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Peat capping: Natural capping of wet landfills by peat formation



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ABSTRACT

Given the bioremediation potential of peat, natural capping of landfills in wetlands with a “peat cap” could provide a sustainable addition to regular capping methods using basal liners with limited life-spans and sand. It is unknown, however, which initial conditions optimise growth of this “peat cap” on top of a sand layer. Here, we tested the combined effects of topsoil addition (clay or organic soil) and vegetation type (*Typha latifolia*, *T. angustifolia*, *Stratiotes aloides* and submerged spp.) on net ecosystem C exchange and water quality in 18 sandy basins situated in a constructed wetland on top of a landfill. Although the highest net C sequestration rates occurred in *Typha* stands on sand, due to lower decomposition-related C losses as compared to clay and organic topsoils, vegetation development was slow and its cover was very low (15%) compared to clay (40%) and organic topsoils (70%). As this strongly impeded the build-up of a uniform peat layer, we conclude that, within a restricted time frame, the application of nutrient-rich topsoils is still necessary for sufficient biomass production to accumulate organic material. By recycling local soils, the accompanying initial C loss becomes negligible on a landscape scale.

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1. Introduction

Wetlands near urban or industrial areas, including drained and excavated peatlands, have been extensively used as landfills throughout the world, e.g. in Germany (Vielhaber and Weiss, 2014), the Netherlands (Buijs et al., 2005; Heida, 1986), Finland (Assmuth and Strandberg, 1993), Canada (Wreford et al., 2000) and the United States (Ewing, 2002). After closure, landfills are usually capped with a technical barrier, consisting of a basal liner (often made of high-density polyethylene; HDPE) and clean sand, and subsequently transformed into dry grasslands with a source-monitoring programme (Ewing 2002; Simon and Muller, 2004). Meanwhile the creation of new wetlands generally remains limited to wetlands

constructed for treatment of landfill leachate or municipal wastewater (Bulc 2006; Vymazal and Kropfelova, 2009). Given the high potential of peat to sequester contaminants such as heavy metals in a similar way to activated carbon (Mclellan and Rock, 1988) and to stimulate the bioremediation of organic pollutants (Couillard, 1994), the construction of new, peat-forming wetlands on top of former landfills could provide a sustainable addition to regular capping methods, which have a limited durability (Allen 2001; Rowe and Sangam, 2002). The realisation of such a “peat cap” would not only serve as an efficient capping method for landfills, but may also provide additional services including carbon (C) sequestration, water retention, recreation and biodiversity (De Klein and Van der Werf, 2014; Knight, 1997; Wild et al., 2001).

To effectively utilise these novel wetlands as natural caps, optimal conditions for C sequestration have to be created. Maximising peat build-up and C sequestration requires simultaneously high biomass production and low decomposition rates, both of which are strongly linked to a suite of biogeochemical processes (Aerts and Toet, 1997; Lamers et al., 2015). It is therefore essential to define the optimal water and soil quality for wetland construction and stimulate growth of peat-forming vegetation. While development of fast growing, rooting wetland macrophytes may be limited on soils low

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in organic matter, addition of richer soil may stimulate vegetation but also raises risks of eutrophication and enhanced decomposition and methane (CH₄) emission. It has yet to be tested whether the benefits of the addition of top layers of clay or organic soil, in terms of increased biomass production, outweigh the negative side effects of additional nutrient mobilisation and C emission.

Highly productive target species, including *Typha latifolia* and *Phragmites australis*, are often applied in constructed wetlands for the extraction of nutrients from wastewater because of their high net primary production (NPP) and nutrient uptake rates (Tanner, 1996; Wild et al., 2001). The high NPP rates (Brinson et al., 1981; Brix et al., 2001) and relatively low decomposition rates of –mainly belowground– biomass (Alvarez and Becares, 2006; Hartmann, 1999) that characterise these species may result in a considerable contribution to the C sequestration function of the newly constructed wetland. On the other hand, these helophyte species are known to function as a chimney for methane (CH₄) emission (Askær et al., 2011) and their radial oxygen (ROL) loss may promote breakdown of organic matter in deeper – otherwise anoxic – sediment layers (Bernal et al., 2016). Next to helophytes, floating aquatic macrophytes, such as *Stratiotes aloides*, may also reach high NPP rates in peatland systems. Through their vigorous vegetative growth (Cook and Urmi-König, 1983; Smolders et al., 1995), *S. aloides* forms dense mats that may cover entire water bodies in only a few growing seasons (Cook and Urmi-König, 1983). As a result, this species can reduce the area of open water faster than helophytes, which mainly colonise from the shores (Van Geest et al., 2003). Furthermore, floating macrophytes facilitate high biodiversity by serving as foundation species (Rantala et al., 2004; Sugier et al., 2010; Van der Winden et al., 2004). They may thus not only help to stimulate peat accumulation and C sequestration, but also promote biodiversity, thereby enhancing the attractiveness of the wetland as a recreational and educational area.

Given the limited lifespan of basal liners used to cover landfills (Allen, 2001; Rowe and Sangam, 2002), a sustainable “peat cap” could be created on top of traditional capping methods to take over the capping function when basal liners degrade. Furthermore, while traditional capping methods generally only aim at immobilising contaminants, the peat layer could add to the bioremediation of (organic) pollutants. Since peatland formation is a novel approach to cap former landfills, however, it still has to be determined which initial conditions, in terms of soil and vegetation type, result in high C sequestration rates and the build-up of a peat layer. The main question that needs to be answered is therefore: How to obtain high biomass production and low decomposition rates simultaneously, in order to have a fast transformation from sand to growing peat?

