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Rewetting former agricultural peatlands: Topsoil removal as a prerequisite to avoid strong nutrient and greenhouse gas emissions

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ABSTRACT

Globally 15%, and in Europe over 50%, of all peatlands have been drained for agricultural use leading to high carbon (C) losses, severe land subsidence and increased flooding risks. For the restoration of C sequestration and peat formation, abandoned peatlands are being rewetted at a large scale, but this transforms them into strong methane (CH_4) sources. Furthermore, due to the high topsoil nutrient contents and/or high buffering capacities of water used for rewetting, this will inevitably result in eutrophication of restored peatlands and downstream areas, which may compromise the regrowth of peat forming vegetation including *Sphagnum* spp.

To experimentally determine the extent of these negative side effects in relation to water quality, and to test topsoil removal as an abatement strategy, we used a controlled laboratory approach in which topsoil and subsoil monoliths of a former agricultural peatland were rewetted with water of different qualities (+P, $+HCO_3^-$, $+P/+HCO_3^-$ and Control), mimicking rainwater vs. surface water storage. In addition, two different *Sphagnum* moss species (*S. squarrosum* and *S. palustre*) were compared.

Without topsoil removal, rewetting led to high P and N mobilisation, algal blooms, and high CH_4, carbon dioxide (CO_2) and dissolved organic carbon (DOC) emissions. P-rich water resulted in further eutrophication. Bicarbonate (HCO_3^-) enrichment by surface water not only stimulated P release and CO_2 emission, but also strongly reduced *Sphagnum* vitality.

We conclude that topsoil removal will, at least in initial stages of rewetting, strongly reduce eutrophication problems (by 80–90%), CH_4 emission (99%), DOC loss (60%) and global warming potential (50–70%) of rewetted former agricultural peatlands. Furthermore, to reduce mineralisation rates and enable *Sphagnum* growth, storage of rainwater rather than surface water is preferred. Finally, removed topsoils can be reused in adjacent subsiding agricultural areas, and thereby optimise the overall C balance and allow higher water levels in rewetted peatlands.

1. Introduction

Land use change, hydrological operations and other forms of anthropogenic forcing have severely compromised the functioning of global wetlands and their services including flood protection, water purification, biodiversity and C sequestration (Foley et al., 2005; Zedler and Kercher, 2005). Approximately 15% of peatlands worldwide have been drained to accommodate agriculture, peat extraction, forestry or urbanisation (Joosten, 2009), although considerable differences exist between countries, with 10% to 85% of peatlands drained within a single country (e.g. Brock et al., 1999; Hooijer et al., 2012; Zanello et al., 2011; Meckel et al., 2006; Hoeksema, 2007). While the accumulation of thick peat layers has generally taken thousands of years ($-1.1\ mm\ yr^{-1}$; Ovenden et al., 1998), drainage of these systems has resulted in strong degradation by oxygen intrusion, enhancing aerobic decomposition of organic matter and carbon (C) emission. Together with compaction and consolidation (Hooijer et al., 2012), this has caused fast land subsidence ($2–150\ mm\ yr^{-1}$; Syvitski et al., 2009). Given the projected sea-level rise, this continuing subsidence of peatlands – often located in heavily populated coastal areas, river deltas and

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floodplains – poses a serious risk to public safety due to higher flooding risks (Syvitski et al., 2009; Temmerman et al., 2013).

Pristine, growing peatlands (mires) generally form net C sinks, in which the fixation of carbon dioxide (CO_2) into layers of organic matter exceeds the emission of methane (CH_4) and CO_2, leading to net ecosystem exchange (NEE) rates ranging from −5 to −40 g C m^{-2} yr^{-1} (Belyea and Malmer, 2004; Gorham, 1991; Lamers et al., 2015; Saarnio et al., 2007). Drained and degraded peatlands, on the other hand, are almost always net C sources (Alm et al., 1999; Waddington et al., 2001), with NEE rates ranging from +80 to +880 g C m^{-2} yr^{-1} (Lamers et al., 2015) and a yearly global emission of 30 to 370 Mt C yr^{-1} (Armentano, 1980). These high C emission rates from drained peatlands are the result of strongly increased aerobic ecosystem respiration rates and therefore mainly consist of CO_2 (Nykänen et al., 1995; Silvola et al., 1996; Waddington and Day, 2007). Emissions of CH_4, on the other hand, are much lower for drained peatlands than for pristine or rewetted peatlands (Moore and Knowles, 1989; Salm et al., 2009), as a result of the inhibition of CH_4 production and stimulation of CH_4 oxidation under aerobic conditions (Lai, 2009; Maljanen et al., 2010). Since CH_4 is a much more potent greenhouse gas than CO_2, its enhanced emission after rewetting strongly increases the global warming potential (GWP) of these restored systems. Due to the high availability of easily degradable organic matter (and therefore of acetate, CO_2 and H_2), and nutrients for methanogens (Aerts and Toet, 1997; Fiedler and Sommer, 2000), CH_4 emissions from rewetted, former agricultural peatlands may be considerable (Lamers et al., 2015). So far, however, only few studies have been published on CH_4 fluxes in these systems (e.g. Hendriks et al., 2007; van de Riet et al., 2013).

