 30–60 cm, the most common layer heights are 15–20 cm for stones, 20–30 cm for gravel and 5–10 cm for sand. Results found in literature and from this study, suggest the independence of STW performance from layer heights. Thus, the lower depth of the granular layer results in lower costs, due to the lower material requirements and walls height. Taking into account the economic analysis (Chapter 9), we estimated that the increasing of the filter height from 30 to 60 cm results in a rising of investment cost of about 34%.

In Chapter 5, the effects of a different material as the main filter layer were investigated. In fact, in one bad constituting the pilot plant studied in this thesis, the gravel layer of about 30 cm was substituted by wood shavings. This allows for some waste reuse, reducing the use of new materials and, consequently, the STW environmental impact and investment cost of about 3.4 %.

Wood shaving were selected for this experiment since this is the organic material normally employed as support in composting treatment. Taking into account that, currently in Spain the normal practice is to compost biosolids from STW, using this material all the filter can be mixed by means a shovel, withdrawn, and directly transported to a composting plant.

The results discussed in Chapter 5 suggest the applicability of wood shaving as filter medium, in fact results obtained from this bed does not differ from the performances of the standard bed filled with gravel medium. Similarly, in recent studies, the effect of replacing the sand with a compost layer of 5–10 cm was evaluated in a pilot plant (Troesch et al. 2008a, 2008b). The results indicated that the vegetal compost layer was a better growing media for plants, but the lower filtration capacity of compost altered the filter function. Nevertheless, dewatering efficiencies were similar with both media.

However, in our experimentation, we detected a certain compaction of the wood medium, which was not related to a reduction of the water percolation. In fact, after more than 2 years of experiments, leachate volume collected by means drainage pipes was similar to the other beds of the pilot plants.

Concluding, according to the experience acquired from this experimentation, the following consideration should be taken onto account for the gravel medium design: 1) the filter layer can be constituted by granular medium or wood shaving, allowing for some cost reduction and new materials saving; 2) costs can be more significantly reduced by limiting the medium height to 30 cm; 3) it is important to ensure hydrological connection along the entire vertical profile, for this reason the granulometry of the medium has to increase progressively from the bottom to the top of the layer as specified above.

Plant species

As emerged from the dewatering and stabilization fundamentals discussed above, plants have a key role in STW treatment. Plants with high evapotranspiration rate, root development and oxygen transfer should be appropriate for STW. Moreover, plants should tolerate high variations in water availability, salinity and pH.

The most common plant species used for this purpose in Europe is *Phragmites australis*, even if, as mentioned in Chapter 3, other rooted wetland plant species such as *Typha* sp., *Cyperus papyrus* L. and *Echinochloa pyramidalis*, have been successfull-y used.

Taking into account that *Phragmites australis* is considered an invasive plant in the USA, we investigated the efficiency of *Typha* sp. for sludge treatment (Chapter 5) because of its elevated evapotranspiration rate and its availability in the Mediterranean region.

Concerning plants development, *Typha* sp. presented some growing problems, in particular during the first treatment phase. *Typha* sp. especially suffers the water lack in summer when, due to the high evapotranspiration rate, sludge dewatering was particularly fast. This fact results in a reduced plants density in beds planted with *Typha* sp. during all the experiment duration. Difficulties in *Typha* sp. adaptation to sludge systems was already presented in previous studies (Koottatep et al., 2005; Magri et al., 2010). However, a lower sludge loading rate during the first months of operation is always suitable, in order to facilitate plants growth.

Unexpectedly, the scarce plants density did not affect treatment efficiency. In fact, according to our experience developed at pilot scale, *Phragmites australis* and *Typha* sp. demonstrate comparable efficiencies in sludge dewatering and stabilisation. Indeed, after more than two years of experimentation, both plants show good treatment performances and no significant differences were detected between plant species (Chapter 5).

Additional considerations

With regard to the economics end environmental aspects, the study presented in Chapter 9 highlights the reduced costs and the low environmental impact of STW with respect to conventional sludge management currently diffused in small communities. However, some additional considerations should be taken into account in order to further reduce costs and impacts of the treatment.

It is evident that the correct dimensioning of the system will enhance treatment performances. In this way, the elevated volume reduction results in longer cycles between emptying operations, which account for around two-thirds of the STW operation costs. For this reason, the appropriate considerations about loading rates, surface requirements, number and depth of beds are important in order to reduce significantly the maintenance cost. Moreover, as extensively discussed on Chapter 7, the improvement of sludge quality in term of stabilisation will allow the biosolids application to agricultural fields without require post treatment. This aspect will also significantly reduce STW costs, as calculated in Chapter 9 (Figure 9.2).

However, the economic study (Table 9.5) puts forward the significance of the investment costs with respect to the operation requirements. As mentioned above, investment costs can be reduced limiting the employment of raw materials, thus concrete for walls and gravel for the filter. Besides, the reduction of raw materials use will limit the environmental impact of STW. In fact, LCA results (Figure 9.3) indicate the relevant impact of raw material mostly in the eutrophication and global worming categories.

Operational criteria

After beds' dimensioning and construction, it is important to define the appropriate operational criteria in order to enhance treatment performances. Operational criteria, at the moment not standardised, consist basically into the establishment of feeding/resting timing and the final resting period needed to complete the treatment.

As mentioned in Chapter 6, the operational criteria can determine biosolids quality, sludge layer increasing rate within the beds and, consequently, the number of the emptying procedures needed. Besides, emptying procedure and biosolids transport affect significantly the treatment cost, being the most expensive STW operation (Chapter 9).

For this reason, in this thesis efforts have been done in order to determine feeding/resting patterns by means a model able to predict sludge dewatering (Chapter 6). Moreover, the final resting period required in order to improve biosolids properties has been investigated in full-scale systems (Chapter 7) and in the pilot plant (Chapter 5).

Feeding/resting patterns

The model performed indicates the days of resting needed in order to achieve a certain sludge water loss between feedings. The appropriate resting timing between loadings ensures the dewatering of the upper sludge layer before the following application of fresh sludge. If adequately dewatered, sludge layer will not be excessively affected by the new sludge application. This results in a suitable dewatering degree along all the vertical profile of the sludge layer.

Contrarily, if feeding events are not adequately paused, dewatering processes do not take place correctly and wet conditions persist in the sludge layer. This behaviour was detected in Seva's facility, were the water content was higher in the bottom layer (Figure 4.4), probably due to the high sludge loading rate, but also to the wrong feeding patter applied. In fact, in this system, feeding was manual and not standardised. Here, the subsequent feeding events do not allow the sufficient water loss, resulting in scarce dewatering performances.

Even if not considered within our model, the stabilisation degree of the organic matter was also affected by the Seva's feeding (Figure 4.5 and 4.6). Bearing in mind that such conditions of the sludge layer are not reversible, it becomes really important to start feeding with the correct pattern in order to avoid system's failures.

