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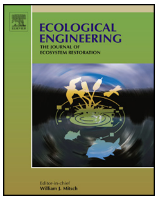

Peatland restoration and the dynamics of dissolved nitrogen in upland freshwaters

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ABSTRACT

This study assesses the impact of peatland restoration on fluvial N dynamics of south Pennine headwaters (UK) using a space-for-time approach. We monitored dissolved nitrogen in catchment drainage waters at intact, bare, and early stage restoration peatland sites over a two year period (Jan 2013–Dec 2014). Our study demonstrates that peatland restoration is effective in reducing dissolved inorganic nitrogen (DIN) leaching to levels lower than, or comparable to, the intact peatland site despite the adoption of a restoration approach involving fertilizer application in the revegetation process. In comparison with the bare site, DIN leaching was ~92% (10.2 kg N ha^{-1} yr^{-1}) lower at the restored site – where vegetation cover has been recently reintroduced. Whilst restoration increased the proportional significance of dissolved organic nitrogen (DON) when compared to the bare site, it was not to a level significantly different from what existed at the intact site. The results also reveal a strong positive relationship ($P < 0.001$) between DON and dissolved organic carbon (DOC) at all the sites, suggesting similarity of source material. Nitrate decreased with increasing DOC concentrations across the sites, suggesting the influence of organic C supply on NO_3^- immobilisation. In all but the bare site, average DON concentration was low in winter, but high in summer, and DIN concentration exhibited a winter-high and summer-low pattern; although year-on-year variations in this seasonal pattern was observed mainly at the bare site. Overall, our study has shown that restoration/revegetation is effective in advancing ecosystem recovery of degraded peatlands. Understanding nitrogen behaviour and trajectories as peatland restoration moves beyond early phases will require long-term catchment-scale monitoring.

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1. Introduction

There is widespread recognition of the uniqueness of peatlands in terrestrial biogeochemical cycles (Gorham, 1991; Evans et al., 2006a; Gallego-Sala and Prentice, 2013). Peatlands accounts for ~10% of the world's freshwater resources (Holden, 2005) and ~50% of global soil carbon storage (Gorham, 1991; Evans et al., 2006a). They provide a considerable range of ecosystem services, including clean water supply, recreation, food and fibre, and tend to attract large human populations to their vicinity (Bonn et al., 2009). In peatlands of the northern hemisphere, the formation of peat is primarily comprised of *Sphagnum*; a peat moss species that is highly resistant to decay, and thus, sequesters a vast amount of carbon (C) (Gorham, 1991; Holden, 2005). However, the stability and ability of peatlands to sequester C is greatly influenced by climatic parameters and anthropogenic factors (Ellis and Tallis, 2000; Pastor et al., 2003; Bragazza, 2008; Gallego-Sala and Prentice, 2013). A substantial increase in annual or mean monthly air temperature can increase water stress in peatlands and lower water table, resulting in faster decomposition of the peat matrix, and increased flux of dissolved organic carbon (DOC) (Freeman et al., 2001; Fenner et al., 2009). Drought severity also regulates nitrous oxide release from mires but could decrease peatwater ammonium (NH_4^+) concentrations (Freeman et al., 1993; Dowrick et al., 1999). Some peatland environments are in close proximity to major urban and industrial conurbations (Evans et al., 2000; Parry et al., 2014), thus exposing them to substantial pressures from numerous land-based activities (such as hill farming, rotational burning, grazing, field sports and tourism), and industrial atmospheric pollution (Evans, 1977; Yeloff et al., 2006; Mikutta and Rothwell, 2016). These pressures have resulted in significant degradation, including loss of vegetation, widespread gully erosion, and increased export of C, heavy metals and nitrogen (N) to the fluvial environment (Tallis, 1997;

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Evans et al., 2006a; Daniels et al., 2008; Rothwell et al., 2008; Allott et al., 2009; Evans and Lindsay, 2010).

Surface water acidity due to high deposition of sulphur and inorganic nitrogen has been reported in major peat-dominated upland regions of the UK (Evans et al., 2000; Helliwell et al., 2007a). Whilst a decrease in anthropogenically deposited sulphur in North America and northern Europe has been recorded in recent times (Fowler et al., 2007; Monteith et al., 2007; Yallop et al., 2010), atmospheric inorganic N deposition still remains high in some upland regions, exceeding 25 kg N ha^{-1} yr^{-1} (Helliwell et al., 2007a, 2007b; Edokpa et al., 2016). High concentrations of inorganic N promote the inhibition or loss of lichen and certain moss species including *Sphagnum*, as N uptake capacity is exceeded and vascular plants with high N demands (e.g. *Molinia caerulea*) and easily degradable litter become more competitive (Press et al., 1986; Baddeley et al., 1994; Lamers et al., 2000; Limpens et al., 2003; Solga et al., 2005; Caporn et al., 2006; Pilkington et al., 2007; Sheppard et al., 2011). The shift in vegetation composition from highly decay-resistant *Sphagnum*-dominated species to easily degradable vascular plant-dominated vegetation, in addition to N-enriched *Sphagnum* litter stimulates decomposition and N mineralization whilst reducing peat accumulation and C sequestration (Lamers et al., 2000; Berendse et al., 2001). This increases the loss of C and N to upland streams that are connected to landscape processes and downstream waters through their influence on nutrient transport which may not proceed conservatively as transformation occur between inorganic and organic species (Alexander et al., 2007; Scott et al., 2007). In this regard, particulate- and dissolved organic N (PON and DON) play a key role in driving and sustaining N spiralling within a fluvial network by providing a pathway for downstream N transfer (Durand et al., 2011).