Since over 85% of drained peatlands have been used for agriculture (Joosten, 2009), they have been heavily fertilised and often also limed, resulting in an extremely nutrient-rich, buffered top layer of the soil. Especially phosphate (PO_4^{3-}) has accumulated in these soils, since it is strongly bound in iron (Fe) complexes and to organic matter under aerobic conditions (Lamers et al., 2015; Smolders et al., 2006, 2008). Upon rewetting, however, there is a considerable risk of PO_4^{3-} mobilisation and eutrophication of the peatland and downstream areas (Patrick and Khalid, 1974; Rupp et al., 2004; van Dijk et al., 2007). Furthermore, heavily fertilised soils often also turn into sources of N after rewetting (van de Riet et al., 2013; Van Dijk et al., 2004; Zak and Gelbrecht, 2007). Thus, although rewetting of drained peatlands may counteract land subsidence and CO_2 losses by restoring the anaerobic soil conditions and inhibiting complete organic matter oxidation, it may have several negative side effects in former agricultural systems. Therefore, despite seeming counterintuitive, removal of the easily degradable eutrophic topsoil may be a useful abatement strategy to prevent strong greenhouse gas (GHG) emission and nutrient mobilisation after rewetting and thus restore peat formation (Emsens et al., 2015).

The actual effect of rewetting a drained peatland will also strongly depend on the quality of the water used. Instead of conserving rainwater by building dams to counteract desiccation in peatlands, many areas have been rewetted by flooding them with surface water (Grootjans et al., 2002; Lamers et al., 2002; Roelofs, 1991). Especially in agricultural areas, the quality of this surface water may be compromised, with high nutrient concentrations and/or high buffering capacity (alkalinity). Both factors can be expected to have a strong influence on the peatlands' biogeochemistry and other services, including C sequestration (Lamers et al., 2015).

To fully restore the C sequestering function of a system and promote regrowth of peat, restoration of the original peat-forming vegetation is essential. Limitations in seed or spore dispersal, absence of viable seeds or spores in the soil or unfavourable habitat conditions may hamper natural return of peat-forming vegetation, such as *Phragmites*, *Carex* or *Sphagnum* species (Aggenbach et al., 2013; Campeau and Rochefort, 1996). Of these species, *Sphagnum* mosses produce more recalcitrant organic matter than other peat-forming species due to the unique characteristics of these ecosystem engineers (Van Breemen, 1995), including habitat acidification (Clymo, 1963; Hajek and Adamec, 2009; Van Breemen, 1995), production of organic matter with high phenolic contents (Yavitt et al., 2000), and high water retention, keeping the environment moist and anaerobic (Clymo, 1973). While *Sphagnum* can be reintroduced successfully on oligotrophic, acidic cut-over peatlands (Campeau and Rochefort, 1996; Robroek et al., 2009; Smolders et al., 2003), revegetation of eutrophic or alkaline soils may pose a serious problem to these mosses, since they may easily be out-competed by vascular plants (Aggenbach et al., 2013; Berendse et al., 2001; Smolders et al., 2008) or suffer from high pH (Andrus, 1986; Clymo, 1973; Hajek et al., 2006; Lamers et al., 1999).

Increased CH_4 production and nutrient mobilisation after rewetting of former agricultural peatlands have already been shown in the field (e.g. Hendriks et al., 2007; Zak and Gelbrecht, 2007). In field studies, however, determining the extent and impact of these two processes is difficult due to multiple biogeochemical interactions, complex hydrology and large variations in climatic parameters. Therefore, we chose a controlled, experimental approach to determine the extent of the GHG emission and eutrophication after rewetting of a former agricultural peatland using water of different qualities, mimicking rainwater, or surface water with high P and/or HCO_3^- availability. This controlled mesocosm approach also allowed us to quantify the effect of topsoil removal on the restoration of ecosystem services, by using both topsoils (5–20 cm) and subsoils (25–45 cm). Furthermore, to test whether the original fen vegetation, characterised by *Phragmites australis* and *Sphagnum* spp., could be restored after rewetting and topsoil removal, we introduced two species of *Sphagnum* (*S. palustre* and *S. squarrosum*) that are typical of this vegetation, and studied their growth potential under the different water and soil conditions and their contribution to C sequestration.

2. Material and methods

2.1. Experimental set-up

In autumn 2013, 32 peat monoliths (25 × 12 × 20 cm; length × width × height) were randomly collected from a drained and fertilised, agricultural peatland managed as a pasture in the north-western part of The Netherlands (Ilperveld; 52°44′075″N; 4°94′960″E). Cores were taken from two depths, 5–25 cm (topsoil; n = 16) and 25–45 cm (subsoil; n = 16), immediately transferred into glass aquaria (25 × 12 × 30 cm; $l \times w \times h$) and transported to the lab. Both layers consisted of *Sphagnum*/*Carex* peat, but as a result of drainage, the top layer was decomposed further, as illustrated by a higher bulk density (0.41 ± 0.03 vs. 0.19 ± 0.01 kg DW L^{-1} FW), lower organic matter content (47.9 ± 9.3 vs. 78.4 ± 1.8) and lower content of porewater phenolic compounds (2.75 ± 0.64 vs. 4.14 ± 0.85 mg L^{-1}). The two chosen depths were based on measurements of the Olsen-P (Olsen et al., 1954) profile showing much higher P availability (951 ± 71 vs. 153 ± 22 mg P kg DW^{-1}) in the top layer due to fertilisation and decomposition.

In the lab, demineralised water was added to 2 cm above soil level (0.6 L per aquarium), after which the aquaria were left for 2 weeks to acclimatise in a water bath at 15 °C (NESLAB cryostat, Thermoflex 1400, Breda, The Netherlands) with a light regime of 16 h light (400 W, Philips, Master Son-T PiaPlus, Belgium, 150 μmol m^{-2} s^{-1} PAR), mimicking Dutch summer conditions. All soils received artificial rainwater at a rate of 750 mm yr^{-1} corresponding to Dutch annual rainfall (250 mL; three times a week),