According to the model developed, resting time between feedings is a function of the dewatering degree desired; which strictly depends on the evapotranspiration rate and the sludge layer height. Thus, the model is able to predict sludge dewatering in a time lapse by means the introduction of the climate conditions which influence the evapotranspiration rate and the sludge layer height. The last parameter can be measured or, at the beginning of the process, can be calculated considering the mean sludge increasing rate around 10 cm/year.

The case study performed in Chapter 6 suggest that the optimum resting periods oscillates between 3.5 days for layer height around 20 cm (three first years of operation) and 40 days for layers around 80 cm height (corresponding to the last years of operation before emptying). Of course these values are also influenced by the evapotranspiration rate, which improve the dewatering degree but, according to our model, do not influence significantly the resting time needed to achieve good results.

Model results are in a certain way confirmed by the full-scale systems studied in Chapter 4. In Alpens and Sant Boi de Lluçanès the sludge layers were both around 20 cm height, feeding was stopped during 4 days in the first system and during 10 days in the second. Model outputs in Figure 9.4a would correspond to the facilities studied, suggesting around 4 days to achieve the maximum sludge dewatering. Results from the PCA exposed above confirm a satisfactory dewatering degree in both facilities. Even if in Sant Boi de Lluçanès total solids concentration was slightly higher, Alpens values were elevated, confirming that 4 resting days were sufficient to achieve sludge dewatering.

According to the model the sludge loading rate can be determined as a function of evapotranspiration, time between feedings and sludge height. Besides, it is suitable to feed in one time or, at least, during one day all the sludge corresponding to the loading rate. In this way sludge fed will exercise higher pressure, which aids water percolation.

Moreover, the study conducted on the greenhouse gasses emissions (Chapter 8), indicates the elevated change in bed's conditions caused by sludge feeding events. In fact, the peaks in methane emissions detected immediately after sludge loading, suggest the enhancement of anaerobic conditions. Subsequently, the return to aerobic conditions is facilitated by sludge moisture reduction. It is important to remark that such peaks of methane emissions does not affect significantly the environmental impact cause by STW. Even though, the punctual feeding event, will limit anaerobic condition within the wetlands and will facilitate the return to aerobic conditions. In this way, aerobic degradation of the organic matter will be enhanced and methane and odor emissions limited.

Concerning the entire systems (with all the beds), feeding patters have been briefly described above. All the beds available are fed during some years; afterwards, when the sludge layer within the beds approaches the walls height, feeding is intensified in two beds. When these beds are filled, feeding is stopped here during the final resting period and follows the rotation to the next beds. Subsequently, after the resting period, beds are emptied and then fed again, entering in the normal loading rotation.

Final resting period

After the last feeding event, a final resting periods, longer than the others, is suitable. This period is needed in order to allow the dewatering of the last sludge fed, and to improve sludge dewatering and stabilisation of the entire sludge layer. During the feeding period, biosolids are continuously load with fresh sludge, which add water and organic matter to the entire sludge layer. Thus, an additional resting period is required after the last loading.

This period is highly variable, according to the biosolids characteristics, the climate conditions and the operational requirements, among others. It is important to consider that treatment surface is reduced during resting time, thus this period should be brief, but sufficient to obtain a dry and stabilised product suitable for its reuse.

Biosolids properties required are different, depending on the reuse selected. Generally, the most suitable solution for biosolids is agricultural reuse, due to their organic matter and nutrients concentrations. Of course, contaminants concentration of the final product should be monitored and analysed before sludge spreading to agricultural fields.

As widely discussed in this thesis (Chapters 3, 4, 5 and 7), heavy metals are the only parameter currently regulated by the legislation in force for sludge land application. Even thought, the determination of pathogens concentration is also recommended.

In this thesis, the parameters mentioned above were analysed both in sludge during the treatment (between feedings) and in biosolids after resting periods ranging between 4 and 18 months. In the cases considered here, heavy metals concentration was always below the law threshold, even in the sludge during the treatment. This could be due to the fact that all the STW studied only treat municipal wastewater, thus low in heavy metals. On the other hand, pathogens (*E.coli*) in sludge during the treatment were often above the limits proposed by a law draft (currently not in force). Even if pathogens were present in lower concentrations in biosolids, this is one of the critical points of this treatment. In fact within wetlands the high temperatures needed for hygienisation are not reached, for this reason in many countries, like Italy or Spain, biosolids from STW are post treated in composting plants.

In our opinion, further investigations are required in order to determine the optimal resting period corresponding to different climate conditions. However, the following consideration should be taken into account before proceeding to the emptying operations:

1) to analyse heavy metals concentration, pathogens and the parameters eventually suggested by the legislation;
2) to assess the mineralisation degree by means the quantification of the biodegradable matter present in the biosolids;
3) to stop feeding during the dry season, in order to enhance sludge dewatering.

Sludge withdrawn

Once determined the resting period required and analysed the biosolids, final product is withdrawn together with the plants, which could be previously harvested (Figure 7.1). Attention must be paid in order to withdraw only the sludge layer, in this way plant roots remaining within the granular medium will regenerate the vegetation without requiring replanting.

After withdrawn, which should be done in the dry season, the emptied bed is levelled and feeding cycle will restart with lower loading rates to allow plants regrowth. Sometimes replanting is done in order to enhance plants regrown after emptying; while the granular medium is usually not replaced.

If possible, wetland emptying should be carried out in correspondence with the sowing time, in order to allow the direct biosolids spreading to the nearest agricultural fields. Of course, agricultural uses are justified when the fields are located near the facility, also for this reason we considered that the application of this treatment is more reasonable in small rural community that in large cities.

Design and operational problems

The most evident problem that can occur in STW is the scarce vegetation density (Figure 10.4). Usually this results from the high sludge loading rate during the first period of feeding. In fact, all plants species require a period of adaptation to the bed's conditions, wastewater can be used to this purpose during the firsts loadings. As a consequence of the poor vegetation development, and thus elevated sludge loading rates, treatment efficiency is sensibly reduced. In this case, the dewatering degree achieved is scarce, due to the reduced water percolation and the absence of plants evapotranspiration.

In case of overloading and lack of vegetation, oxygen transfer through the sludge layer is prevented. Thus, the development of bubbling (Figure 10.6) indicates the enhancement of anaerobic conditions and the consequent production of odours and methane.

Sludge overloading is thus one of the main cause of STW problems. It can be due to different causes. The insufficient system dimensioning can be an important factor in this sense. In fact it depends on the sludge loading rate established, which is strictly dependant on the sludge production. It is also important to take into account that the loading during the first period need to be reduced in order to help vegetation growth. Thus, the correct consideration may be done during STW dimensioning in order to avoid sludge overloading.

