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Performance of sludge treatment wetlands using different plant species and porous media

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ABSTRACT

The aim of this study was to evaluate the dewatering and mineralisation efficiency of three sludge treatment wetlands (STW) configurations differing on plant species (*Phragmites australis* and *Typha* sp.) and filter medium (gravel and wood shavings). Sludge dewatering and mineralisation were monitored in three pilots STW for 2 years. The sludge volume was reduced by 80% in all configurations tested, the total solids (TS) increased to 16–24% TS and the volatile solids (VS) decreased to 50% VS/TS. After a resting period of three months the biosolids showed a high stabilisation (dynamic respiration index around 0.26–0.70 $mgO_2/gVS\,h$), caused no phytotoxicity (germination index >100%), and had low heavy metals and pathogens concentrations (*E. coli* < 240 MNP/g; absence of *Salmonella*). The lack of statistical significance ($p > 0.05$) between the results obtained from the different STW configurations suggests that STW may be either planted with *P. australis* or *Typha* sp., and that wood shavings may replace gravel as filter medium.

1. Introduction

Sludge treatment wetlands (STW) consist of shallow tanks with a granular filter planted with emergent rooted wetland vegetation. Sewage sludge is loaded on the wetland surface, where it is dewatered by drainage through the granular filter and plant evapotranspiration. In this way, a concentrated sludge residue remains on the surface of the bed. After some days without feeding, thickened sludge is a new loaded, starting the following feeding cycle. During the feeding period, the sludge thickness increases around 10 cm/year [1]. When the sludge approaches the top of the tank, feeding is stopped during a final resting period (from 1–2 months to 1 year), in order to improve sludge dryness and mineralisation. The final product (biosolids) is subsequently withdrawn, starting the following operating cycle. During this treatment, the sludge composition changes, as a result of dewatering (drainage and evapotranspiration) and mineralisation processes [1]. The resulting final product may be suitable for land application [2], although it could be post-treated to improve sludge hygienisation [3].

Since the late 1980s, STW have been used in Europe for dewatering and stabilisation of sludge from aerobic and anaerobic digesters [1], conventional activated sludge [1,4,5], extended aeration systems [1,5–7], septic tanks [5,8,9], and Imhoff tanks [3]. The largest experience comes from Denmark, where there are over 140 full-scale systems currently in operation [10]. Other systems implemented in northern Europe are located in Poland [3,11], Belgium [12] and the United Kingdom [13]. In the Mediterranean region, full-scale systems are operating in Italy [14,15], France [5,9,16] and Spain [7]. According to the studies carried out during the last decades, in pilot and full-scale STW, the main STW design parameters include: bed number and dimensioning, sludge loading rate, plant species and granular medium, wet and dry period. Plant species and granular medium play an important role due to their contribution to water evapotranspiration and percolation, respectively. Plants are indeed a key element of STW, since they assist sludge dewatering and mineralisation. Compared to unplanted STW, planted ones achieve a higher TS concentration (18% TS vs. 20–21% TS) and sludge volume reduction (84–86% vs. 81%) [13]. Plant species used in treatment wetlands must be able to grow in watery, muddy, anaerobic conditions. In addition, they must tolerate variations in the water level, pH and high salinity [12]. The most widely used plant species in constructed wetlands for wastewater and sludge treatment is common reed (*Phragmites australis*) [17]. Hardej and Ozimek [11] evaluated the effect of sewage sludge on growth and morphometric parameters of *P. australis* and demonstrated the high adaptation capacity of common reed to the sewage sludge environment, observing that the shoot density was over two-fold that commonly found in natural systems. However, *P. australis* is often regarded as an invasive species and its use in constructed wetlands is either restricted or prohibited in United States and New Zealand [18]. Thus the investigation on new species

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suitable for STW can enhance STW spreading in different environments. For example, Cattail (*Typha* sp.) has been extensively used in wastewater treatment wetlands, due to its high evapotranspiration rate. In spite of this, it has been scarcely used in STW [19–21]. The literature highlights some problems with the adaptation of *Typha* sp. compared to *Echinochloa pyramidalis* and *Zizaniopsis bonarienses*, reflected into scarce plant grown [19,21]. Few studies have been recently carried out with different plant species [22,23]. However, up to date, the effect of plant species on sludge dewatering, mineralisation and hygienisation in STW has not been yet investigated.

