# Greenhouse gas emissions from sludge treatment reed beds

Yubo Cui, Shunli Zhang, Zhaobo Chen, Rui Chen and Xinnan Deng

## ABSTRACT

Sludge treatment reed bed systems (STRBs) are considered as an alternative technology for surplus sludge treatment. Organic matter is decomposed by various microbial reactions, resulting in gases such as $CO_2$ and $CH_4$ emitting into the atmosphere. The aim of this study is to investigate gas emission from STRBs. The static transparent chamber was adopted to measure gas emission; it allows sunlight to enter and plants to photosynthesise. The comparison of total solids and volatile solids showed STRBs have a higher efficiency in dewatering and mineralization than a conventional unplanted sludge drying bed (USDB). The $CO_2$ emission ranged from 28.68 to 100.42 g $CO_2$ $m^{-2}$ $d^{-1}$ in USDB, from 16.48 to 65.18 g $CO_2$ $m^{-2}$ $d^{-1}$ in STRBs; $CH_4$ emission ranged from 0.26 to 0.99 g $CH_4$ $m^{-2}$ $d^{-1}$ in USDB, from 0.43 to 1.95 g $CH_4$ $m^{-2}$ $d^{-1}$ in STRBs. Both gas fluxes decreased towards the end of vegetation and reached the highest rates during the hot and dry summer. After the system was loaded by sludge, the fluxes of $CO_2$ and $CH_4$ significantly decreased in the USDB, whereas they increased in STRBs. In terms of $CO_2$ equivalent, the global warming potential of $CH_4$ was 13.13 g $CO_2$eq $m^{-2}$ $d^{-1}$ and 15.02 g $CO_2$eq $m^{-2}$ $d^{-1}$ in USDB and STRBs, respectively.

**Key words** | greenhouse gases, sludge treatment reed beds, surplus activated sludge

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## INTRODUCTION

The major concern of wastewater treatment processing is its production of large amounts of surplus sludge. During recent decades, with the rapid growth of population and industrialization (Hong *et al.* 2009), the number of municipal and industrial wastewater treatment plants has dramatically increased. The surplus sludge produced from biological wastewater treatment consists of excessive biomass. Thus, it is characterized by low solids (0.5–2% d.s.) with high organic matter (loss on ignition: 50–80% TS) (Wang *et al.* 2008). Owing to these characteristics, surplus sludge has a relatively large volume and is difficult and expensive to treat and gather. So surplus sludge treatment involves stabilization and dewatering. Stabilization reduces the biodegradable parts of organic matter. Dewatering reduces the volume of the sludge (Uggetti *et al.* 2011).

Constructed wetlands are widely applied in the treatment of different types of wastewater such as municipal, industrial and agricultural wastewater, to reduce pollutant matters of water bodies, showing the economic, social and environmental benefits (Uggetti *et al.* 2012). Constructed wetlands, such as sludge treatment reed bed systems (STRBs), have been in operation in Northern Europe since 1990 (Nielsen *et al.* 2014), which were recently adapted to treat surplus sludge in China. Compared to conventional sludge treatment technologies, more advantages lie in STRBs: the influence on the environment is smaller because it needs little energy and uses no chemicals. Furthermore, it is easier to operate with higher efficiency and at lower cost. In addition, the final production of sludge treatment is suitable for agricultural applications (Nielsen 2005).

The STRB system consists of a series of beds, each bed filled with stones, gravel and sand layers, with common reed planted in these beds. Surplus sludge is usually pumped and spread into the beds by loading pipes, so the sludge is rapidly fed to the beds' surface and excess water is drained by the drainage set into the bottom of the beds (Uggetti *et al.* 2010). Consequently, concentrated sludge residue remains on the surface. Therefore, it is vital not to overspread or overload; if this happens, dewatering and degradation will be less efficient, odor nuisances will take place, etc. (Nielsen & Willoughby 2005). The loaded beds need a resting period for dewatering. After resting for several

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days, the system is fed anew, starting subsequent feeding and resting cycles (Nielsen 2005; Nielsen 2007).