Excess N enrichment of aquatic systems has raised a number of environmental concerns including the acidification and eutrophication of inland and coastal waters, shifts in food web structure and loss of biodiversity (Vitousek et al., 1997; Durand et al., 2011). Although, Peterson et al. (2001) demonstrates that headwater streams retain and transform over 50% of inorganic N input from their catchments, Alexander et al. (2007) found that first-order headwaters contribute about 65% of the N flux in second-order streams and 55% and 40% in fourth- and higher-order rivers including navigable waters, respectively. Because of the important role of headwater streams in downstream nutrient and solute transfer, quantitative information on N response to upland peat degradation and restoration is needed to better understand its potential downstream influence on the cycling and chemical speciation of N in this acid-sensitive fluvial environment.

There has been considerable investment on restoring the ecological integrity of degraded peatlands in Europe (Evans et al., 2005; Allott et al., 2009; Cole et al., 2014; Parry et al., 2014) and North America (Cooper and MacDonald, 2000; Rochefort et al., 2003; Andersen et al., 2006; Strack and Zuback, 2013), with re-vegetation representing a key component of the ongoing restoration effort. Studies have shown that vegetation and its uptake capacity play important roles in the immobilisation, biological cycling and leaching of N (Rothe and Mellert, 2004; Rowe et al., 2006). For example, Rowe et al. (2006) suggests that the upper threshold for nitrate (NO_3^-) leaching in deciduous woodland and acid grassland was lower than in coniferous woodland and heathland. Rothe and Mellert (2004) also observed higher NO_3^- concentration in mature spruce stands than in mature beech stands, whilst significantly higher NO_3^- concentrations were observed in clearcut areas compared with sites treated with small-scale regeneration techniques. Shuttleworth et al. (2014) demonstrates that re-vegetation of eroding peat gully systems stabilizes interfluve surfaces and decreases the export of legacy lead to levels that are lower or comparable with intact peat systems. Some other studies have shown that restoration of degraded blanket peat raises water table, sequesters C and reduces the export of particulates (Evans et al., 2005, 2006b; Allott et al., 2009; Dixon et al., 2014; Shuttleworth et al., 2014).

Whilst, existing studies on peatland restoration tend to focus on hydrology, sediment transfer, metals and C dynamics, there is a lack of research on the impact of degraded peatland restoration on N cycling dynamics, particularly in acid-sensitive upland peat environments. After more than a decade of restoration activities on some degraded peatland environments in the UK, and with some experimental restoration projects approaching the end of their active phase, an evaluation of its success with respect to organic matter cycling will require a holistic assessment of its impacts on not only DOC dynamics, but also dissolved N dynamics including DON – a component of dissolved organic matter (DOM) that links the C and N cycles. This study is therefore timely, as it investigates the impact of peatland restoration on the dynamics of dissolved inorganic N (DIN) and DON using a space-for-time (SFT) experimental design to assess sites with different status of degradation and restoration. This experimental design has a long tradition in ecological and environmental investigations (Fukami and Wardle, 2005; Pickett, 1989). Here, we analyse data from an intact peat site (I), a bare peat site (B), and an early stage restoration site (ESR). The findings of the study will provide a better understanding of the spatio-temporal dynamics of N in degraded peatlands undergoing restoration. Within this context, the data presented will provide significantly new insight on the impact of degraded peatland restoration on N speciation and help evaluate/refine present and future restoration strategies.

2. Study sites

This study was conducted in three small first-order headwater catchments draining into the Ashop River in the Peak District, south Pennines (Fig. 1). The catchments receive an average rainfall of ~1200 mm a^{-1} (2012–2014 data) and atmospheric inorganic N deposition of ~26 kg N ha^{-1} yr^{-1} (2010–2012 modelled mean value by the Centre for Ecology and Hydrology). The soils of the catchments are mostly blanket peats, characterised by underlying geology of carboniferous millstone grits. Detailed characteristics of the sites are provided in Table 1. There were two control sites: an intact Blanket bog (control 1) and a bare peat (control 2) site. To the researchers' knowledge, site I has not been exposed to any form of physical degradation or changes in slope or vegetation. The typical vegetation cover at this habitat is a general mix of *Calluna* vulgaris, *Empetrum nigrum*, *Vaccinium* spp., Ericaceous dwarf shrubs, pleurocarpus and *Rubus chamaemorus* mosses (Rodwell, 1991). Site B is severely eroded with no vegetation cover, and serves as a reference or control catchment for evaluating the impact of vegetation succession on N dynamics over time, at the restoration catchment. The ESR site was previously a bare peat catchment but has undergone restorative treatment since late 2010. As a result, the proportion of bare peat in the catchment has reduced to ~10%, with the emergence of vegetation cover that consists of grasses (39%), acrocarpous moss ssp. (27%), *Calluna vulgaris* (11%), pleurocarpous moss ssp. (6%), *Polytrichum* spp (4%), *Rumex acetosella* (3%), Liverwort spp. (2%), and less than 1% cover of *Chamerion/Epilobium* spp., *Cladonia* spp., *Salix* spp., *E. vaginatum* and *Betula* spp., as well as 34% cover of dead plant material (Pilkington et al., 2015). Details of the type of treatment received by this restoration catchment during the water quality survey are shown in Table 2.

3. Material and methods

The design of this study was based on the SFT approach. A potential limitation of this approach is the risk of misrepresenting spatial

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