The granular filter has a total height around 30–60 cm [24]. The filter has different layers of granular media: stones (diameter around 5 cm) at the bottom protect draining pipes, while gravel (diameter from 2 to 10 mm) and sand (diameter from 0.5 to 1 mm) provide a primary physical filtration and a rooting medium for plants during the start-up phase [6,13]. The filter material has a significant contribution in STW investment costs [25]. Thus, in order to reduce STW costs, it could be interesting to find a material able to substitute the granular medium without affecting STW performances. However, there is only a pilot scale study dealing with new filter composition, in which the sand layer was replaced by 5–10 cm of compost [5,26]. According to this research, the compost layer improved the reed growth, but did not influence the dewatering performance of the system.

The objective of this study was to assess new STW design configurations in order to reduce STW costs and widen the range of application of this technology. Sludge dewatering, mineralisation and biosolids properties for agricultural reuse were evaluated in 3 pilot STW differing in plant species (*P. australis*, *Typha* sp.) and granular medium (gravel, wood shavings).

2. Materials and methods

2.1. Experimental set-up

The experiments were carried out outdoors in a pilot plant located at the roof of the Department of Hydraulic, Maritime and Environmental Engineering of the Univeristat Politècnica de Catalunya in Barcelona, Spain (41°23′N, 2°21′E, 79 m A.M.S.L.). This pilot plant consists of three PVC containers with a surface area of 1 m^2 and a height of 1 m. Any possible climatic effect caused by the location of the plant on the roof of the building has not been considered, as the influence would be the same on all the beds. The plant was set up in winter 2008, and operated from March 2009 to March 2011. The STW treated thickened activated sludge collected in an extended aeration system from a wastewater treatment plant (WWTP) with a capacity of 10,000 population equivalent (PE) located near Barcelona, Spain. The sludge was manually loaded once per week from a corner of the bed. Notice that the high water content of the feeding sludge (about 2.5% TS) allows the uniform distribution over the bed surface. This feeding frequency was selected in accordance with the feeding procedure adopted in the full-scale systems located in the same region [7]. The sludge loading rate was approximately 20 kg TS/m^2·year from March 2009 to January 2010, and 40 kg TS/m^2·year from February 2010 to March 2011. In accordance with the recommendation provided in literature [1], a lower loading rate was applied during the first period in order to ease the plant growth.

Three STW configurations were compared (Fig. 1):

- STW 1 was planted with common reed (*P. australis*). The granular filter was constituted by 10 cm 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 cattail (*Typha* sp.). The granular filter was the same as in STW 1.
- STW 3 was planted with cattail (*Typha* sp.). The granular filter was constituted by 10 cm of stones (d_{50} = 250 mm), 30 cm of wood shavings ($d \sim 500$ mm) and 10 cm of sand (d_{50} = 1 mm).

Three perforated PVC pipes were placed at the bottom of each STW in order to collect the leachate and ease filter aeration.

2.2. Experimental procedure

The operating cycle of the plant comprised a feeding period (from March 2009 to March 2011) and a final resting period (from March 2011 to June 2011). During the feeding period, one composite sludge sample from each bed was taken every 4 months in order to monitor the season effect (campaigns I–V). During the final resting period one biosolids sample per bed was taken every month (campaigns VI–VIII). During each sampling campaign, the sludge was collected from 6 points of each bed and subsequently mixed to obtain one representative composite sample from each bed.

During the feeding period, sludge dewatering and mineralization processes were monitored by analysing: total and volatile solids (TS and VS) and chemical oxygen demand (COD). During the resting period the biosolids were characterised as follows: the stability was determined by the dynamic respiration index (DRI) in March, May and June 2011 (campaign V, VII and VIII); the germination index (GI) indicating the phytotoxicity, faecal bacteria indicators (*Salmonella* sp. and *Escherichia coli*), heavy metals and nutrients (Total Kjehldahl Nitrogen (TKN) and Total Phosphorous (TP)) were determined in March and June 2011 (campaigns V and VIII). The aim of monitoring these parameters during the final resting period was to determine the minimum duration of such a period, ensuring biosolids dryness and stabilisation in order to reduce transport costs and obtain an appropriate product for agricultural reuse.

Influent sludge was also characterised for TS, VS, COD, DRI, TKN and TP. The sludge height or thickness, representing the sludge residue accumulated within the beds after dewatering, was measured weekly before loading. Meteorological data were gathered from a municipal meteorological station located nearby in Barcelona (Fig. 2b).

2.3. Analytical methods

Sludge samples were collected using the Eijkelkamp soil coring kit and analysed (in triplicate) following the Standard Methods [27]. COD, TKN, TP and heavy metals were analysed from sludge dried at room temperature until constant weight; therefore the results are expressed on dry matter basis (kg TS).