In such systems, sludge residue is dried by dewatering and stabilized by mineralization of the organic matter. The dewatering process is mainly due to percolation through the layers and drainage, and partly evapotranspirated by reeds (Uggetti *et al*. 2012). The organic matter of sludge residue is removed by microbial reactions such as aerobic respiration, fermentation, methanogenesis, etc. (Peruzzi *et al*. 2013; Nielsen *et al*. 2014). Through such reactions, organic matter is converted to various gases, mainly $CO_2$ and $CH_4$ (Olsson 2014). The gases are emitted from sludge residue by diffusion through the surface and active transportation through the pores of the reeds. The presence of plants in STRBs may increase gas emissions from sludge residue. Plants affect microbial reaction and their byproducts, such as root exudates, which contribute to available carbon in the sludge residue (Picek *et al*. 2007). Root exudates are biodegradable and easily decomposable, and they can be decomposed and converted to $CO_2$ and $CH_4$ by microorganisms, which will increase gas emissions (Duan *et al*. 2009). According to previous research, emissions from wastewater treatment constructed wetlands range from −32 to 38,000 mg $CH_4/m^2$ d (Søvik & Kløve 2007). In Olsson's research (Olsson *et al*. 2014), the experiment measured $CO_2$ and $CH_4$ emissions in STRBs, an occasionally loaded sludge depot and a natural reed wetland, which showed relatively high $CH_4$ emissions in the sludge depot (2 mg $m^2$ $h^{-1}$ compared to 0.5 mg $m^2$ $h^{-1}$ and 0.2 mg $m^2$ $h^{-1}$ in the STRB and natural wetland, respectively); $CH_4$ emissions were very low in the STRB although $CO_2$ emissions were high (1,200 mg $m^2$ $h^{-1}$ compared to 500 mg $m^2$ $h^{-1}$ and 100 mg $m^2$ $h^{-1}$ in the sludge depot and natural wetland, respectively).

$CO_2$ and $CH_4$ are well known as greenhouse gases (GHG) due to their ability to absorb solar radiation and its consequences on climate change. Therefore, it is important to maintain low GHG emissions and to limit global warming (IPCC 2007). $CH_4$ is produced in anaerobic environments and $CO_2$ is produced by the aerobic respiration of sludge residue and plants. The gas dynamics are strongly influenced by numerous factors, especially temperature and water content. In addition, temperature also has a strong impact on gas fluxes, making it seasonal (Uggetti *et al*. 2010).

There is very limited knowledge about GHG emissions from STRBs. Considering the increasing application of STRBs, it is important to study the GHG emissions from STRBs, and their relation to STRBs' functions and sludge properties. The aims of this study were to compare greenhouse gas emissions between an unplanted sludge drying bed (USDB) and STRBs.

## METHODS

### Study sites

The study was conducted at Dakai wastewater treatment plant (WWTP), Dalian, China, where municipal wastewater is treated through activated sludge treatment plants. The excess sludge from the WWTP was treated in an STRB system (Cui *et al*. 2013). STRBs with a total surface of 9.0 $m^2$ were established in 2012 to treat surplus activated sludge. The surface area of each bed was 3.0 $m^2$, with a width of 1.0 m and length of 3.0 m. The system's arrangement and structure are shown in Figure 1(a) and 1(b). The beds were numbered from left to right in consecutive order: 1, 2 and 3 as shown in Figure 1(b). Each bed was filled with 20 cm slag, 20 cm gravel, 5 cm coarse sand, and 20 cm quartz sand blended with fine sand in the top 3 cm depth layer. In addition, 65 cm were reserved for sludge storage. Bed 1 was plant-free. Beds 2 and 3 were planted with common reed (Phragmites australis) with 15 pots/$m^2$. Beds 1 and 2 were set with aeration pipes equally spaced and connected along the draining pipes, while Bed 3 had no aeration pipe. The STRBs were fed once per week with surplus sludge (0.3 $m^3$ of surplus sludge per week) from a pipe located along the width of the bed. The influent was surplus activated sludge with a high water content; such a characteristic allows sludge to flow along the basin and the influent to remain in the basin while the water filters through the sludge residue layers and lower layers as shown in Figure 1(a). The surplus activated sludge with 0.86% dry weight (d.w.) was loaded on the bed with an average sludge loading rate of 15.67 kg d.w. $m^{-2}$ $y^{-1}$ (Cui *et al*. 2012). The samples were collected in 2013, and the systems had already been fed with sludge since 2012; during each year, the loading period lasted 8 months, corresponding to a maximum sludge layer increasing rate of approximately 3 cm/year with an operating cycle life span of 2 years.