The scarce number of beds, and the consequent variation in the feeding/resting period can also cause sludge overloading. In fact, if the number of beds is insufficient, it will not be possible to apply the right resting period needed in order to enhance sludge treatment between feedings. Moreover, according to our model, it is important to take into account that the resting period required between feedings vary according to the sludge layer height.

Figure 10.4 Vegetation growth problems in Alpens (on the top) and in Santa Eulalia (on the bottom), Spain.

11. Conclusions

In this PhD Thesis different aspects of sludge treatment wetlands (STW) were investigated in order to gain knowledge on this novel technology with potential to be implemented in the Mediterranean Region. The following conclusions can be drawn from this work:

- The comparison of sludge treatment wetlands with conventional technologies like centrifugation and composting suggests a high efficiency of the system in terms of sludge dewatering (around 30% TS) and stabilisation (40-50% VS), which leads to a final product that may be suitable for agricultural crop fields and land reclamation, even without further composting of the treated sludge. This provides an opportunity for on-site sludge treatment, especially in WWTPs of small communities.
- Sludge dewatering and mineralisation in the three full-scale STW evaluated in Catalonia (Spain) compare well with conventional sludge treatment technologies like centrifugation and composting, suggesting the potential applicability of STW in small communities of the Mediterranean Region. Indeed, sludge dryness was increased from 1-3% to 20-30 %TS; and sludge mineralisation was shown by the decrease in VS from 52-67 to 30-49 %VS/TS.
- Process performance was similar in pilot STW planted with P. australis and Typha sp., regardless of the filter material (gravel or wood shavings). In all cases the sludge volume was reduced by 80% and the TS concentration increased from around 2% to 16-24%, in accordance with the results from the studied full-scale STW. The VS content (50%VS/TS) observed after a period of two months without feeding suggests that a longer resting period is needed to enhance the biosolids mineralisation.

- The biosolids obtained from the pilot plant and the full-scale systems located in Catalonia and Denmark were characterised by: 16-26 %TS, 42-54 %VS/TS, DRI between 0.5-1.4 mgO2/gTS·h, germination index between 201-203 %, faecal bacteria indicators and heavy metals complying with current regulations. According to this, STW biosolids are a partly stabilised product, with nearly the stabilisation degree of compost, which do not cause phytotoxicity; indicating their suitability for agricultural reuse. However, attention should be paid to the concentration of pathogens.
- A dewatering model was developed and calibrated with moisture data from pilot STW. According to the validation test, the model implemented is able to estimate water loss (%) with time, thus moisture reduction within the sludge layer. Case studies have shown that it is possible to determine the most appropriate feeding frequency as a function of the sludge height within the STW; and the sludge loading rate as a function of the sludge height, feeding frequency and evapotranspiration. On the whole, the model implemented is a useful tool to establish standardised criteria for STW operation.
- The static chamber method was successfully adapted to the determination of gas emissions from STW. In spite of the spatial and temporal variation in the CH_4 and N_2O emissions in STW, there was no specific difference in the emissions along the wetland resulting from homogeneous sludge distribution and biodegradation in STW. Aerobic conditions before feeding, characterised by low methane emissions (10 $mgCH_4/m^2 \cdot d$) and high nitrous oxide emissions (950 $mgN_2O/m^2 \cdot d$), were strongly changed by fresh sludge feeding which enhances CH_4 emissions (5,400 $mgCH_4/m^2 \cdot d$) and decreases N_2O emissions (20 $mgN_2O/m^2 \cdot d$). The Global warming potential of CH_4 and N_2O emissions from STW correspond 17 $kgCO_2eq/PE \cdot y$ which is from 2 to 9 times lower than that of sludge centrifugation (36 $kgCO_2eq/PE \cdot y$) and transport (162 $kgCO_2eq/PE \cdot y$).
- According to the economic and environmental assessment, STW with direct land application is the most cost-effective scenario, which is also characterised by the lowest environmental impact. If compost post-treatment is required, the costs increase only slightly but environmental impacts increase significantly. Centrifugation costs are higher than STW for systems up to 1,800 PE, but become similar for 2,000 PE systems. However, environmental impacts are higher in all categories compared to STW with direct land application. Sludge transport to external treatment is always the most expensive and environmentally unfriendly scenario.

12. References

Adani, F., Lozzi, P., Genevini, P.L. (2000). Determination of biological stability by oxy-gen uptake on municipal solid waste and derived products. Compost Sci.Util. 9, 163-178.Allen, R.G., Pereira, L.S., Raes, D., Smith, M. (1998). Crop evapotranspiration: Guideline for computing crop water requirements. FAO Irrigation and Drainage Paper 56. FAO. Rome, Italy, 1998.

Ahmad, S., Li, C., Dai, G., Zhan, M., Wang, J., Pan, S., Cao, C. (2009) Greenhouse gas emission from direct seeding paddy field under different rice tillage systems in central China. Soil and Tillage Research 106(1), 54-61.

Alm, J., Saarnio, S., Nykänen, H., Silvola, J., Martikainen, P. (1999) Winter CO2, CH4 and N2O fluxes on some natural and drained boreal peatlands. Biogeochemistry 44, 163-186.

Andreoli, C.V, Pegorini, E.S., Fernandes, F., Santos H.F., (2007). Land application of sewage sludge. In: Sludge treatment and disposal. Cleverson, Von Sperling & Fernandes Eds. IWA Publishing, London, UK, 2007.

APHA-AWWA-WPCF (2001). Standard methods for the examination of water and wastewater (20th ed.), American Public Health Association, Washington DC.

ASCE-EWRI (2005). The ASCE Standardized Reference Evapotranspiration Equation. Technical Committee Report of the Environmental and Water Resource Institute of the American Society of Civil Engineers from the Task Committee on Standardization of Reference Evapotranspiration.

Baird, K.J., Maddock, T. (2005). Simulating riparian evapotranspiration: a new methodology and application for groundwater models. Joutnal of Hidrology 312 (1-4), 176-190.

Barrena, R., D'Imporzano, G., Ponsá, S., Gea, T., Artola, A., Vázquez, F., Sánchez, A., Adani, F. (2009a). Journal of Hazardous Materials 162 (2-3), pp 1065-1072.

Barrena, R., Casals, E., Colón, J., Font, X., Sánchez, A., Puntes, V. (2009b). Evaluation of the ecotoxicity of the model nanoparticles. Chemosphere 75, 850-857.Bastviken, S.K., Eriksson, P.G., Premrov, A., Tonderski, K. (2005) Potencial denitrifi-cation in wetland sediments with different plant species detritus. Ecological En-gineering 25(2), 183-190.

Begg J.S., Lavigne R.L., Veneman P.L.M. (2001). Reed beds: constructed wetlands for municipal wastewater treatment plant sludge dewatering. Water Science & Technology 44 (11-12), 393-398.