The dynamic respiration index (DRI) was determined according to Adani et al. [28] and Barrena et al. [29]. The germination index (GI) was determined with cucumber (*Cucumis sativus*) and lettuce (*Lactuca sativa*) seeds as described by Barrena et al. [30]. Aqueous extracts of sludge were prepared by shaking a fresh sample with distilled water 1:10 (w/v) and filtering. After 7 days of incubation at 20 °C in darkness, the relative seed germination, relative root elongation and germination index (GI) were determined according to Eqs. (1)–(3) [31].

$$\text{Relative seed germination}(\%) = \frac{\text{number of seeds germinated in sludge extract}}{\text{number of seeds germinated in control}} \times 100 \quad (1)$$

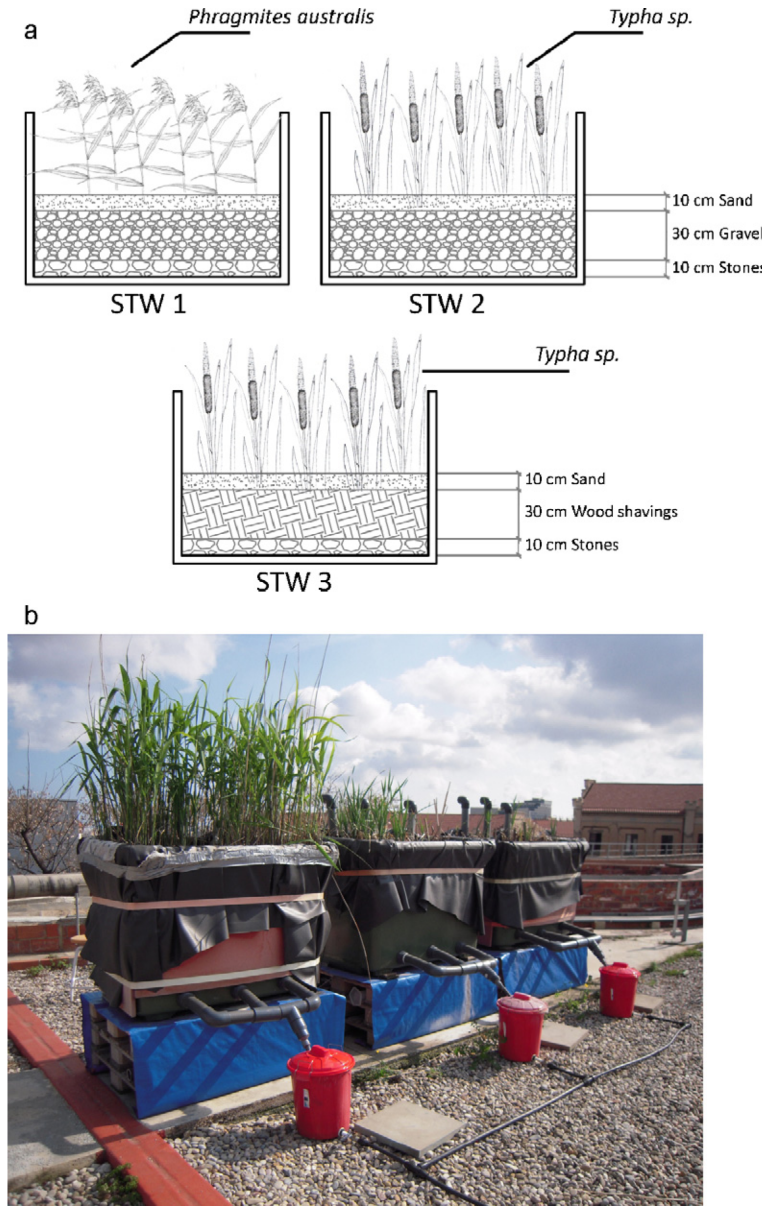

Fig. 1. Schematic diagram (a) and a picture (b) of the 3 sludge treatment wetlands (STW) of the pilot plant.

$$\text{Relative root growth}(\%) = \frac{\text{mean root length in sludge extract}}{\text{mean root length in control}} \times 100 \quad (2)$$

$$\text{GI}(\%) = \frac{(\%\ \text{seed germination}) \times (\%\ \text{root growth})}{100\%} \quad (3)$$

2.4. Statistics

The statistical significance of experimental results was evaluated by the ANOVA test using the Minitab 15.0 statistical software.