### Sludge sampling

The properties of influent sludge and sludge residue were measured in the STRBs. Samples were collected from the top layer and bottom layer, corresponding to 0–2 and −6 ± 1 cm down the sludge residue surface at three

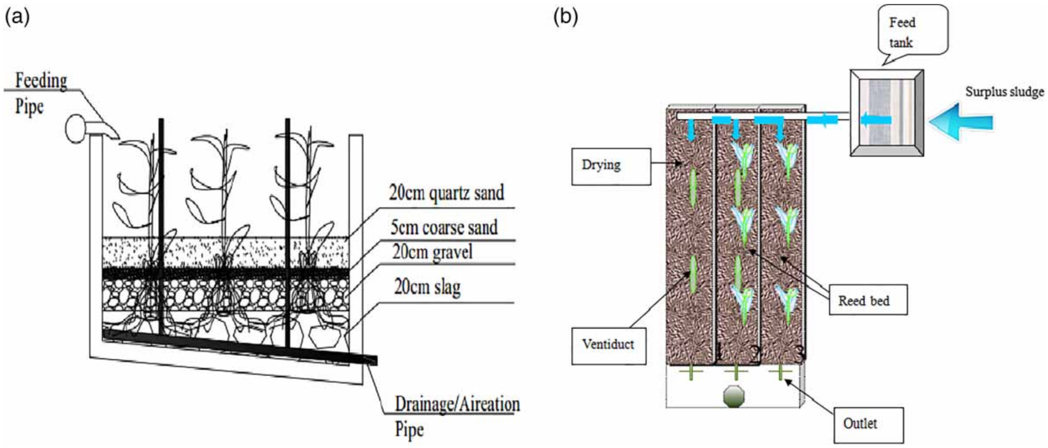


**Figure 1** | (a) Schematic diagram of an STRB planted with reeds. (b) Schematic drawing of the whole STRB system: the beds are consecutively numbered from left to right, 1, 2 and 3. Bed 1 was plant-free. Beds 2 and 3 were planted with common reed. Beds 1 and 2 were set with aeration pipes, while Bed 3 had no aeration pipe.

random spots. The STRBs and USDB were sampled twice in every month from May to December, loading periods were from April to November, and the samples were collected the day just before loading. Therefore, the sludge residue was resting for 6 days during the loading period before its next feeding. According to the local weather and season, results were analyzed seasonally. To assess sludge qualities, the following parameters were measured: total solids (TS) and volatile solids (VS). For the estimation of TS, the samples were dried in a ventilated oven at 105 °C until they reached a constant weight. The dried samples were ground using a mortar, and sub-samples were burned in a muffle furnace at 550 °C for 30 min to estimate VS.