Bernal M.P., Alburquerque J.A., Moral R. (2008). Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresource Technology, 100 (22), 5444-5453.

Bertrán, E., Sort, X., Soliva, M., Trillas, I. (2004). Composting winery waste: sludges and grape stalks. Bioresource Technology 95, 203-208.

Bianchi, V., Peruzzi, E., Masciandaro, G., Ceccanti, B., Ravelo, S., Iannelli, R. (2010). Efficiency assessment of a reed bed pilot plant (Phramites australis) for sludge stabilisation in Tuscany (Italy). Ecological Engineering, doi:10.1016/j.ecoleng.2010.05.008

Bolzonella, D., Pavan, P., Battistoni, P., Cecchi, F. (2005). Mesophilic anaerobic digestion of waste activated sludge: influence of the solid retention time in the wastewater treatment process. Process Biochemistry 40, 1453-1460.

Borin, M., Milani, M., Salvato, M., Toscano, A. (2010). Evaluation of Phragmites australis (Cav.) Trin. evapotranspiration in Northern and Southern Italy. Ecological Engineering, doi:10.1016/j.ecoleng.2010.05.003.

Brix, H., Schierup, H.-H. & Arias, C.A. (2007). Twenty years experience with constructed wetland systems in Denmark – what did we learn? Water Science and Technology 56, 63-68

Brodrick, S.J., Cullen, P., Maher, W. (1998) Denitrification in a natural wetland receiv-ing secondary treated effluent. Water Research 22(4), 431-439.

Burgoon, P.S., Kirkbride, K.F., Henderson, M. and Landon, E. (1997). Reed beds for biosolids drying in the arid Northwestern United States. Water Science and Technology 35 (5), 287-292.

Caselles-Osorio, A., Puigagut, J., Segú, E., Vaello, N., Granés, F., García, D. and García, J. (2007). Solids accumulation in five full-scale subsurface flow constructed wetlands. Water Research 41 (6), 1388-1398.

Cambell, M.W. (2000). Sludge management-Future issues and trends. Water Science and Technology 41 (8), 1-8.

Catalan Water Agency (2008). La gestió de fangs de depuració d'aigües residuals municipals a Catalunya. Evolució històrica i estratègia de futur (nou programa de fangs). (In Catalan) In: III Jornades Tècniques de Gestió de Sistemes de Sanejament d'Aigües Residuals, Barcelona, Spain. (2007) URL:http://www.mediambient.gencat.net/aca/ca//agencia/agenda/jornades_tecniques003/ponencies/inici.jsp

Catalan Water Agency (2010). Programa d'actuació per a la gestió dels fangs residuals generats en els porcesos de depuració d'aigües residuals urbanes de Catalunya. (In Catalan). URL: https://aca-web.gencat.cat/aca/documents/ca/home/Programa_de_fangs%20101004.pdf

Catalan Water Agency (2011a). Diagrama I flux de la gestió de fangs. (In Catalan). URL: https://aca-web.gencat.cat/aca/documents/ca/tractament_fangs/diagrama_flux.pdf

Catalan Water Agency (2011b). Sistemas de saneamiento. (In Spanish) URL: https://aca-web.gencat.cat/aca/appmanager/aca/aca?_nfpb=true&_pageLabel=P18600879181247495336951

Chazarenc, F., Merlin G., Gonthier Y. (2003). Hydrodynamics of horizontal subsurface flow constructed wetlands wetlands. Ecological Engineering 21, 165-173.

Chen, G., Yue, P-L., Mujumdar, A.S. (2002). Sludge dewatering and drying. Drying technology 20 (4), 883-916.

Chu, C.P., Lee, D.J. (1999). Three stage of consolidation dewatering of sludges. Journal of Environmental Engineering 125 (10), 959-965.

Cole, S. (1998). The Emergence of Treatment Wetlands. Environmental Science and Technology A-Pages 32 (9), 218A-223A.

Committee for European Normalization (2003). EN 13725:2003. Air Quality Determination of odor concentration by dynamic olfactometry. Brussels, Belgium.

Consejo de Ministros. Resolución de 14 de junio de 2001, de la Secretaría General de Medio Ambiente, por la que se Dispone la Publicación del Acuerdo de Consejo de Ministros, de 1 de junio de 2001, por el que se Aprueba el Plan Nacional de Lodos de Depuradoras de Aguas Residuales 2001-2006. Boletín Oficial del Estado 166, de 12 de julio de 2001.

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.

Council of the European Union (1991). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 Establishing a Framework for the Community action in the Field of Water Policy. Official Journal of the European Union L 327, 30/05/1991, 1-73.

Council of the European Union (2000). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 Establishing a Framework for the Community action in the Field of Water Policy. Official Journal of the European Union L 327, 30/05/1991, 1-73.

Crill, P.M., Harriss, R.C., Bartlett, K.B. (1991) Methane fluxes from terrestrial wetland environment. In: Rogers, J.E., Whitman,W.B. (Eds.), Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides and Halome-thanes. American Society for Microbiology,Washington, DC.

Crites, R.W., Tchobanoglous, G. (1998). Small and Decentralized Wastewater Management Systems. McGraw-Hill, New York.

Cui, Y., Sun, T., Zhao, L., Jiang, T., Zhang, L. (2008). Performance of wastewater sludge ecological stabilization. Journal of Environmental Sciences 20, 385-389.

De Maeseneer, J.L. (1997). Constructed wetland for sludge dewatering. Water Science and Technology 35 (5), 279-285.

Draeger, K., Pundsack, J., Jorgenson, M., Mulloy, W.E. (1999). Watershed Effects and Biosolids Land Application: Literature Review. Water Environmental Research Foundation. Project 96-REM-2.

Duchemin, E., Lucotte, M., Canuel., R. (1999) Comparison of static chamber and thin boundary layer equation method for measuring greenhouse gas emissions from large water bodies. Environmental Science Technology 33, 350-375.

Edwards, J.K., Gray, K.R., Cooper, D.J., Biddlestone, A.J. and Willoughby, N. (2001). Reed bed dewatering of agricultural sludges and slurries. Water Science and Technology 44 (11-12), 551-558.

Elvira, C., Goicoechea, M., Sampedro, L, Mato S., Nogales, R. (1996). Bioconversion of solid paperpulp mill sludge by earthworms. Bioresource Technology,57, 173–177.

Environment DG, EU. (2000). Working Document on Sludge 3rd Draft. URL: http://ec.europa.eu/environment/waste/sludge/pdf/sludge_en.pdf (July 2003)

Environment DG, EU. (2001). Working Document on Biological Treatment of Biowaste 2nd Draft. URL: http://ec.europa.eu/environment/waste/sludge/pdf/sludge_en.pdf.