3. Results and discussion

3.1. Sludge dewatering

During the whole feeding period (2 years), each wetland was loaded with approximately 1.675 m^3 of sludge. On the other hand, the volume of sludge accumulated in each bed after the last 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. The results are satisfactory, even if a higher volume reduction was found in Greece where, after 12 years of operation, the volume reduction was 99% with a TS concentration around 50% [32]. Similarly to Melidis et al. [32], Stefanakis and Tsihrintzis [33] detected a volume reduction of 87–95%. Such high percentages are probably due to the weather conditions and the period of operation. Moreover, it should be taken in account that these studies were carried out in cylindrical tanks of 0.57 m^2 of surface. Thus the walls effect could have played an important role by enhancing sludge dewatering and consequently volume reduction.

After almost 1 year of operation (March 2010), the sludge thickness was around 22 cm and the TS below 10% on weight basis. During the second year of operation (from March 2010 to March 2011), the sludge layer increased by only 5 cm, with a TS concentration around 13–15%. This is attributed to the difference between plant growth during the first and second years. Indeed, a scarce root system development during the first months of operation can

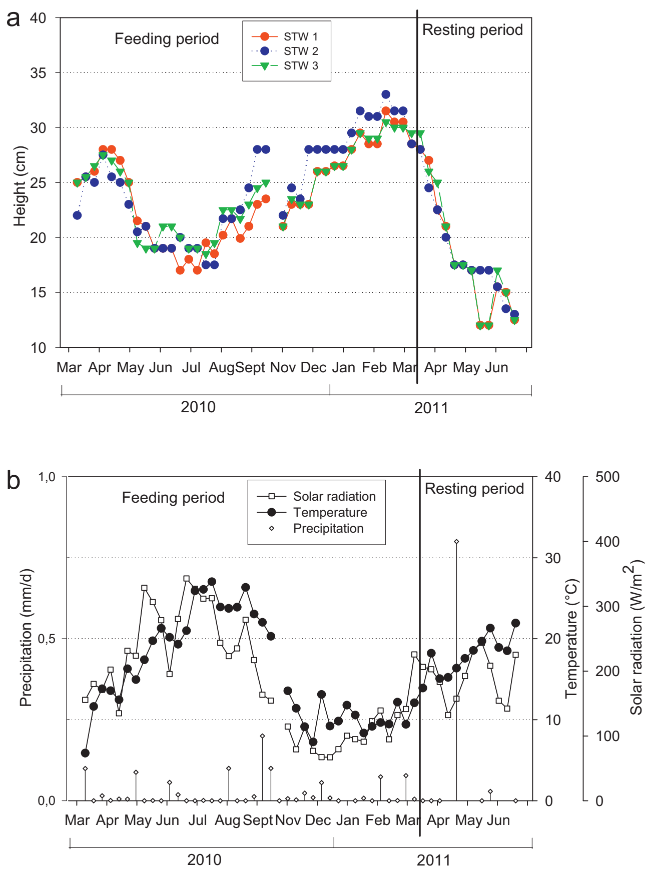

Fig. 2. Sludge thickness (a); precipitation, ambient temperature and solar radiation during the feeding and resting periods (b). The vertical line stands for the last sludge loading.

reduce sludge dewatering during the initial phase. In Danish systems with a sludge loading rate of 60 kg TS/m^2·year the sludge layer increases some 10 cm/year [1]. The dryer and warmer climate of the Mediterranean region compared to Northern Europe and the lower sludge loading rate (40 kg TS/m^2·year) may explain the difference between this value (10 cm/year) and our results (5 cm/year). Besides, our trial lasted 2 years and a longer-term study is needed in order to confirm these results. Nevertheless, Stefanakis et al. [23] obtained similar values in a pilot plant located in the Mediterranean region during the second year of operation (sludge layer increase of 5 and 15 cm/year for 30–60 and 75 kg TS/m^2·year, respectively). In general, the sludge thickness followed a similar pattern in the 3 STW (Fig. 2a). The highest value (around 30 cm) was recorded in winter (between January and March 2011), while during the spring the sludge thickness decreased rapidly reaching the lowest values in July 2010 (17 cm). The pattern of the sludge thickness highlights the importance of climate conditions in sludge dewatering. Indeed, the trend followed by the sludge thickness was clearly opposite to ambient temperature and solar radiation (Fig. 2b), which have a direct influence on the evapotranspiration rate. This confirms the importance of the evapotranspiration rate on sludge dewatering and volume reduction.

