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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.

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 (CH_4) 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 (CH_4) 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, CO_2 sequestration rates and CH_4 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}	$251 \pm 57^{a,b}$
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 m^2, 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 m^2 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