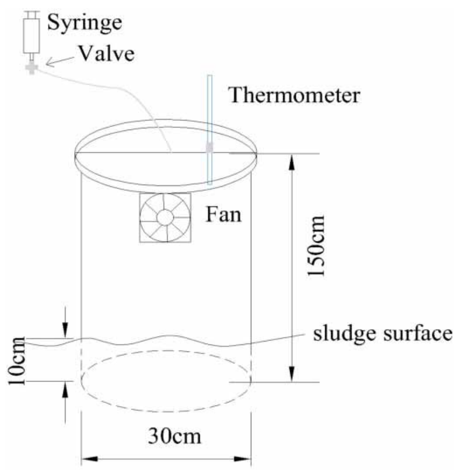


**Figure 2** | Diagram of a closed gas sampling chamber (Livingston & Hutchinson 1995).

## Gas sampling and analysis

Gas samples were collected using the static chamber method (Livingston & Hutchinson 1995). The chamber was made of plastic with a diameter of 30 cm and height of 150 cm, the total volume was 26.4 L (Figure 2; Liikanen *et al.* 2006a, b). The gas samples were collected on the same day as sludge residue sampling. Gases were sampled at 0, 20, 40 and 60 min, the increase in the concentration of the emissions of $CH_4$ and $CO_2$ occurred over time with linear regression. On the sampling day, gas was collected every 2 hours during the day and every 4 hours during the night. The weighted average method was used to calculate average daily values. The lid of the chamber contained a rubber septum allowing for collection of gas samples through syringes; a thermometer was inserted inside the chamber to record the chamber temperature. Each chamber was equipped with a fan to mix gas inside. The chamber was set approximately 10 cm into the sludge residue. To limit disturbance from soil structure changes right after installation, a plastic collar with the same diameter as the chamber was set and left in the beds for at least one day before collection; it was also placed approximately 10 cm into the sludge residue. During sampling, the collar was replaced by the chamber in the same spot.

Gas samples were extracted using syringes attached to a short plastic tube on the lid. After sampling, the tube was immediately closed by rubber spring clip. The gas in the syringes was injected into evacuated plastic gas bags (Replete Ltd, Dallian, China). The bags were stored at room temperature and analyzed the next day. The gas samples were analyzed using a gas chromatograph (GC-2010, Shimadzu Corporation, Kyoto, Japan) with a flame ionization detector; $CO_2$ was directly detected by a portable gas detector

(GT901, Shenzhen Keernuo Technology Co., Ltd, Shenzhen, China) plunged into the rubber septum. Fluxes of $CH_4$ and $CO_2$ were determined from the increase in concentration in the chamber over time with regression analysis.

### Data analysis

For $CH_4$, global warming potential (GWP) of 25 $CO_2$-equivalents on a 100-year time scale (IPCC 2007) was used to calculate the number of $CO_2$-equivalents in 103 kg ha$^{-1}$ y$^{-1}$ emitted as $CH_4$. Gas emission was analyzed by site and season in a two-way analysis of variance (ANOVA). The $p$-value was used to identify significant differences at the 5% significance level. Prior to testing, the assumption of equal variances was checked using Levene's test ($p > 0.05$) and data were appropriately transformed if the assumption was not met. The statistical analyses were run on SPSS16.0 software. (IBM, Armonk, New York, USA).

## RESULTS AND DISCUSSION

### Sludge properties

The results summarized in Table 1 and Table 2, indicate that STRBs are efficient for sludge dewatering and stabilization. The influent sludge had a low TS content ranging from 0.28 to 0.86% with an average value of 0.55%, which had no relation to season or weather during the observation. The influent sludge had a relatively low organic matter concentration (average 60 ± 3% VS/TS), due to a relatively high solids retention time during the wastewater treatment process. In general, the TS of the surplus increased from an average 0.55% in the influent to 14–25% within Bed 1 and more than 30% in the bottom of Bed 2 and Bed 3. The results also indicate that STRBs (Bed 2 and Bed 3) had a relatively higher TS than the USDB; there was no significant difference between Bed 2 and Bed 3 due to a short resting time and thin sludge residue layer. These results demonstrate the importance of plants to STRBs in terms of plant evapotranspiration. In addition, there was no significant difference between the beds in spring or in winter, because of the length of the frost period in late winter. In this study, the sludge rested in water and then the sludge residue began to thaw; in late spring, reeds grew and then thrived in the summer, and were capable of dewatering and growing in autumn and early winter due to relatively warm and damp weather. Compared with the conventional centrifuges, which are able to reach 14–18% TS with surplus activated sludge (Gonçalves *et al.* 2007), the STRBs have higher efficiency and lower energy consumption (Uggetti *et al.* 2010).