In accordance with the sludge thickness, the TS concentration followed a seasonal pattern during the feeding period (Fig. 3a). The influent TS concentration was almost constant around 2%. In all the STW, the TS concentration ranged around 11–13% in autumn 2009 and spring 2010, increased to 41–46% in summer 2011 and decreased to 13–15% in autumn 2010 and spring 2011. Indeed, as reported above, the sludge thickness was the lowest in summer 2010 (below 20 cm) and the highest in winter 2010 (around 30 cm).

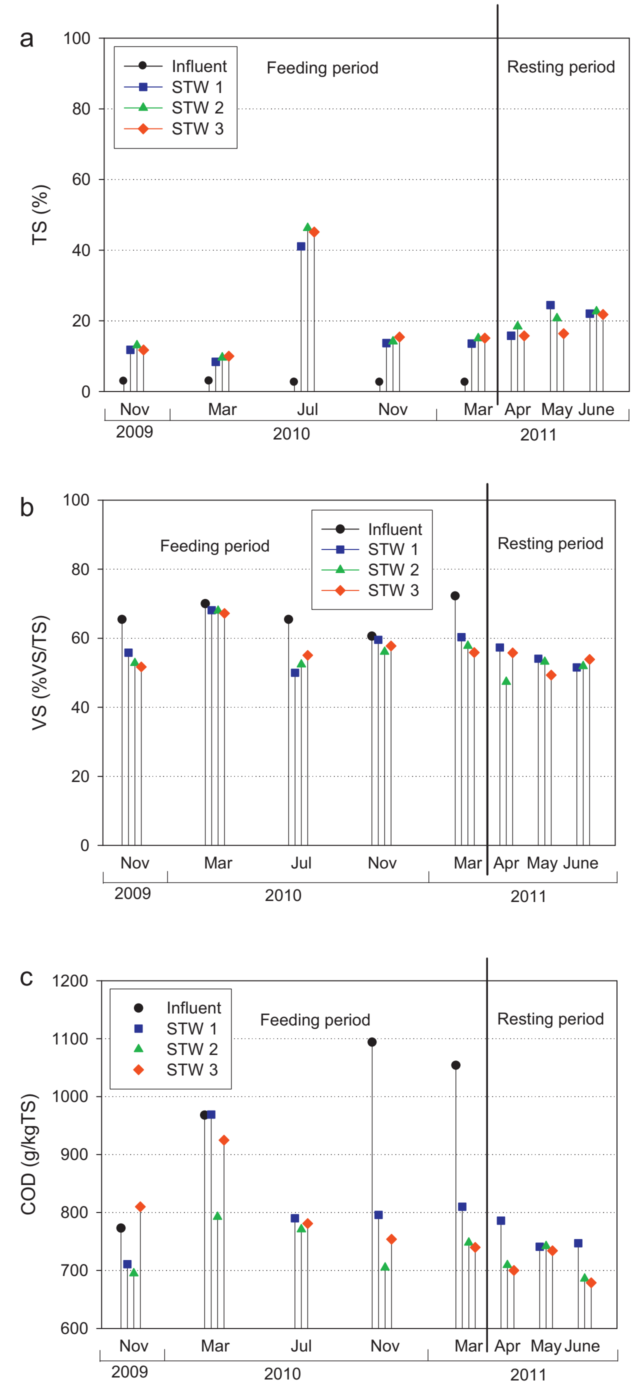

Fig. 3. Total solids (TS) (a), volatile solids (VS) (b) and chemical oxygen demand (COD) (c) concentration in the influent and the in 3 sludge treatment wetlands (STW) during the feeding and resting periods (number of samples $n = 3$). The vertical line stands for the last sludge loading.

Similar seasonal oscillation where found in Greece (20–30 cm in summer and 35–50 cm in winter) [23]. In this case the higher absolute values are probably the results of the higher sludge loading rate applied (60–70 kg TS/m^2·year) The TS concentration found in this study (ranging between less than 20% in winter and more than 40% in summer) are in accordance with the values found in other pilot plants located in France [26] and Greece [32]. The TS concentrations found by Stefanakis and Tsihrintzis [23] with a higher sludge loading rate were similar to our results, suggesting that loading rates higher that 40 kg TS/m^2·year could be successfully applied in the Mediterranean region. Indeed, similar dewatering efficiencies from several full-scale systems are reported in the literature [24].