A newly constructed wetland (2011) on top of a former landfill close to Amsterdam (The Netherlands) provided the optimal test location to find an answer to this question. At this location, a 4 ha research area was provided to test the effect of soil type and vegetation on C sequestration potential for peat formation (through net ecosystem exchange (NEE) of C) and surface water quality. Our experimental set-up comprised 18 rainwater-fed sand basins lined with HDPE basal liner at 50–80 cm depth, of which 6 received an additional layer of organic soil and another 6 clay soil. Vegetation was allowed to develop spontaneously over the course of 3 years, resulting in 4 dominant vegetation types: *Typha latifolia*, *T. angustifolia*, *Stratiotes aloides* (the only species that required active introduction as it mainly reproduces clonally (Smolders et al., 1995) and submerged species. We analysed vegetation development, CO₂ sequestration rates and CH₄ fluxes of all combinations of vegetation type and soil. Based on the outcome of this study, we discuss the optimisation of the construction as well as the possibility of recycling local clay or organic soils to stimulate vegetation development.

Table 1;

Soil characteristics of sand, organic soil and clay used in the construction of the Volgermeerpolder (mean ± SEM, n = 6). Different superscript letters indicate statistically significant differences between soils (P < 0.05). Abbreviations FW and DW refer to fresh weight and dry weight, respectively.

		Sand	Clay	Organic
Water content	%	19.3 ± 0.7 ^a	41.2 ± 1.7 ^b	64.2 ± 0.8 ^c
Bulk density	kg DW L ⁻¹ FW	2.01 ± 0.03 ^c	1.62 ± 0.03 ^b	1.35 ± 0.02 ^a
Organic matter	%	0.5 ± 0.2 ^a	5.0 ± 0.5 ^b	21.5 ± 0.8 ^c
NO ₃ ⁻	μmol L ⁻¹ FW	13.0 ± 5.2	2.0 ± 1.4	22.6 ± 15.2
NH ₄ ⁺	μmol L ⁻¹ FW	91 ± 23 ^a	353 ± 71 ^b	251 ± 57 ^{ab}
Olsen-P	μmol L ⁻¹ FW	354 ± 39 ^a	792 ± 107 ^b	475 ± 20 ^a
C	%	0.7 ± 0.1 ^a	2.7 ± 0.2 ^b	9.3 ± 0.4 ^c
N	%	0.04 ± 0.00 ^a	0.16 ± 0.01 ^b	0.46 ± 0.02 ^c

2. Materials and methods

2.1. Site description

In 2011, the first phase of the construction of a “natural cap” was completed on top of the former landfill Volgermeerpolder (52°25'21.03" N, 4°59'32.23" E) in the Netherlands, which was used as a waste dump between 1927 and 1981 (Buijs et al., 2005). After sealing off the contaminated landfill with sheets of high-density polyethylene (HDPE, 2 mm; life expectancy 80–100 years) and a 50 cm layer of clean sand, a new wetland of 100 ha was created on top, consisting of several basins enclosed by clay levees (designed by Vista Landscape Architecture and Urban Planning, Amsterdam, the Netherlands and developed by ACV (Egbring, 2011)); See Fig. S1 for an aerial photograph of the research area). In 18 of these basins, ranging in size from 550 to 1600 m², we compared basins without (controls; n = 6) or with the application of an additional layer of 30 cm of clay (n = 6; originating from a freshwater wetland in the north-western part of the Netherlands; 52°40'15" N, 5°7'2" E) or organic soil (n = 6; originating from a nearby peatland area; 52°17'13" N, 4°46'12" E) on top of the clean layer of sand. Soil characteristics are displayed in Table 1. Basins were naturally filled with rainwater and a water depth of 40–80 cm (minimum depth of 20 cm during droughts) was maintained by pumping in water from large rainwater storage basins (with characteristics similar to sand soils, see Table 1) during dry periods in spring and summer.

Vegetation in all basins developed naturally, with the exception of *Stratiotes aloides*, which was introduced in enclosures in the summer of 2011 (50 plants per basin, originating from a nearby pond; 52°23'10.1" N, 4°56'55.0" E) in the north-eastern corner of all basins (most suitable given the predominant south-western winds) in summer 2011. Natural development resulted in dominant vegetation of helophytes (*Typha latifolia*, *T. angustifolia*, *Phragmites australis*, *Glyceria maxima*) and submerged vegetation (*Potamogeton pusillus*, *P. pectinatus*, *Myriophyllum spicatum*, *Characeae*, *Elodea nuttallii*). In September 2013, we allocated one plot of 1 m² per basin in representative patches (when present) of *Typha latifolia* (n = 16), *T. angustifolia* (n = 8), *S. aloides* (n = 15) and bare soil (n = 17), totalling 56 plots. *Phragmites australis* and *Glyceria maxima* were excluded, as they were only present in two basins. While the patches of bare soil were unvegetated in September 2013, submerged plant species colonised the plots in most basins, starting in spring 2014 and eventually reached 169 ± 66 g DW m⁻² in August 2014. Total plant cover of all dominant vegetation types was determined at the peak of the growing season, in August 2014.