Organic mineralization is shown by a 7% decrease in VS. Indeed, the stable VS all year-round in the bottom layers of the beds indicates a quite good degree of stabilization. The VS decreased from an average 60% in the influent to 52% in the bottom layers. In summer, the VS content reached the lowest point, which indicated that VS content was mostly degraded in correspondence with the vigorous growth of the plants. The difference between the surface and bottom layer compared with Uggetti's research was relatively small, which contributed to the shallow depth of the stored sludge (Uggetti *et al.* 2012). Plants not only play an important role in dewatering, but also in sludge stabilization or mineralization, which possibly results from the transportation of oxygen from the air to underground microorganisms.

### Greenhouse gas fluxes

The static chamber method was successfully used to observe gas emissions from the sludge treatment wetlands.

**Table 1** | TS(%) from samples taken during reed growth cycle from May to December, seasons correspond to sludge samples taken seasonally during drying treatment in beds

| Bed number | Sample depth | Spring | Summer | Autumn | Winter |
|---|---|---|---|---|---|
| Bed 1 | Surface (0–2 cm) | 14.8 | 20.2 | 21.2 | 22.6 |
| | Bottom (−6 ± 1 cm) | 17.5 | 24.0 | 23.6 | 24.6 |
| Bed 2 | Surface (0 cm) | 16.7 | 30.0 | 24.7 | 21.2 |
| | Bottom (−6 ± 1 cm) | 19.9 | 35.2 | 27.5 | 25.1 |
| Bed 3 | Surface (0 cm) | 15.3 | 29.2 | 24.9 | 22.5 |
| | Bottom (−6 ± 1 cm) | 18.5 | 34.4 | 28.6 | 25.7 |

**Table 2** | VS(%TS) from samples taken during reed growth cycle from May to December, seasons correspond to sludge samples taken seasonally during drying treatment in beds

| Bed number | Sample depth | Spring | Summer | Autumn | Winter |
|---|---|---|---|---|---|
| Bed 1 | Surface (0 cm) | 58.7 | 57.4 | 58.3 | 58.6 |
| | Bottom (−6 ± 1 cm) | 57.4 | 54.0 | 56.3 | 56.4 |
| Bed 2 | Surface (0 cm) | 56.7 | 52.6 | 54.7 | 55.3 |
| | Bottom (−6 ± 1 cm) | 54.5 | 50.6 | 51.5 | 52.4 |
| Bed 3 | Surface (0 cm) | 57.3 | 55.9 | 53.4 | 54.5 |
| | Bottom (−6 ± 1 cm) | 55.7 | 50.6 | 51.2 | 51.7 |