After the last sludge loading (March 2011) there was a sharp decrease in the sludge thickness in all the STW (Fig. 2a). During only 3 months resting, the sludge thickness decreased from 28 cm to 12 cm in all the STW. Notice that during the spring, ambient temperature and solar radiation also increased. The decrease in sludge thickness is supported by the increase in TS concentration (Fig. 3a). Indeed, from March to June 2011 the TS concentration increased by 9%, 7% and 6% in STW 1, STW 2 and STW 3, respectively; reaching a final concentration around 22% TS in all the STW. This is attributed to the time elapsed from the last loading, and probably to the higher temperature of May compared to April 2011. Consistent values (sludge thickness decrease from 25 to 15 cm) were found during the resting period between March and September in a pilot plant located in Greece [23].

3.2. Sludge mineralisation and stabilisation

Sludge mineralisation was determined by the VS and COD. The influent VS concentration ranged between 60% and 72% VS/TS (Fig. 3b), while the COD ranged between 770 and 1.040 gO_2/kg TS (Fig. 3c). The VS concentration of the STW (50–60% VS/TS) was rather higher than in the literature (36–42% VS/TS) [24], probably due to the high VS concentration of the influent indicating its partial mineralisation (up to 70% VS/TS). Nevertheless, a certain VS removal was noticeable in all campaigns. VS reduction ranging from 2% in winter (March and November 2010) to 10–15% in summer (July 2010) is representative of the microbial activity enhancement by temperature. A similar pattern was found by Stefanakis et al. [34], who pointed out a reduced microbial activity at low temperatures. Similarly, the COD removal was low (8%), with values around 700 gO_2/kg TS in all STW after 3 resting months (Fig. 3c). However, a certain COD removal was observed in all campaigns. The best results were obtained during the second year, when the COD decreased from 1000 to 700 gO_2/kg TS (30% COD removal). This fact could be due to better plant growth which enhances the bacterial activity. Again, no significant differences between STW configurations were observed ($p > 0.05$). During the final resting period the VS removal was relatively low (2–9%), reaching values around 52% VS/TS in all the STW (Fig. 2c). Thus, it seems that organic matter mineralisation takes place already during the sludge feeding period, and only slightly during the final resting period. Besides, the reduced sludge moisture may inhibit microbial activity which consequently reduced the organic matter mineralisation.

The stabilisation of the final product (i.e. biosolids) was evaluated by the DRI. 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, the higher the stability. This parameter is widely used to measure the readily biodegradable organic fraction that has already been decomposed in compost samples. In our study, the DRI was used to determine the minimum duration of the final resting period in order to obtain biosolids with the same stabilisation as composted ones. As shown in Table 1, DRI_{24h} of the influent was 9.10 mgO_2/gVS h. During the resting period of the STW, the DRI_{24h} decreased from 2.24–3.69 mgO_2/gVS h (March 2011) to 0.26–0.70 mgO_2/gVS h (June 2011). This confirms that sludge stabilisation was taking place already during the feeding period and continued during the resting period. The DRI of biosolids obtained after 3 resting months was lower than 1 mgO_2/gVS h in all the STW, thus it can be considered a stabilised product according to the European Working Document on Biological Treatment of Biowaste [35]. Thus, the results from the present study indicate that the final product from all the STW can be considered a stabilised product already after 3 resting months in the Mediterranean environment and it does not require any stabilisation post-treatment. Notice that these results were obtained in the Mediterranean region during summer, when the temperature, solar radiation and evapotranspiration rate are the highest of the year. The values found in this study are in accordance with recent results obtained in a similar climate conditions. In fact, values ranging between 0.04 and 0.42 mgO_2/gMS h where detected in a recent study carried out in Greece during the same season [22].

Table 1
Dynamic respiration index (DRI) in biosolids from the 3 sludge treatment wetlands (STW) at the beginning (March 2011) and at the end (May 2011) of the resting period (number of samples I = 3).

Sampling	Source	DRI_{24h} (mgO_2/gVS h)	DRI_{24h} (mgO_2/gMS h)
	Influent	9.10 ± 0.51	7.45 ± 0.51
Campaign V March 2011	STW 1	3.69 ± 0.33	2.29 ± 0.33
	STW 2	2.24 ± 0.11	0.31 ± 0.03
	STW 3	3.33 ± 0.43	0.37 ± 0.01
Campaign VII May 2011	STW 1	1.44 ± 0.05	0.79 ± 0.05
	STW 2	1.40 ± 0.08	0.76 ± 0.08
	STW 3	1.54 ± 0.03	0.85 ± 0.03
Campaign VIII June 2011	STW 1	0.26 ± 0.02	0.14 ± 0.02
	STW 2	0.62 ± 0.01	0.31 ± 0.03
	STW 3	0.70 ± 0.03	0.37 ± 0.01