2.2. Soil and water analyses

Surface water and pore water samples were taken in November 2013 and February, May and July 2014. Pore water was collected by attaching vacuum syringes to ceramic soil moisture cups

Table 2
Chemical characteristics of the water layer and pore water (mean \pm SEM, n=6) of basins with sandy soils, or with sand with added clay or organic topsoil. Values represent averages between November 2013 and September 2014. All concentrations are given in $\mu\text{mol L}^{-1}$, whereas alkalinity is presented in meq L^{-1} . Different superscript letters indicate significant differences between soil types ($P < 0.05$).

	Water layer			Pore water		
	Organic	Clay	Sand	Organic	Clay	Sand
pH	7.8 \pm 0.2	7.8 \pm 0.1	7.8 \pm 0.1	7.0 \pm 0.1	7.1 \pm 0.0	6.9 \pm 0.1
Alkalinity	3.8 \pm 0.2 ^b	3.7 \pm 0.5 ^b	2.4 \pm 0.2 ^a	19.1 \pm 2.8 ^b	19.0 \pm 2.5 ^b	8.4 \pm 0.9 ^a
CO ₂	244.5 \pm 68.6	262.8 \pm 97.8	112.0 \pm 17.0	3669 \pm 240 ^b	2974 \pm 295 ^{a,b}	2373 \pm 425 ^a
HCO ₃ ⁻	3322 \pm 208 ^b	3162 \pm 589 ^{a,b}	1975 \pm 96 ^a	14303 \pm 1659 ^b	14659 \pm 1613 ^b	7757 \pm 880 ^a
NO ₃ ⁻	0.40 \pm 0.11	0.82 \pm 0.28	0.47 \pm 0.08	4.51 \pm 0.74	4.51 \pm 1.16	4.56 \pm 0.41
NH ₄ ⁺	2.91 \pm 0.46	6.27 \pm 2.41	2.13 \pm 0.09	34.19 \pm 17.51	20.93 \pm 8.58	49.46 \pm 12.82
o-PO ₄ ³⁻	2.07 \pm 0.64 ^b	1.29 \pm 0.63 ^{a,b}	0.27 \pm 0.05 ^a	13.31 \pm 1.63 ^b	1.98 \pm 0.48 ^a	4.36 \pm 0.86 ^a
Total-P	5.07 \pm 1.62 ^b	4.27 \pm 1.60 ^{a,b}	1.12 \pm 0.19 ^a	189.7 \pm 39.3 ^b	43.5 \pm 10.3 ^a	67.9 \pm 5.7 ^a
Ca	2480 \pm 269	2179 \pm 57	2508 \pm 112	7603 \pm 989 ^b	7544 \pm 991 ^b	3813 \pm 351 ^a
Fe	1.70 \pm 0.11	14.20 \pm 8.84	4.48 \pm 0.46	204.6 \pm 49.4	316.6 \pm 103.9	184.0 \pm 19.3
Mg	655 \pm 84	602 \pm 25	730 \pm 43	1961 \pm 286 ^b	1528 \pm 196 ^{a,b}	1046 \pm 92 ^a
S	1680 \pm 472	1379 \pm 298	2462 \pm 321	988 \pm 567	1888 \pm 700	963 \pm 311

(Eijkelkamp, Giesbeek, The Netherlands), fixed in the soil at 15 cm depth. pH was measured using a standard Ag/AgCl electrode (Orion, Thermo Fisher Scientific, Waltham, MA, USA) combined with a pH meter (Tim840 titration manager; Radiometer Analytical, Lyon, France), whereas alkalinity was determined by titrating down to pH 4.2 with 0.1 N HCl using an auto-burette (ABU901, Radiometer, Copenhagen, Denmark). Total inorganic carbon (TIC) was measured by injecting 0.1 mL of sample into an N₂ flushed chamber filled with 1 mL phosphoric acid (0.4 M), connected to an infrared gas analyser (IRGA; ABB Analytical, Frankfurt, Germany), whereas total organic carbon (TOC) was measured on a TOC-L CPH/CPN analyser (Shimadzu, Kyoto, Japan). Concentrations of phosphate (PO₄³⁻), ammonium (NH₄⁺) and nitrate (NO₃⁻) were measured colourimetrically on an Auto Analyser 3 System (Bran & Luebbe, Norderstedt, Germany) using ammonium molybdate (Henriksen, 1965), hydrazine sulphate (Kamphake et al., 1967) and salicylate (Grasshof and Johannse, 1972), respectively. Concentrations of calcium (Ca), iron (Fe), magnesium (Mg), Total-phosphorus (P) and sulphur (S) were measured by inductively coupled plasma spectrometry (ICP-OES iCAP 6000; Thermo Fisher Scientific, Waltham, MA, U.S.A.). Data were combined to calculate yearly averages of surface water and pore water pH, alkalinity, TIC, TOC and nutrient concentrations.