Figure 3(a) shows that $CO_2$ emissions to the atmosphere were higher in the USDB than STRBs during summer. The emission amounts varied from 28.68 to 100.42 g $CO_2$ $m^{-2}$ $d^{-1}$ in USDB and from 16.49 to 65.19 g $CO_2$ $m^{-2}$ $d^{-1}$ in STRBs. This represented a significant difference between beds ($p < 0.001$) and season ($p = 0.021$). The flux of $CO_2$ was greater compared to the natural wetland due to the steady supply of sludge including organic matter, nutrients, etc. for microbial reactions. Liikanen measured mean $CO_2$ release (respiration) doubled from 7,270 to 13,600 mg $CO_2$ $m^{-2}$ $d^{-1}$ in a 10-year period from a constructed wetlands (Liikanen *et al.* 2006a, b). The measured $CO_2$ flux in Liikanen's study was the total $CO_2$ release from aerobic and anaerobic decomposition processes, respiration of soil animals, and dark respiration of plants (shoots and roots). This flux measured in the dark does not include photosynthesis and is not a measure of net $CO_2$ exchange between the ecosystem and the atmosphere. However, in this study, the static chamber used was made of PLVC, which was totally transparent and allowed sunlight to come through its material. Therefore, the $CO_2$ flux measured included photosynthesis, and the reeds enclosed in the chamber were able to carry out photosynthesis; as a result, the $CO_2$ flux was higher in the bed without reeds than in the reeds bed. Especially during the summer, the $CO_2$ flux from the USDB was three times higher than that from STRBs. Thus, the measured $CO_2$ in this study included amount of $CO_2$ exchanged between the atmosphere, plants and sludge within the chamber. The significant difference between the USDB and STRBs demonstrated that the common reeds had strong capabilities in absorbing $CO_2$ for photosynthesis. To some extent, the reeds acted as a sink for GHG, and it assimilated carbon dioxide by photosynthesis, Picek calculated the carbon budget in a wastewater treatment constructed wetland, and the study indicated that the amount of carbon provided by plants represented one third of the total carbon input into the system (Picek *et al.* 2007).

In general, the $CO_2$ emissions were highest at the end of June, when the maximum values from the USDB and STRBs were 100.42 g $CO_2$ $m^{-2}$ $d^{-1}$ and 65.19 g $CO_2$ $m^{-2}$ $d^{-1}$, respectively. They gradually declined towards the winter to below 0.049 g $CO_2$ $m^{-2}$ $d^{-1}$. Brix found the highest rates of gas exchange through the plant component during hot and dry summer days, which related to the impact of sunlight (Grűnfeld & Brix 1999). A correlation of gas fluxes with surface soil temperature can be expected and indeed found in a natural wetland (Kim *et al.* 1998). Those trends were also shown in this study. This negative relationship between $CO_2$ and time of year was significant in all beds. The difference in $CO_2$ between the USDB and STRBs gradually became smaller, which was due to the different growth period of plants that slowed down or stopped growing in autumn and winter, which meant that the amount of $CO_2$ assimilated by photosynthesis decreased.

The $CH_4$ emissions (Figure 3(b)) are represented with mean values and error bars. In this figure, the variability between values from different beds and different times clearly shows the declining trend of $CH_4$ flux with time. The $CH_4$ emission amount varied from 0.26 to 0.99 g $CH_4$ $m^{-2}$ $d^{-1}$ in the USDB, and from 0.43 to 1.95 g $CH_4$ $m^{-2}$ $d^{-1}$ within STRBs. The $CH_4$ emission rates also differed significantly between beds ($p = 0.015$) and seasons ($p = 0.023$). Indeed, $CH_4$ emissions have been demonstrated to be significantly higher in summer than during winter. Compared to $CO_2$ flux, $CH_4$ emission was higher in STRBs, reaching 1.95 g $CH_4$ $m^{-2}$ $d^{-1}$, especially in summer, when emission values from the STRBs were two times greater than the USDB. However, those differences had decreased towards the end of the vegetation season. The results partly proved the reeds' contribution to methane emissions compared with the USDB; in Grűnfeld and Brix's study on the effect of the presence of *Phragmites australis* on methane emission, they observed a reduction of 62% in the emission of