3.3. Phytotoxicity and hygienisation

Phytotoxicity was tested by means of the germination index. It has been proven to be one of the most sensitive parameters accounting for both low toxicity affecting root growth and heavy toxicity affecting germination [36]. The results of germination tests indicate that the sludge did not cause phytotoxicity (Table 2). In fact, the germination index resulting from seed germination and root growth in sludge samples was higher than in water samples, indicating the absence of toxic effects already at the beginning (March 2011) and at the end (June 2011) of the resting period. The relative seed germination, after 7 days of incubation, was almost always higher than 100% (Table 2). In fact, according to the calculation (Eqs. (1)–(3)), values higher than control samples indicate that the sludge has no toxic effects limiting seeds germination. On the contrary, according to our results, sludge has positive effects on seed germination and root growth. In all samples, the root elongation was near 100% or ever higher for lettuce (152–164%). Consequently, the germination index ranged between 181% and 224% for lettuce and from 94% to 106% for cucumber. As for the other parameters, no significant differences between the three STW configurations ($p > 0.05$) were found. Similar results were found in a recent study investigating the GI in a pilot scale STW with different sludge loading rates [22]. According to this study, the GI oscillated between 74% and 176% after 5 months resting and no significant differences were found between the sludge loading rates applied.

Regarding heavy metals (Table 3), the concentrations were clearly below the limits set by current European legislation [37] already after the last loading. Surprisingly, there was no heavy metals accumulation during the resting period, since heavy metals concentration remained almost constant in all beds (Table 3). Most probably the concentration of heavy metals was already low in the influent sludge, since it was obtained from an urban WWTP.

Not only the stabilisation but also the hygienisation of the biosolids is critical for land application. Sludge hygienisation occurs at therophilic temperatures (>50°) which are not reached in STW. For this reason biosolids post-treatment is nowadays required in many countries. In this study, faecal bacteria indicators were analysed at the beginning and at the end of the resting period (Table 4). *Salmonella* sp. was never detected during the resting

Table 2
Germination index calculation in biosolids from the 3 sludge treatment wetlands (STW) (number of samples $n = 3$).

Sampling	Source	Relative seed germination (%)		Relative root growth (%)		Germination index (%)	
		Lettuce	Cucumber	Lettuce	Cucumber	Lettuce	Cucumber
	Control	100	100	100	100	100	100
Campaign V 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
Campaign VIII June 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

Table 3
Heavy metals (mg/kg TS) in biosolids from the 3 sludge treatment wetlands (STW) at the beginning (March 2011) and at the end (June 2011) of the resting period (number of samples $n = 3$).

Sampling	Source	Cd	Cu	Hg	Ni	Pb	Zn
Campaign V March 2011	STW 1	<0.5	138	<0.5	99	49	829
	STW 2	<0.5	75	<0.5	50	31	643
	STW 3	<0.5	103	<0.5	71	26	517
Campaign VIII June 2011	STW 1	0.7	–	9	11	13	238
	STW 2	0.4	–	5	12	10	227
	STW 3	0.7	–	3	11	18	295
Directive 86/278/EEC		20–40	1000–1750	16–25	300–400	750–1200	2500–4000
3rd Draft EU Working Document on sludge		10	1000	10	300	750	2500

period, whereas *E. coli* decreased from 3500–3900 MPN/g after the last feeding (March 2011) to values below 240 MPN/g after three resting months (June 2001), below the limits proposed by the 3rd Draft Working Document on Sludge (*E. coli* < 500 MNP/g) [38].

3.4. Nutrients

The concentration of nutrients is shown in Table 5. TKN concentration in sludge influent oscillates around 6% TKN/TS. After the last sludge loading (March 2011) the TKN concentration varied between 6% TKN/TS in STW 1 and 4% TKN/TS in STW 2 and STW 3. During the following months, it decreased slowly to values around 3% TKN/TS in the final product (June 2011). The nitrogen decrease observed during the final resting period is a result of sludge mineralisation, ammonification and plant uptake [39]. On the whole, the TKN concentration was quite low (around 3% TKN/TS), in accordance with previous studies [28,29]. The differences between STW configurations were not significant ($p > 0.05$). On the other hand, the TP concentration ranged between 1% and 4% TP/TS in the influent sludge; while lower concentration were detected during the whole resting time, decreasing from 0.3–0.6% TP/TS (March 2011) to 0.01% TP/TS (June 2011). This pattern can be attributed to phosphate immobilisation in microbial cells [40] or plant uptake [39]. In general, sewage sludge is characterised by a considerable variability in nutrient content, depending on the wastewater source and treatment process. Thus, monitoring of these parameters is essential at least at the end of the process in order to ensure appropriate dosages of sludge in agriculture.