In February 2014, soil samples were collected in airtight bags and kept at 4 °C until analyses. Per basin, 5 subsamples of the top 10 cm were pooled for analyses. Samples of known volume were weighed and dried for 48 h at 70 °C to determine dry weight (DW) and bulk density. Organic matter content was determined through loss on ignition (3 h 550 °C). Plant available P was extracted according to Olsen et al. (1954), whereas a salt extraction was performed as described in Harpenslager et al. (2015a) and Harpenslager et al. (2015b). Extracts were analysed for PO₄³⁻, NH₄⁺ and NO₃⁻ as described above. Total concentrations of Ca, Fe, Mg, Total-P and S were analysed as by ICP spectrometry as described above. Carbon (C) and nitrogen (N) contents of the soil were determined using an elemental analyser (Carlo Erba NA1500, Thermo Fisher Scientific, Waltham, MA, U.S.A.).

2.3. C flux measurements

Diffusive fluxes of CO₂ and CH₄ were measured in November (2013), January, April, May, July, August and September (2014) by placing a gastight transparent closed chamber over the 56 plots. For helophyte vegetation, a large closed chamber (ϕ 50 cm, 140 cm, 270 L) was pressed into the soil with three thin pins to fix the chamber but prevent large disturbance to the soil. A smaller, floating closed chamber (ϕ 30 cm; 20 cm; 14.5 L) was used in plots with submerged, floating or no vegetation. Chambers were placed on

the plots 5 min before the start of the measurement and the internal air was subsequently analysed during 6 min for linear increase or decrease in CO₂ and CH₄ concentrations using cavity ring-down spectrometry by connecting the chambers in a closed loop to a greenhouse gas analyser (GGA-LGR, Los Gatos Research, CA, U.S.A.). C-fluxes were measured under both light and dark conditions. For dark measurements, plots were dark-adapted by covering the closed chambers with opaque fabric before measurements. Hourly data on daytime temperature and radiation were collected from a nearby weather station (KNMI, Schiphol; 52° 18' 03.6" N, 4° 46' 26.4" E) and are presented in Fig. S2.

Net ecosystem exchange of CO₂ (NEE_{CO2}; g C m⁻² d⁻¹) comprised the net CO₂ flux from plots vegetated by *T. latifolia*, *T. angustifolia* and *S. aloides* including soil and water respiration (Eq. (1)). By integrating the separate NEE_{CO2} values, the annual CO₂ sequestration of these three species was calculated. Combined light and dark fluxes of CO₂ and CH₄ from bare plots (which included some submerged vegetation during summer) and of plots vegetated by *T. latifolia*, *T. angustifolia* and *S. aloides* were used to calculate the net ecosystem exchange of carbon (NEE_C; g C m⁻² d⁻¹; Eq. (2)). As with NEE_{CO2}, these NEE_C values were extrapolated to annual values by integration of separate measurements. In all analyses, plots that were measured less than 4 times during the year were excluded (n=2). Missing data were estimated using the average value of replicates when available (n=7). Otherwise, consecutive data points were used to cover for the missing values (n=5).

The following equations were used, with negative fluxes indicating C-sequestration and positive fluxes indicating loss to the atmosphere:

$$\text{NEE}_{\text{CO}_2} = (h_{\text{light}} \times \text{CO}_2 - C_{\text{light}}) + (h_{\text{dark}} \times \text{CO}_2 - C_{\text{dark}}) \quad (1)$$

$$\begin{aligned} \text{NEE}_C = & (h_{\text{light}} \times (\text{CO}_2 - C_{\text{light}} + \text{CH}_4 - C_{\text{light}})) \\ & + (h_{\text{dark}} \times (\text{CO}_2 - C_{\text{dark}} + \text{CH}_4 - C_{\text{dark}})) \end{aligned} \quad (2)$$

with h_{light} and h_{dark} being the hours of light and dark per day during that time of year, $\text{CO}_2 - C_{\text{light}}$ and $\text{CH}_4 - C_{\text{light}}$ the fluxes of CO₂ and CH₄ measured under light conditions and $\text{CO}_2 - C_{\text{dark}}$ and $\text{CH}_4 - C_{\text{dark}}$ the fluxes of CO₂ and CH₄ measured under dark conditions. All fluxes are expressed as g C m⁻² d⁻¹.

2.4. Statistical analyses

Data on soil characteristics and annual NEE rates were tested for normality of residuals and homogeneity of variance using Shapiro Wilk's test and Levene's test, respectively. Non-normal or heteroscedastic data were log transformed to authorise use of

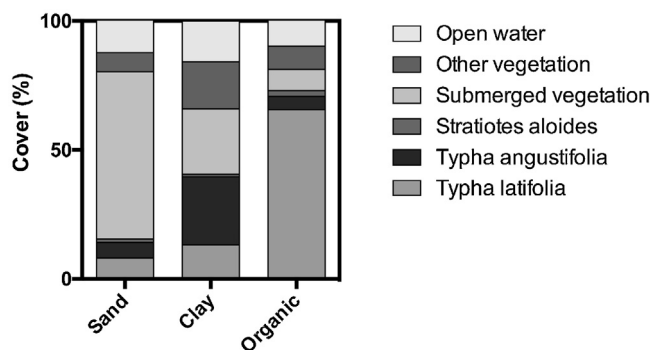


Fig. 1. Vegetation cover (%) of *Typha latifolia*, *T. angustifolia*, *Stratiotes aloides*, other emergent vegetation, submerged vegetation and bare soil after three years of development on sand soils, or soils with an additional clay or organic topsoil.

parametric tests. Data were analysed using two-way ANOVAs with “Soil” and “Plant species” as fixed variables, followed by Tukey post-hoc tests. Monthly NEE rates were tested for normality of residuals using Levene’s test, after which they were analysed using repeated measures followed by Bonferroni post-hoc tests. All analyses were carried out in SPSS (V21, 2012, IBM Statistics).