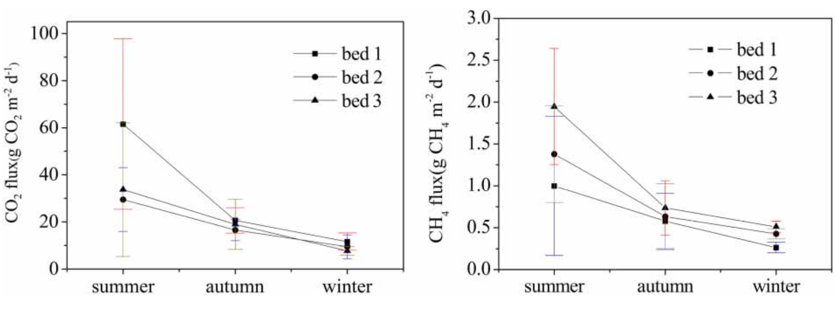


**Figure 3** | Rates of $CO_2$ (a) and $CH_4$ (b) emissions through the bed surfaces during the 2013 vegetation season.

methane after cutting the plants (Grűnfeld & Brix 1999). Figure 2 also shows that emissions from Bed 2 were greater than those from Bed 3. These results partly contradict our expectation that the aeration pipes in Bed 2 should improve gas exchange between the sludge residue and air, particularly an increased oxygen concentration in the sludge residue, which could provide an relatively aerobic environment for the decomposition of organic matter, therefore reducing $CH_4$ emissions from anaerobic microbial reactions. Bed 2 emitted more $CH_4$ than Bed 3 due to the different growth conditions of reeds where Bed 2 had more plant shoots or plant biomass compared with Bed 3. As a result, it may take more time for sludge residue to crack due to the decreased effects of the sun and wind since the dense vegetation may reduce the amount of sunshine reaching on the sludge residue surface and may restrict plant stem movement caused by wind. All these lead to the prevention of gas exchange between atmosphere and sludge residue. In addition, the STRBs were operated for only 2 years, so the stored sludge was not thick enough for the reeds to root. Therefore, the amount of oxygen released by the roots was relatively small in the sludge residue. Part of the oxygen is released through the pores where oxygen escapes from the plant into the root zone, meaning that methane can enter the plant aerenchyma system and subsequently be emitted into the atmosphere (Laanbroek 2010). In addition to the leading role of reeds in the emission of methane by their aerenchyma system, they also contribute to the production of methane by the production of organic carbon. The highest methane efflux rates relative to the standing plant biomass has been observed, because of the plants' carbon fixation. But these results did not contradict the better performance of dewatering and organic matter decomposition, because more plants mean more carbon fixation and more carbon production; as a result, there were more bacteria in the sludge and more microbial activities. Grűnfeld & Brix (1999) also found that when reed is not present, part of the produced methane is held back in the substrate, and when the STRBs are eventually emptied, the trapped methane will be released. Furthermore, the short resting period in this experiment is very likely to have affected the emissions of $CH_4$ from the beds, making them larger than those in other research (Grűnfeld & Brix 1999).

## Gas fluxes before and after sludge loading

Figure 4 shows that the $CO_2$ and $CH_4$ fluxes decreased just after sludge loading in the USDB (Bed 1), whereas the $CO_2$ flux was low before the feeding event and increased rapidly after sludge loading within the reed beds. During the following days, the emissions from the USDB increased, whereas the $CO_2$ flux within the STRBs decreased. The results indicated that there were two different emission mechanisms within the system. For the USDB, the declining trend of emission from before to after feeding indicates that the new, wet sludge layer initially functioned as a lid that prevented gas exchange (Nielsen 2005) between the underlying sludge and the atmosphere. The increased fluxes between Day 2 and Day 6 after loading demonstrate that by that time the new sludge layer had been partly cracked due to the combinational effect of sunshine and wind, which allowed the release of gases from the underlying layer.

The increased $CO_2$ and $CH_4$ fluxes from before to after sludge loading were also reported in another study (Uggetti et al. 2012). Researchers observed that the increase of the $CH_4$ fluxes after sludge loading was an effect of the new load change, the $O_2$ conditions and the organic matter