3.5. Performance comparison

In this work two different plant species and gravel filter configurations were compared to assess their suitability for sludge dewatering and mineralisation. *P. australis*, planted in STW 1 was compared with *Typha* sp. planted in STW 2 and STW 3. In addition, in STW 3 the gravel layer was replaced by wood shavings, to study the reuse of this solid waste as filter medium for water percolation. The results indicate similar dewatering and mineralisation performances in all the STW configurations, at least during 2 years of operation. According to the one-way ANOVA, no significant differences were found between TS, VS, COD concentrations of different

Table 4
Salmonella and *Escherichia coli* in biosolids from the 3 sludge treatment wetlands (STW) at the beginning (March 2011) and at the end (June 2011) of the resting period (number of samples $n = 3$).

Sampling	Source	*Salmonella* (absence/presence in 25 g)	*E. coli* (nmp/g)
Campaign V March 2011	STW 1	Absence	6600
	STW 2	Absence	4500
	STW 3	Absence	3900
Campaign VIII June 2011	STW 1	Absence	93
	STW 2	Absence	240
	STW 3	Absence	150
Directive 86/278/EEC		–	–
3rd Draft EU Working Document on sludge		Absence	<500

Table 5
Total Kjeldahl Nitrogen (TKN) and Total Phosphorus (TP) in biosolids from the 3 sludge treatment wetlands (STW) during the resting period (number of samples $n = 3$).

Sampling	Source	TKN (%TS)	TP (%TS)
	Influent	6.8–6.5	1.1–3.9
Campaign V March 2011	STW 1	6.19	0.57
	STW 2	4.26	0.29
	STW 3	4.35	0.25
Campaign VI April 2011	STW 1	4.85	0.01
	STW 2	3.65	0.01
	STW 3	4.45	0.01
Campaign VII May 2011	STW 1	3.30	0.03
	STW 2	2.75	0.02
	STW 3	5.72	0.03
Campaign VIII June 2011	STW 1	3.85	0.14
	STW 2	3.36	0.02
	STW 3	3.50	0.03

wetlands ($p > 0.05$). Moreover, the parameters analysed during the resting period (stabilisation and germination index, heavy metals, pathogens and nutrients) indicate that the biosolids from all the STW configurations have the same characteristics. On the whole, all the configurations proposed in this study seem suitable for sludge treatment in STW. This means that *P. australis*, which is widely used in STW, could be substituted by *Typha* sp., which is a non-invasive species and can enlarge the range of application of this technology. This 2-year study also shows that the gravel layer can be replaced by wood shavings without affecting the treatment performance. It should be noticed that the life-span of wood shaving would probably be short in comparison with gravel. However, this material is a waste that can be easily withdrawn and replaced during the emptying operation (every 5–10 years). Finally, changing the filter composition allows the reuse of a waste material, reducing the exploitation of natural resources. This can significantly decrease the environmental impact and investment cost of STW [25]. Indeed, related experiments were conducted in constructed wetlands for wastewater treatment [41] in order to test the nutrient removal efficiency of a substrate with wood chips and humus material.

4. Conclusions

In this study the performance of different STW configurations was evaluated by comparing 2 plant species (*P. australis*; *Typha* sp.) and 2 filter media (gravel; wood shavings). The results obtained in terms of sludge dewatering (TS increase to 21–22%) and mineralisation (VS reduction to 50% VS/TS) were satisfactory and fell into the range of values found in the literature. Moreover, the biosolids were characterised by a high stability (DRI 0.26–0.70 $mgO_2/gVS\,h$), caused no phytotoxicity (germination index >100%), and had low heavy metals and pathogens concentrations. Such parameters indicate the quality of the final product, which could be seen as an organic fertiliser. Besides, the lack of statistical significance between STW configurations ($p > 0.05$) suggests that STW may be either planted with *P. australis* or *Typha* sp., and that wood shavings may replace gravel as filter medium without affecting the treatment performance. It should be taken into account that this is a 2-years pilot scale study; a longer term study possibly at full-scale would be needed in order to confirm these results.

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