3. Results

3.1. Nutrient characteristics

Although all soils were characterised by high pH and alkalinity, the applied top layers of clay and organic soil had an even higher alkalinity than the original sand soil ($P=0.017$; $F=5.479$), due to a higher HCO_3^- availability ($P=0.007$; $F=7.285$). Ca ($P=0.009$; $F=6.792$) and, in organic topsoils, Mg ($P=0.025$; $F=4.841$) also showed higher concentrations than in pure sandy soils (Table 2). Mobilisation of HCO_3^- to the water layer caused a high surface water pH and alkalinity (Table 2), which was again higher for basins with an organic or clay topsoil ($P=0.016$; $F=5.519$). Concentrations of NO_3^- and NH_4^+ were low in both the water layer and the pore water (Table 2), and did not differ between soil types. Soils with organic topsoils had a higher availability of PO_4^{3-} in the pore water than soils consisting of sand or sand with clay topsoils ($P<0.001$; $F=27.147$; Table 2). Pore water Fe: PO_4^{3-} ratios were high for all soils, but lower ($P=0.004$; $F=8.351$) for sand and added organic topsoils (16.9 ± 4.4 , and 57.5 ± 16.9 , respectively) than for clay topsoils (217.2 ± 79.9). Surface water concentrations of PO_4^{3-} ($P=0.028$; $F=4.597$) and Total-P ($P=0.013$; $F=5.877$) were higher for added organic soils than sand, but PO_4^{3-} concentrations remained below $3 \mu\text{mol L}^{-1}$ in all treatments throughout the year (Table 2).

3.2. Vegetation development

Addition of clay or organic soil stimulated cover by helophytes (Fig. 1; $P<0.001$; $F=17.184$). While control basins showed a helophyte cover of only $14.1 \pm 2.5\%$ after 3 years, the addition of clay and organic topsoils resulted in an increase to $39.5 \pm 9.6\%$ and even $70.6 \pm 6.4\%$, respectively. Furthermore, while basins with sand soils only showed colonisation by helophytes from the shore, development of the helophyte vegetation in basins with an applied layer of clay or organic soils occurred both from the shores and by germination or vegetative growth throughout the basin. In organic soils, this led to an explosive development of *Typha latifolia*, which eventually covered around 65% of the basins. On clay soils, *Typha angustifolia* was the dominant species, covering around 30% of the basins (Fig. 1). Sand soils showed the lowest cover by emergent species, with $77.3 \pm 2.7\%$ of the basin remaining unvegetated or covered by

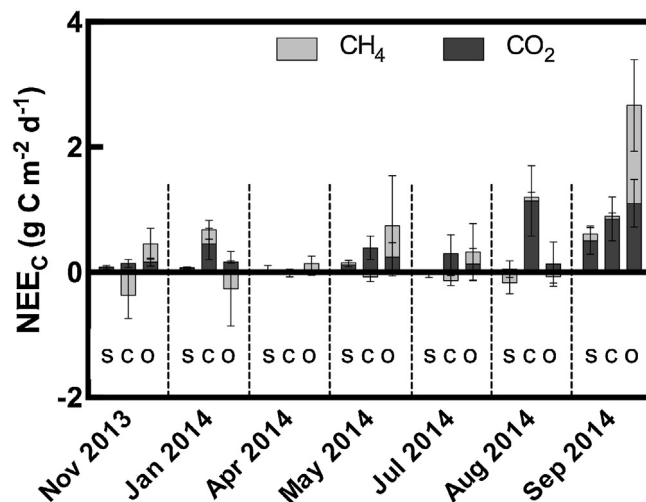


Fig. 2. NEE_c rates of plots without emergent vegetation on sand soils (S), or sand with clay (C) or organic (O) topsoils between November 2013 and September 2014 (Mean \pm SEM), separated into CO₂ and CH₄ fluxes. Rates result from processes in soil, water and submerged vegetation (occurring on these plots between spring and autumn 2014). Note that negative values indicate C sequestration, whereas positive values represent a loss of C to the atmosphere.

submerged species (Fig. 1). Other species occurring were *Eleocharis palustris*, *Bolboschoenus maritimus*, *Alisma plantago-aquatica* and *Phragmites australis*, none of which covered more than 10% of a basin. Only in two basins with added clay soil, a significant area (15 and 60%) was covered by *Glyceria maxima*. Introduced *Stratiotes aloides* persisted in their enclosures in most basins during subsequent years, but only migrate outside these enclosures in a few basins, which is why this species showed the lowest coverage (less than 5%) of the three measured species (Fig. 1).