Fang, M., Wong, J. W. C. (1999). Effects of lime amendment on availability of heavy metals and maturation in sewage sludge composting. Environmental Pollution 106, 83-89.

Ferrer I., Ponsá S., Vázquez F., Font, X. (2008). Increasing biogas production by thermal (70 ºC) sludge pre-treatment prior to thermophilic anaerobic digestion. Biochemical Engineering Journal 42(2), 186-192.

Ferrer, I., Vázquez, F., Font, X. (2010). Long term operation of a thermophilic anaerobic reactor: process stability and efficiency at decreasing sludge retention time. Bioresource Technology, 101(9), 2972-2980.

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.

García, J., Rousseau, D.P.L, Morató, J., Lesage, E., Matamoros,V. and Bayona, J.M. Contaminant removal processes in subsurface-flow constructed wetlands: a review. Critical Reviews in Environmental Science and Technology 2010, 40, 561-661

García, J., Aguirre, P., Barragán, J., Mujeriego, R., Matamoros, V., Bayona, J.M. (2005) Effect of key design parameters on the efficiency of horizontal subsurface flow constructed wetlands: long-term performance pilot study. Ecol. Eng. 25, 405–418.

Giraldi, D., Masciandaro, G., Peruzzi, E., Bianchi, V., Peruzzi, P., Ceccanti, B., Iannelli, R. (2008). Hydraulic and biochemical analysis on real scale sludge consolidation reed beds in Tuscany (Italy). In: Proceedings of the 11th International Conference on Wetland Systems for Water Pollution Control, Indore, India. Billore S., Dass P., Vymazal J. (Eds).

Giraldi, D., Masciandaro, G., Peruzzi, E., Bianchi, V., Peruzzi, P., Ceccanti. B., Iannelli, R. (2009a). Hydraulic and biochemical analysis on full-scale sludge consolidation reed beds in Tuscany (Italy). Water science and technology 60 (5), 1209-1216.

Giraldi, D., Iannelli, R. (2009b). Short-term water content analysis for the optimization of sludge dewatering in dedicated constructed wetlands (reed bed systems). Desalination 246, 92-99.

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

Gonzéles, M. (2011). The soil (In Spainsh). Edicions UPC, Barcelona, Span.

Gregorich, E. G., Rochette, P., VandenBygaart, A. J., Angwers, D. A. (2005) Green-house gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil and Tillage Research 83(1), 53-72.

Groffman, P.M., Gold, A., Addy, K. (2000) Nitrous oxide production in riparian zones and its importance to national emission inventories. Chemosphere Global Change Sci. 2(3–4), 291– 299.

Grünfeld, S., Brix, H. (1999) Methanogenesis and methane emissions: effects of water table, substrate type and presence of Phragmites australis. Aquat. Bot. 64, 63–75.

Guide Marais Artificiels (2010). Guide pour l'étude des technologies conventionnelles du traitement des eaux usées d'origine domestique (In French). URL: http://www.mddep.gouv.qc.ca/eau/eaux-usees/domestique/index.htm

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

Hallberg, K.B., Johnson, D.B. (2005). Microbiology of a wetland ecosystem constructed to remediate mine drainage from a heavy metal mine, Science of the Total Environment 338 (1–2), pp. 53–66.

Hanson, R.S., Hanson, T.E. (2006) Methanotrophic bacteria. Microbiol. Rev. 60(2), 439–444.

Hardej, M. and Ozimek, T. (2002). The effect of sewage sludge flooding on growth and morphometric parameters of Phragmites australis (cav.) trin.ex steudel. Ecological Engineering 18, 343-350.

Hernández, T., Moreno, J.I., Costa, F. (1999). Influence of sewage sludge application on crop yields and heavy metals availability. Soil Science & Plant Nutrition 37 (2), 201-210.

Hedges, P.D., Fermor, P.H., Dušek J. (2008). The hydrological sustainability of constructed wetlands for wastewater treatment. In: Wastewater Treatment Plant Dynamics and Managementin Constructed and natural Wetlands. Vymazal Editor. Springer Publishing, Třeboň, Czech Republic, 2008.

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.

Hospido, A., Moreira, M. T., Martín, M., Rigola, M, Feijo, G. (2005). Environmental Evaluation of Different Treatment Processes for Sludge form Urban Wastewater Treatments: Aerobic Digestion versus Thermal Processes. International Journal of LCA 10 (5), 336-345.

Houillon, G., Jolliet, O. (2005). Life Cycle Assessemnt of Processes for the treatment of Wastewatre Urban Sludge: Energy and Global Warming Analysis. Journal of Cleaner Production 13, 287-299.

Huertas, E., Güell, D., Folch, M., Salgot, M., Torrens, A. (2004) Reed bed for sludge dewatering in Catalonia (Spain). In: Proceedings of the 9th International Conference on Wetland Systems, Avignon, France. Liénard A. (ed).

IPCC (Intergovenmental panel on climate change) 2006 Guidelines for national green-house gas inventories. Volume 5. Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan.

IPCC (Intergovenmental panel on climate change) 2007 Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assess-ment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

ISO (International Organization for Standardization) (2006a). Environmental Management - Life Cycle Assessment- General Principles and Framework. ISO 14040. Geneva, Switzerland, 2006.

ISO (International Organization for Standardization) (2006b). Environmental Management - Life Cycle Assessment – Requirement and guidelines. ISO 14044. Geneva, Switzerland, 2006.

Kadlec, R.H., Knight, R.L. (1996). Treatment Wetlands. CRC Press, Boca Ratón, FL.

Kadlec, R.H., Wallace, S.D. (2009). Treatment Wetlands, second ed. CRC Press, Boca Raton, FL.

Kampschreur, M. J., Tammink, H., Kleerebezem, R., Jetten, S. M., Loosdrecht, M. (2009) Nitrous oxide emission during wastewater treatment. Water research (43), 4093-4103.

Kengne Noumsi, I. M., Amougou, A., Bemmo, N., Strauss, M., Troesch, S., Ntep, F., Tsama Njitat, V., Ngoutane Pare, M., Koné, D. (2206). Potentials of sludge drying beds vegetated with Cyperus papyrus L. and Echinachloa pyramidakis (LAM.) Hitchc. & chase for faecal slduge treatment in tropical regions. In: Proceedings of the Conference on The Use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands. Lisbon, Portugal. Dias V. and Vymazal J. (ed).

Kim, B.J. and Smith, E.D. (1997) Evaluation of sludge dewatering reed beds: a niche for small systems. Water Science and Technology 35 (6), 21-28.

Komilis, D. P., Tziouvaras I. S. (2209). A statistical analysis to assess the madurity and stability of six composts. Waste Management 29, 1504-1513.