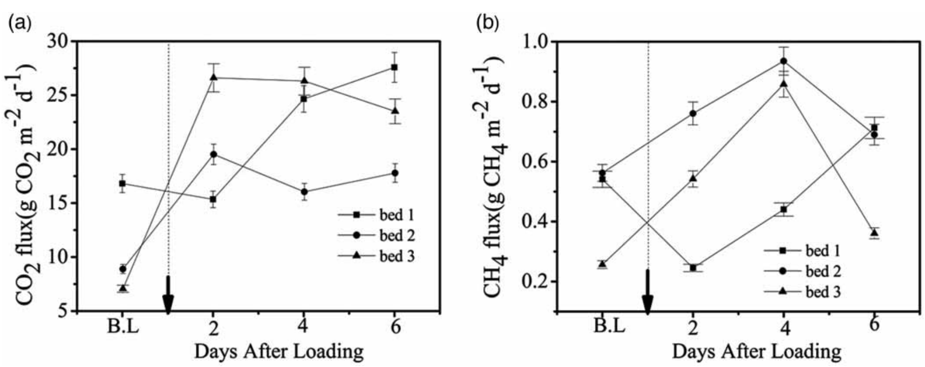


**Figure 4** | Fluxes of $CO_2$ (a) and $CH_4$ (b) before loading (B.L.) of sludge and 2, 4 and 6 days after loading in the STRBs. The black arrows indicate the time of sludge loading. B.L. represents pooled data from three sampling events before loading.

from the sludge feeding for aerobic and anaerobic microbial decomposition.

### Global warming potential

Wetlands are the main source of global $CH_4$ emissions, and natural and cultivated wetlands represent nearly 40% of the sources of atmospheric methane (IPCC 1995). For $CH_4$, there is a GWP of 25 times $CO_2$-equivalents on a 100-year time scale. $CO_2$ emissions connected to wastewater and sludge treatment processes are considered to be originally fixed from the atmosphere and therefore climate neutral (IPCC 2006). However, $CH_4$ emissions have been demonstrated to be considerably higher during summer than in winter. By calculating the mean value emission of $CH_4$ seasonally, then summing them up, the GWP of methane emission results in 13.13 and 15.02 g $CO_2$eq $m^{-2}$ $d^{-1}$ from unplanted and planted beds, respectively. The GWP of methane in this study was lower than Uggetti (Uggetti *et al.* 2012) recorded in the sludge treatment wetlands. Compared with other conventional methods of sludge treatment, Uggetti found that the total GWP of centrifugation had twice as high GWP from sludge treatment wetlands due to high energy consumption, whereas untreated sludge transported had almost nine times higher GWP. Furthermore, it has been shown that the environment impact of STRB is 500 times lower than conventional sludge treatments like centrifugation and 2,000 times lower than sludge transport. Therefore, the implementation of sludge treatment wetlands in small and remote communities would reduce GHG from sludge management. The study showed the benefits of treatment wetlands as an environmentally friendly technology.

## CONCLUSION

The most important difference between the sludge treatment beds is planting with reeds, and this factor was especially significant in summer when the reeds thrived. The result of this is that the dewatering process is relatively poor in the USDB compared with STRBs, which is shown in the TS of the sludge residue. In addition, the organic residues in the STRBs were more decomposed than in sludge drying beds, which is demonstrated by the VS of the sludge samples.

The transparent static chamber allowed solar radiation to enter and enabled the plant to conduct photosynthesis within the closed area. The $CO_2$ flux of the USDB was two times higher than that of STRBs, which indicated the plants' high-capability for absorbing $CO_2$.

Emissions of $CO_2$ and $CH_4$ are strongly changed by fresh sludge loading, which enhances emissions from vegetated beds. In contrast, emissions of both $CO_2$ and $CH_4$ decreased after loading sludge into the unplanted bed; this was probably due to the new sludge layer preventing gas exchange between the deeper layer and the atmosphere. In subsequent days, the sludge became cracked due to the effect of sunlight and wind, and gas emissions increased a few days later after loading.

The GWP of methane emissions were 13.3 and 15.02 g $CO_2$eq $m^{-2}$ $d^{-1}$ from USDB and STRBs, respectively, which is much lower than values for traditional sludge centrifugation and transport.

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