3.3. Net ecosystem exchange of bare plots

Combined net ecosystem exchange (NEE_c) rates of CO₂ and CH₄ of bare soil, overlying water and submerged vegetation (present between spring and autumn) were determined throughout the year. Overall, values ranged from $+0.07$ to $+1.11 \text{ g C m}^{-2} \text{d}^{-1}$ (loss to the atmosphere) for most of the year. During autumn, emission of CO₂ and CH₄ increased to rates of up to $+4.2$ and $+3.8 \text{ g C m}^{-2} \text{d}^{-1}$ respectively (Fig. 2). CO₂ respiration rates did not differ among the different soil types, whereas the relative contribution of CH₄ to the NEE_c was higher for organic soils than for sand soils ($P=0.021$; $F=5.192$). None of the bare plots showed net CO₂ fixation (negative NEE_{CO₂}), indicating that the contribution of submerged species to the net C sequestration was negligible (Fig. 2). Net ecosystem exchange of vegetation

The CO₂ exchange rates of vegetated plots (NEE_{CO₂}) closely followed seasonal variation in temperature and light availability (Fig. S2), with the highest CO₂ fixation rates (indicated by negative NEE_{CO₂} rates) occurring in July and net emissions (positive NEE_{CO₂} flux) of CO₂ in autumn and winter (Fig. 3). Both *Typha* species had a significantly higher negative NEE_{CO₂} than *S. aloides* ($P=0.003$; $F=7.719$; Figs. 3 and 4A). There appeared to be differences, however, among annual NEE_{CO₂} rates from plots with *Typha* spp. for the three different soil types (Fig. 4A), with 4 times higher net fixation on sand than on plots with added organic topsoil (-590 vs. $-140 \text{ g C m}^{-2} \text{y}^{-1}$; $P=0.058$). The lower average *Typha* NEE_{CO₂} rates on organic topsoils resulted from high variation, due to two basins showing net effluxes of C, while the other four showed net fixation rates. When including CH₄ fluxes (NEE_c), 4 out of 6 plots of *Typha* on organic topsoils became net C sources, resulting in an average NEE_c of $+300 \text{ g C m}^{-2} \text{y}^{-1}$, whereas negative NEE_c rates of *Typha* vege-

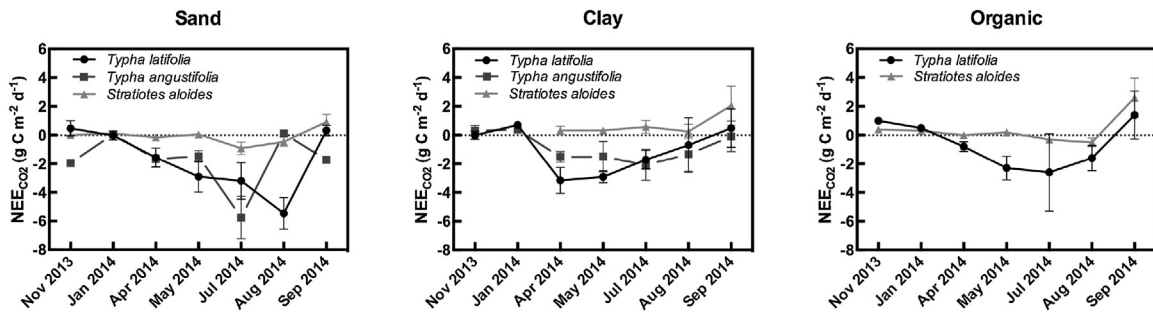


Fig. 3. Net ecosystem exchange of CO₂ of *Typha latifolia*, *T. angustifolia* and *Stratiotes aloides* on sand and sand with clay or organic topsoils between November 2013 and September 2014 (Mean ± SEM). Note that negative values indicate C sequestration, whereas positive values represent a loss of C to the atmosphere.

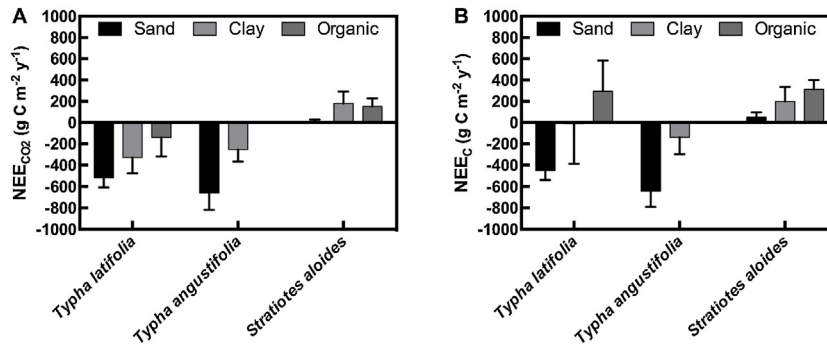


Fig. 4. Cumulative yearly net ecosystem exchange (NEE) rates of CO₂ (left panel) and of CO₂ and CH₄ (right panel) from plots vegetated by *T. latifolia*, *T. angustifolia* (absent on organic topsoils) or *S. aloides* (Mean ± SEM). Note that negative values indicate C sequestration, whereas positive values represent a loss of C to the atmosphere.

tated clay and sand soils indicated that these remained net sinks, with NEE_C rates of −10 to −100 and −450 to −600 g C m^{−2} y^{−1}, respectively (Fig. 4B).

4. Discussion

For the development of a constructed wetland designed for land-fill peat capping, fast accumulation of recalcitrant organic matter by highly productive vegetation is very important to ensure sediment accretion by the formation of a growing peat layer on top. By varying soil and water characteristics as well as emergent, submerged or floating macrophyte species, our large-scale field experiment showed that it is possible to gain a high coverage of highly productive macrophytes within a time-span of 3 years. There is, however, a trade-off between confined high CO₂ sequestration rates (up to 590 g C m^{−2} y^{−1}) on the one hand and high colonisation rates (up to 80%) on the other.