Koottatep, T. Surinkul, N., Polprasert, C., Kamal, A.S.M., Kone, D., Montangero, A., Heinss, U., Strauss, M. (2005). Treatment of septage in constructed wetlands in tropical climate: lessons learnt from seven years of operation. Water Science and Technology 51 (9), 119-126.

Knowles, R. (1982) Denitrification. Microbiological Reviews, 46(1), 43–70.

Krugel S., Nemeth L., Peddie C. (1998). Extending thermophilic anaerobic digestion for producing Class A biosolids at the Greater Vancouver Regional District Annacis Island wastewater treatment plant. Water Science & Technology 38, 409-416.

Lasaridi, K.E., Stentiford, E. I. (1998). A simple respirometric technique for assessing compost stability. Water Research 32, 3717-3723.

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.

Lambe, W., Withman, R. V. (1979). Soil Mechanincs. SI version. John Wiley & Sons, New York, NY, 553p.

Liénard, A., Duchène Ph., Gorini, D. (1995). A study of activated sludge dewatering in experimental reed-planted or unlpanted sludge drying beds. Water Science and Technology 32 (3), 251-261.

Liénard, A, Troesch, S., Molle, P., Thirion, F., Héritier, P., Baudez, J-C., Esser, D. (2008). Valorisation of sludge treated in drying reed beds: first empting and spreading experiences in France. (In French). Ingénieries nº special 2008, 51-64.

Liikanen, A., Huttunen, J.T., Karjalainen, S. M., Heikkinen, K., Väisänen, T. S., Nykänen H, Martikainen, P.J. (2006) Temporal and seasonal changes in green-house gas emissions from a constructed wetland purifying peat mining runoff waters. Ecological Engineering 26(3), 241–51.

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.

Lundin, M., Olofsson, M., Pettersson, G. J., Zetterlund, H. (2004). Environmental and Economic Assessment of Sewage Sludge Handling Options. Resources, Conservation and Recycling 41, 255-278.

Luduvice, M. (22072007). Sludge stabilization. In: Sludge treatment and disposal. Cleverson, Von Sperling & Fernandes Eds. IWA Publishing, London, UK, 2007.

Machefert, S.E., Dise N.B., Goulding K.W.T., Whitehead, P.G. (2002) Nitrous oxide emission from a range of land uses across Europe. Hydrology and Earth System Sciences 6, 325–337.

Magri, M. E., Suntti, C., Voltolini, C. A., Philippi, L.S (2010). Performance of different macrophytes species in constructed wetlands systems for anaerobic sludge dewatering, experience from Southern Brazil. Proceeding of Conference: 12th International Conference on Wetland Systems for Water Pollution Control. 4-8 October. Venice (Italy).

Maljanen, M., Komulainen, V. M., Hytönen, J., Mertikainen, P.J., Laine, J. (2004) Car-bon dioxide, nitrous oxide and methane dynamics in boreal organic agricultural soils with different soil characteristics. Soil Biology and Biogeochemistry 36(11), 1801-1808.

Maljanen, M., Kohonen, A.R., Virkajärvi, P., Martikainen, P. (2007) Fluxes and pro-duction of N2O, CO2 and CH4 in boreal agricultural soil during winter as af-fected by snow cover. Tellus, 59B, 853–859.

MARM (2010). Handbook for wastewater treatment systems establishment in small communities (In Spainsh). Ministerio Medio Ambiente y medio Rural y Marino (ed). Madrid (Spain).

Martikainen, P.J., Nykänen, H., Crill, P., Silvola, J. (1993) Effect of lowered water table on nitrous oxide fluxes from northern peatlands. Nature 366, 51–53.

Melidis, P., Gikas, G. D., Akratos, C.S., Tsihrintzis, V. A. (2010). Dewatering of primary settled urban sludge in a vertical flow wetland. Desalination 250, 395-398.

Metcalf and Eddy (2003). Wastewater Engineering: Treatment, Disposal and Reuse. McGraw-Hill. New York.

Mininni, G., Bolzonella, D., Braguglia, C.M., Cecchi, F. (2006). New perspectives of combined anaerobic digestion of secondary sludge together with organic solid waste. In:

Proceedings of IWA Specialized Conference: Sustainable sludge management: state of the art, challenges and perspectives, Moscow, Russia, 303-309.

Moore, T.R., Dalva, M. (1993) The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soils. J. Soil Sci. 44, 651–604.

Moss, L.H., Epstein, E., Logan, T. (2002). Evaluating Risks and Benefits of Soil Amendments Used in Agriculture. International Water Association and Water Environmental Research Foundation (co-ed.). Alexandria VA.

Mujeriego, R. and Carbó, M. (1994). Sludge reuse in agriculture (In Catalan). Technical Report. Consorci de la Costa Brava. Girona (Spain).

Muller, W., Fricke, K., Vogtmann, H. (1998). Biodegradation of organic matter during mechanical biological treatment of MSW. Compost Science and Utilization 6, 42-52.

Murray, A:, Horvath, A., Nelson, K. L. (2008). Hybrid life-Cycle Environmental and Cost Inventory of Sewage Sludge Treatment and End-Use Scenarios: A Case Study form China. Environmental Science & Technology 42, 3163-3169.

Nassar, A.M., Smith, M., Afiti, S. (2006). Sludge dewatering using the reed bed systems in the Gaza Strip, Palestine. Water and Environmental Journal 20, 27-34.Nielsen, S. (2003a). Sludge drying reed beds. Water Science and Technology 48 (5), 101-109.

Neue, H. U., Gaunt, J. L., Wang, Z. P., Becker-Heidmann, P., Quijano, C. (1997) Car-bon in tropical wetlands. Geoderma 79(1-4), 163-185.

Nielsen, S. (1990). Sludge dewatering and mineralization in reed beds systems. In: Constructed Wetlands in Water Pollution Control (Adv. Wat. Pollut. Control 11). Cooper, P.F. and Findlater B.C. (eds), 245-255, Pergamon Press.

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. (2004). Lits de séchage de boues plantés de roseaux: foncionnement at problèmes. Ingéniers, numero spécial, 67-75.

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

Nielsen, S. (2005). Mineralization of hazardous organic compounds in a sludge reed bed and sludge storage. Water Science & Technology 51(9), 109-117.

Nielsen, S. (2007). Helsinge sludge reed beds systems: reduction of pathogenic microorganisms. Water Science & Technology 56(3), 175-182.

Nielsen, S. (2008). Sludge treatment and drying reed bed systems 20 years of experience. Processing of Conference: European Conference on Sludge Management. 1-2 September, Liège (Belgium).Obarska-Pempkowiak, H., Tuszynska, A. and Sobocinski, Z. (2003). Polish experience with sewage sludge dewatering in reed systems. Water Science and Technology 48 (5), 111-117.

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

Nykänen, H., Alm, J., Silvola, J., Tolonen, K., Martikainen, P.J. (1998) Methane fluxes on boreal peatlands of different fertility and the effect of long-term experimental lowering of the water table on flux rates. Glob. Biogeochemical Cycles 12, 53–69.

Obarska-Pempkowiak, H., Tuszynska, A., Sobocinski, Z. (2003). Polish experience with sewage sludge dewatering in reed systems. Water Science & Technology 48(5), 111-117.

Oleszkiewicz, J. A. and Mavinic, D. S. (2002). Wastewater biosolids: an overview of processing, treatment and management. Journal of Environmental Engineering Science 1, 75-88.

Pagans, E., Barrena, R., Font, X., Sánchez, A. (2006). Ammonia emissions from the composting of different organic wastes. Dependency on process temperature. Chemosphere 62 (9), 1534-1542.

Palatsi, J., Gimenez-Lorang, A., Ferrer, I., Flotats, X. (2009) Start-up strategies of thermophilic anaerobic digestion of sewage sludge. Water Science & Technology 59(9), 1777-1784.

Panuvatvanich, A., Koottatep, T., Kone, D. (2009). Influence of sand layer depth and percolate impounding regime on nitrogen transformation in vertical flow constructed wetlands treating faecal sludge. Water Research 43, 2623-2630.

Peregrina, C.A., Lecomte, D., Arlabosse, P., Rudolph, V. (2006). Life Cycle Assessemnt (LCA) Applied to the Design of an Innovative Drying Process for Sewage Sludge. Process Safety and Environmental Protection 84, 270-279.

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).

Peruzzi, E., Macci, C., Doni, S., Mascinadaro, G., Peruzzi, P., Aiello, M., Ceccanti, B., 2009. Phragmites australis for sewage sludge stabilization. Desalination 246, 110-119.

Pempkowiak, J. and Obarska-Pempkowiak, H. (2002). Long-term changes in sewage sludge stored in a reed bed. The Science of the Total Environment 297, 59-65.

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).

Peruzzi, E., Macci, C., Doni, S., Mascinadaro, G., Peruzzi, P., Aiello, M., Ceccanti, B. (2009). Phragmites australis for sewage sludge stabilization. Desalination 246, 110-119.

Peruzzi, E., Masciandaro, G., Macci, C., Doni, S., Ceccanti, B. (2010). Pollutant monitoring in sludge treatment wetlands. In: 12th International Conference on Wetlands Systems for Water Pollution Control. 4-8 October. Masi F. and Nivala J. (ed) Venice (Italy).

Picek, T., Cizkova, H., Dusek, J. (2007) Greenhouse gas emissions from a constructed wetland plants as important sources of carbon. Ecological Engineering 31(2), 98–106.

Platzer, C., Mauch, K. (1997) Soil clogging in vertical flow reed beds – Mechanisms, parameters, consequences and… solutions? Water Science & Technology 35(5), 175-181.

Pomares, F., and Canet, R. (2001). Organic Waste Utilization in Agriculture: Origin, Composition and Characterization (In Spanish Aplicación agrícola de residuos orgáncos). Boixadera J. and Teira M.R. (ed.). Lleida (Spain).

Ponsá, S., Gea, T, Alerm, L., Cerezo, J., Sánchez, A. (2008). Comparision of aerobic and anaerobic stability indices throught a MSW biological treatment process. Waste Management 28 (12), 2735-2742.

Puigagut, J., Villaseñor, J., Salas, J. J., Bécares, E., García, J. (2007) Subsurface-flow constructed wetlands in Spain for the sanitation of small communities: A comparative study. Ecological Engineering 30(4), 312-319.

Reed, S.C., Crites, R.W. and Middlebrooks, E.J. (1988). Natural Systems for Waste Management and Treatment. McGraw Hill Book Co., pp261.

Regina K, Nykänen H, Silvola J., Matrikainen, J. (1996) Fluxes of nitrous oxide from boreal peat-lands as affected by peatland type, water table level and nitrification capacity. Biogeochemistry 35, 401–418.

Regulation (CE) No. 2003/2003 of the European Parliament and of the Council relating to fertilizers (Official Journal L 304, 21/11/2003, pp. 1194).

Ruggieri, L., Gea, T., Artola, A., Sánchez, A. (2008). Influence of different co-substrates biochemical composition on raw sludge co-composting. Biodegradation 19, 403-425.

Sánchez, M., Hernadez, S., Bravo, C., Antolin, J. M., Gallardo, T. (2010). Sustainable use of biodegradables products in agricultural fertilisation, reforestation, plant production and recovery of degraded areas (In Spanish) In: Special Orbit. International Congress about organic soils and organic matter from waste. 17-18 November. Valladolid (Spain).

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

Scaglia, B., Adani, F. (2008). An index for quantifying the aerobic reactivity of municipal solid wastes and derived waste products. Science of the Total Environment 394, 183-191.

Shaw, E.M. (1994). Hydrology in practice, 3rd edition. Chapman & Hall eds. London, UK, 1994.

Sheiran, A.S., Sheoran, V. (2006). Heavy metal removal mechanism of acid mine drainage in wetlands: a critical review. Minerals engineering 19(2), 105-116.

Singh, R.P., Agrawal, M. (2008). Potencial enefits and risks of land application of sew-age sludge. Waste Management 28, 347-358.

Soliva, M. (2001). Composting and Management of Organic Residues (In Catalan). Estudis i monografies nº21. Diputació de Barcelona. Barcelona (Spain).

Søvik, A.K., Augustin, J., Heikkinen, K., Huttunen, J.T., Necki, J.M., Karjalainen, S.M., Kløve, B., Liikanen, A., Mander, Ü., Puustinen, M., Teiter, S., Wachniew, P. (2006) Emission of the greenhouse gases nitrous oxide and methane from constructed wetlands in Europe. J. Environ. Qual. 35, 2360–2373.

Søvik, A.K., Kløve, B. (2007) Emission of N2O and CH4 from a constructed wetland in southeastern Norway. Science of the Total Environment 380, 28-37.

Stefanakis, A. I., Akratos, C.S., Melidis, P., Tsihrintzis, V. A. (2009). Surplus activated sludge dewatering in pilot-plant sludge drying reed beds. Journal of Hazardous Materials 172, 1122-1130.

Suh, Y., Roiusseaux P. (2002). An LCA of Alternative Wastewater Sludge Treatment Scearios. Resources, Conservation and Recycling 35, 191-200.

Sümer E., Weiske A., Benckiser G., Ottow J.C.G. (1995) Influence of environmental conditions on the amount of N2O released from activated sludge in a domestic wastewater treatment plant. Experientia 51, 419–422.