4.1. CO₂ fixation by vegetation

In our field experiment, the *Typha* species *T. latifolia* and *T. angustifolia*, showed the highest CO₂ sequestration potential. These species showed NEE_{CO₂} rates ranging from −140 to −590 g C m^{−2} y^{−1}, which are comparable to, or even higher than, those reported for *Typha*-dominated marshes (−100 to −270 g C m^{−2} y^{−1}) (Bernal and Mitsch, 2012; Reddy et al., 1993) and those reported for temperate, minerotrophic fens (−208 to +190 g C m^{−2} y^{−1}) (Bubier et al., 2003; Koch et al., 2008; Martikainen et al., 1995; Saarnio et al., 2007). Both submerged vegetation and *S. aloides* had a negligible contribution to the C sequestration potential of the system. While submerged vegetation showed relatively high production rates, these species are generally also characterised by high decomposition rates (Kirschner and Velimirov, 2001; Wetzel, 1983), which results in a low contribution to the total C budget. The poor development of the population of *S. aloides*, on

the other hand, can be explained by the high alkalinity and low CO₂ availability in the basins, which lowers underwater photosynthesis and thus prevents vegetative reproduction (Harpenslager et al., 2015a).

We found that when basins were constructed with sand, only 15% became covered by helophytes over the course of 3 years, while the remaining 85% consisted of submerged species or bare soil. The next question therefore is how to speed up the colonisation by highly productive species in newly constructed wetlands and thereby maximise net biomass accumulation. The application of a layer of clay or organic soil strongly increased the cover by helophyte species, resulting in an average cover of 40% and 70% after 3 years, respectively. This strongly enhanced colonisation rate most likely resulted from the higher nutrient availability in the applied soils, including higher P availability in both pore water and surface water of, especially, organic topsoils. The higher nutrient availability in the clay and organic topsoils enabled colonisation by *Typha* spp. throughout the basins, instead of being restricted to the shores, as was the case in basins with only sandy soils. In these basins, the shores most likely had a higher nutrient availability than the rest of the basin, due to run-off from the surrounding clay levees.

4.2. Carbon balance

Increasing organic matter content of the applied soil led to decreases in the net CO₂ fixation rates of soils vegetated by *Typha*. On sand soils, *Typha* spp. reached NEE_{CO₂} rates of approximately −590 g C m^{−2} y^{−1}, whereas plots of *Typha* growing on clay and organic topsoils had NEE_{CO₂} rates of −290 g C m^{−2} y^{−1} and −140 g C m^{−2} y^{−1}, respectively. Since biomass production of *Typha* stands growing on clay or organic topsoils was approximately 1.5 times higher than on sandy soils (Overbeek et al., unpublished data), the lower net fixation can only have resulted from a higher emission of CO₂ from clay and organic soils through decomposition of labile organic matter, possibly as a result of a higher ROL by the

dominant helophytes (Bernal et al., 2016). Moreover, the organic soils generally showed higher emissions of CH₄ than sand and clay soils, especially from plots with *T. latifolia* vegetation, which may largely result from plant-mediated CH₄ transport (Bastviken et al., 2011; Bellisario et al., 1999; Bubier, 1995; Nykänen et al., 1995). Since contributions of ebullition events could not be included in the measurements, these CH₄ fluxes most likely underestimate actual emissions. Therefore, our results show that fast build-up of organic matter in the initial stages of a system may coincide with a net greenhouse gas emission, because of the 34 times higher global warming potential (GWP) of CH₄ compared to CO₂ over a time span of 100 years (IPCC, 2013).

4.3. Eutrophication risks

Apart from influencing C dynamics, there is a risk that higher nutrient concentrations in the soil may lead to enhanced nutrient mobilisation to the water layer (Zak and Gelbrecht, 2007), which may stimulate phytoplankton blooms resulting in lower production by peat-forming vegetation. Although clay soils contained more P than the other two soils, the very high Fe: PO₄³⁻ ratios (much higher than the threshold value of 1 mol mol⁻¹) prevented mobilisation of P to the water layer in all soils. For helophytes, however, a large part of the P in the soil will still be available through root uptake, which is supported by the higher plant-available (Olsen-extracted) P levels. Despite the high Fe: PO₄³⁻ ratio, PO₄³⁻ availability was somewhat higher in both pore water and the water layer in soils with an organic layer than in soils with only sand or an added layer of clay. Still, since surface water PO₄³⁻ concentrations remained mainly below 3 μmol L⁻¹ and no problems with algal blooms were observed, eutrophication problems did not occur.

4.4. Balancing pros and cons of topsoil addition

Although the application of clay and organic topsoils strongly increased both cover and net primary production of helophyte species, the strong variation observed in C fluxes and nutrient availability of these soils implies a “hit-or-miss” outcome in terms of other services, such as net C sequestration. Potential negative side effects in terms of nutrient mobilisation or CH₄ emission can, however, be estimated beforehand. Geurts et al. (2010) offered simple calculations based on pore water PO₄³⁻, Fe and NH₄⁺ concentrations to determine whether there is a risk of high mobilisation of P or N to the overlying water layer. On the other hand, expected C loss, as either CO₂ or CH₄, from the applied topsoils can be derived from simple laboratory incubations (Moore and Dalva, 1993; Van De Riet et al., 2013).