Summerfelt, S.T., Adler, P.R., Glenn, D. M., Kretschmann R.N. (1999). Aquaculture sludge removal and stabilization within created wetlands. Aquacultural Engineering 19, 81-92.

Svantröm, M., Fröling, M., Modell, M., Peters, W. A., Tester, J. (2004). Environmental assessment of Supercritical Water Oxidation of Sewage Sludge. Resources, Conservation and Recycling 41, 321-338.

Tanner, C.C., Sukias, J.P.S. (1995) Accumulation of organic solids in gravel-bed con-structed wetlands. Water Sci. Technol. 32, 229–239.

Tanner, C.C., Adams, D.D., Downes, M.T. (1997) Methane emissions from constructed wetlands treating agricultural wastewaters. Journal of Environmental Quality 26, 1056–1062.

Tarantini, M., Buttol. P., Maiorino, L. (2007). An environmental LCA of alternative scenarios of urban sewage sludge treatment and disposal. Thermal Science 3, 153-164.

Teiter, S., Mander, U. (2005) Emission of N2O, CH4 and CO2 from constructed wet-lands for wastewater treatment and form riparian buffer zones. Ecological Engi-neering 25(5). 528-541.

Terzaghi, K., Peck, R.B., 1967. Soil Mechanics in Engineering Practice. second ed. Wiley, New York.

Tiquia, S. M., Tam, N. F. Y., Hodgkiss, I. J. (1996). Effects of composting on phytotoxicity of spent pig manure sawdust litter. Environmental Pollution 93 (3), 249-256.

Troesch, S., Liènard, A., Molle, P., Merlin, G., Esser, D. (2008a). Treatment of septage in sludge drying reed beds: a case study on pilot-scale beds. Proceeding of Conference: 11th International Conference on Wetland Systems for Water Pollution Control. 1-7 Novembre. Indore (India).

Troesch, S., Liènard, A., Molle, P., Merlin, G., Esser, D. (2008b). Sludge drying reed beds: a full- and pilot-scales study for activated sludge treatment. Proceeding of Conference: 11th International Conference on Wetland Systems for Water Pollution Control. 1-7 November. Indore (India).

Tresch, S., Liénard, A., Molle, P., Merlin, G., Esser, D. (2009a). Sludge drying reed beds: a full- and pilot-scales study for activated sludge treatment. Water Science and Technology 60 (5), 1145-1154.

Tresch, S., Liénard, A., Molle, P., Merlin, G., Esser, D. (2009b). Treatment of septage in sludge drying reed beds: a case study on pilot-plant beds. Water Science and Technology 60 (3), 643-653.

Tsang K.R., Vesilind, P. A. (1990). Moisture distribution in sludges. Water Science and Technology 22 (12), 135-142.

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

Uggetti, E., Llorens, E., Pedescoll, A., Ferrer, I., Castellnou, R., García, J. (2009b). Sludge drying reed beds: a case study. Journal of Residuals Science and Technology 6(1), 57-59.

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.

Uggetti, E., Ferrer, I., Molist, J., García, J. (2011). Technical, economic and environmental assessment of sludge treatment wetlands. Water Research, 45 (2), 573-582.

Vincent, J., Molle, P., Wisniewski, C., Lienard, A. (2010). Sludge drying reed beds for septage treatment: towards design and operation recommendations. Proceeding of Conference: 12th International Conference on Wetland Systems for Water Pollution Control. 4-8 October. Venice (Italy).

Von Sperling and Andreoli (2007). Introduction to sludge management. In: Sludge Treatment and Disposal. Cleverson, Von Sperling & Fernandes Eds. IWA Publishing, London, UK, 2007.

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

Vymazal, J. (2005). Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment. Ecological Engineering 25(5), 478-490.

Walter, I., Cuevas, G., García, S., Martínez, F. (2000). Biosolid effects on soil and native plant production in a degraded semiarid ecosystem in central Spain. Waste Management & Resource 18 (3), 259-263.

Warmar, P.R., Termeer, W.C. (2005). Evaluation of sewage sludge, septic waste and sludge compost applications to corn and forage: Ca, Mg, S, Fe, Mn, Cu, Zn and B content of crops and soils. Bioresouce Technology 96, 1029-1038.

Wang, F.Y., Rudolph, V., Zhu, Z.H. (2008). Sewage sludge technologies. In: Enctyclopedia of Ecology (Ecological Engineering), 3227-3242. Jorgensen and Fath Eds. Elsevier Science Ldt Publishing, USA, 2008. ISSN/ISBN: 9780444520333

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 Research 37(18), 4453-4467.

Werther, J., Ogada, T. (1998). Sewage sludge combustion. Progress in energy and combustion science 25(1), 55-116.

Whiting, G. J., Chanton, J. P. (2003) Primary production control of methane emission from wetlands. Nature 364, 794-795.

Wiley, New York. Troesch, S., Liènard, A., Molle, P., Merlin, G., Esser, D. (2008). Sludge drying reed beds: a full and pilot-scales study for activated sludge treatment. Proceeding of Conference: 11th International Conference on Wetland Systems for Water Pollution Control. 1-7 November. Indore (India).

Yamaoka, M., Hata, K. (2003). Improvements in drying beds for non-concentrated sludge. Advances in Environmental Research 7, 721-725.

Ye, Z.H., Whiting, S.N., Qian, J.H., Lytle, C.M., Lin, Z.Q. and Terry, N. (2001). Wetlands and aquatic processes, trace elements removal from coal ash leachate by a 10 year old constructed wetland, Journal of Environmental Quality 30, pp. 1710–1719.

Yubo, C., Tieheng, S., Lihui, Z., Tingliang, J., Liping, Z. (2008). Performance of wastewater sludge ecological stabilization. Journal of Environmental Sciences 20, 385-389.

Zábranská, J., Dohányos, M., Jeníček, P., Kutil, J. (2000). Thermophilic process enhancement of excess activated sludge degradability – two ways of intensification sludge treatment in Prague central wastewater treatment plant. Water Science & Technology 41(9), 265-272.

Zhongpong, S., Bin, W., Wei, S., Wenming, S., Changzuo, W., Daian, Y., Zheng, L. (2010). Evapotranspiration estimation based on the SEBAL model in the Nansi Lake Wetland of China. Mathematical and Computer Modelling, doi:10.1016/j.mcm.2010.11.039

Zhou, L., Zhou, G. (2009). Measurement and modeling of evapotranspiration over a reed (Phragmites australis) marsh in Northeast China. Journal of Hydrology 372, 41-47.

Zucconi, F., Pera, A., Forte, M., De Bertoldi, M., (1981). Evaluating toxicity of innadure compost. Biocycle 22, 54-57.

Zwara, W., Obarska-Pempkowiak, H., (2000). Polish experience with sewage sludge utilization in reed beds. Water Science & Technology 41(1), 65-68.