By combining information on colonisation and C sequestration potential, we set up a budget for constructed wetlands with a sand soil or sand with an applied clay or organic topsoil (Table 3). These budgets indicate that sand soils with an average cover of 15% of helophytes formed a net sink of CO₂, with an NEE_{CO2} of -51.1 g C m⁻² y⁻¹. Addition of clay topsoils increased the cover of helophytes to 40% and although CO₂ emissions from breakdown of organic matter increased, the system still formed a net CO₂ sink with an NEE_{CO2} of -20 g C m⁻² y⁻¹. Finally, the implementation of an organic top layer increased helophyte cover to 70%, but reduced the net fixation in the helophyte stands due to CO₂ emission by decomposition processes. As a result, these organic topsoils had an NEE_{CO2} of -67.4 g C m⁻² y⁻¹ and form the largest CO₂ sink of the three soil types (Table 3). Due to high CH₄ emissions in some basins with clay or organic soil and stands of *Typha latifolia*, some of these stands turned into net C sources emitting 670 ± 270 g C m⁻² y⁻¹, whereas others remained C sinks with net exchange rates of -370 ± 50 g C m⁻² y⁻¹. The high variation in NEE_C may have resulted from differences in availability of nutrients and

Table 3

C budget of peat capping using sand soils with and without an added layer of clay or organic soil. Note that negative values indicate C sequestration, whereas positive values represent a loss of C to the atmosphere.

Factor	Unit	Sand	Clay	Organic
<i>Typha</i> spp. ^a				
Cover <i>Typha</i>	%	15	40	70
NEE CO ₂	g C m ⁻² <i>Typha</i> y ⁻¹	-590	-290	-140
NEE CO ₂	g C m ⁻² basin y ⁻¹	-88.5	-16	-98
NEE CH ₄	g C m ⁻² basin y ⁻¹	7.6	90	305
Bare soil				
Cover bare soil ^b	%	85	60	30
NEE CO ₂	g C m ⁻² bare plot y ⁻¹	44	160	102
NEE CO ₂	g C m ⁻² basin y ⁻¹	37.4	96	30.6
NEE CH ₄	g C m ⁻² basin y ⁻¹	0.8	-0.7	32
Total				
NEE CO ₂	g C m ⁻² basin y ⁻¹	-51.1	-20	-67.4
NEE CH ₄	g C m ⁻² basin y ⁻¹	8.4	89.3	337
NEE C	g C m ⁻² basin y ⁻¹	-42.7	69.3	269

^a Fluxes from plots of *Typha latifolia* and *Typha angustifolia* were averaged to obtain NEE CO₂ and CH₄ rates.

^b The area not covered by *Typha* is assumed to be bare soil, which may cause a slight underestimation of the net C fixation, due to presence of some other emergent species (<10% cover).

readily decomposable organic matter in the sediment, which causes variability in the availability of substrates for microbial activity. Higher availability of PO₄³⁻, for example, has been reported to stimulate decomposition processes (Rejmankova and Houdkova, 2006), leading to higher C emission.

5. Conclusions and management implications

Because regular covers of landfills, usually constructed of HDPE, have limited lifespans (Allen 2001; Rowe and Sangam, 2002), there is a need to develop more sustainable capping methods for highly contaminated landfills, especially those occurring in wetlands. The realisation of a newly constructed, peat-forming wetland would not only provide such a sustainable method of bioremediation, but would also provide additional biodiversity, recreational and educational services. Although our study focuses on the ecological implications of transforming a polluted landfill into a growing peat-forming wetland, this approach brings other challenges and opportunities to the field of ecological engineering as well. In addition to common challenges that accompany the construction of new wetlands, such as vegetation development, release of greenhouse gasses and nutrient mobilisation from the sediment, building such sites on former landfills involves several safety issues. In these systems, active management remains necessary, including maintaining stable water levels, pumping excess water to a safe overflow area, removal of tree saplings to prevent rupture of the HDPE by roots, preventing major disruption of deeper layers, and monitoring the groundwater quantity and quality for toxic compounds in a comprehensive network (Egbring, 2011).

When such a wetland is created using only sand as a substrate, development will be slow and although patches of vegetation will form a net C sink, terrestrialisation and the uniform build-up of a peat layer throughout the entire wetland will take a long time. Furthermore, when this peat layer develops on top of inorganic sediments, the organic matter may remain mobile, illustrating the need for longer-term monitoring of these areas. By applying nutrient rich soils, such as clay or organic soils, the development of the vegetation is strongly stimulated, especially when these soils already contain propagules of target species. This will also limit sediment disruption and resuspension by wind action. The application of these soils will, however, almost always be accompanied by an initial C loss. When recycling clay or organic soils to stimu-

late vegetation development, however, the effect of C emission will be negligible on a landscape scale because although initially CO₂ and CH₄ emission due to decomposition of labile organic matter would result in a decrease of the thickness of the cap, this material would otherwise decompose when left exposed to the atmosphere (Gebert et al., 2006; Schrier-Uijl et al., 2011; Vermaat and Hellmann 2010). In general, we therefore conclude that when quick results are important, for instance when capping a highly contaminated landfill, the application of -preferably recycled- clay or organic soil is necessary to obtain a biomass production high enough for the build-up of an organic layer within a short period.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecoleng.2017.04.040>.

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