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Research article

Forest clearfelling effects on dissolved oxygen and metabolism in peatland streams

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ABSTRACT

Peatlands cover ~3% of the world's landmass and large expanses have been altered significantly as a consequence of land use change. Forestry activities are a key pressure on these catchments increasing suspended sediment and nutrient export to receiving waters. The aim of this study was to investigate stream dissolved oxygen (DO) and metabolic activity response following clearfelling of a 39-year-old lodgepole pine and Sitka spruce forestry in an upland peat catchment. Significant effects of clearfelling on water temperature, flows, DO and stream metabolic (photosynthesis, respiration) rates were revealed. Stream temperature and discharge significantly increased in the study stream following clearfelling. Instream ecosystem respiration increased significantly following clearfelling, indicating an increase in the net consumption of organic carbon.

1. Introduction

Peatlands account for circa 3% of the Earth's total landmass (Bain et al., 2011), yield 10% of the world's freshwater supply and comprise one-third of the global soil carbon (Joosten and Clarke, 2002). Peatland streams play a significant role in the global carbon cycle and thus climate change, both by sequestering carbon and releasing it to the atmosphere (Billett et al., 2007). Peatland area has significantly diminished since the 1800s due to climate change and land use management (Joosten and Clarke, 2002). Peatland conversion to forestry was commonly adopted in north-western Europe, during the late 20th century (Paavilainen and Päivänen, 1995) with a view to improving an unexploited natural resource. These blanket peat forests have reached harvestable age and concerns have been raised about the impacts of forestry clearfelling on the receiving aquatic systems.

Human land use has been identified as a major threat to aquatic biodiversity and ecosystem functioning globally (Vörösmarty et al., 2010). Headwater streams constitute a large proportion of aquatic systems and have the ability to affect a large percentage of freshwater resources (e.g. Roberts et al., 2007). These streams are considered to have a far greater role in biogeochemical cycling than recognised previously (Benstead and Leigh, 2012) and can highlight land use impacts at the scale of first order streams which may be diluted at the catchment scale (O'Driscoll et al., 2013; Rodgers et al., 2010). Headwater peatland catchments contribute significantly to the biological and genetic diversity of north-western European countries (Ramchunder et al., 2011; O'Driscoll et al., 2012; Drinan et al., 2013). While there have been some detailed considerations of how land use changes affect peatland stream ecosystems using indicators such as macro-invertebrates and diatoms (Brown et al., 2013; O'Driscoll et al., 2012, 2014a; Ramchunder et al., 2009, 2012, 2013), few studies have examined the role of land use change on instream metabolism, and the diurnal fluctuations of DO (but see Young and Huryn, 1999).

In many northern European countries, coniferous trees are currently harvested in sensitive peatland forest catchments, raising concerns about the potential impact on the receiving waters

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(Nugent et al., 2003). An estimated 500,000 ha of peatland was afforested between the 1950s and 1990s in the UK and 300,000 ha in Ireland (EEA, 2004; Hargreaves et al., 2003). Many studies have reported catchment losses of suspended sediment and nutrients following forest harvesting activities on peatland (e.g. Cummins and Farrell, 2003; Rodgers et al., 2010). More recently studies have begun to examine the pathways and mechanisms of nutrient and sediment release (Asam et al., 2012, 2014; O'Driscoll et al., 2014b). While some studies have reported a reduction in DO concentrations in aquatic ecosystems following forest clearfelling activities on blanket peat (Drinan et al., 2013; Finnegan et al., 2014), the potential causes and effects on ecological processes have not been investigated.

Dissolved oxygen (DO) is one of the most vital components of water quality in surface water bodies (Brooks et al., 1997). It is essential for aerobic respiration at all trophic levels, particularly in headwater streams where some organisms (e.g. fish) may have high metabolic demands (Guignion et al., 2010). Diffusion from the atmosphere at the stream surface exchange, mixing of the stream water at riffles, and photosynthesis from in-stream primary production provide the principal sources of in-stream DO. DO can become depleted when water bodies become stagnant leading to increased consumption of oxygen by microbial organisms. Increased inputs of chemicals that react readily with oxygen in the stream (reduction of nitrate (NO_3) to ammonia (NH_4)) can also cause oxygen depletion. Temperature can affect DO concentrations physically with higher solubility of DO observed for colder waters, or indirectly via the significant role of temperature in ecosystem metabolism (Yvon-Durocher et al., 2010).

Alongside natural drivers of DO dynamics, stream DO concentrations can also be affected by forest management activities. Clearfelling may introduce brash material into receiving aquatic systems, potentially increasing organic matter supply and thus biological oxygen demand (BOD) (Lockaby et al., 1997). Forest clearfelling, site fertilisation and preparation can stimulate eutrophication via increased nutrient export to receiving waters, and increased light availability and temperature following canopy removal. Eutrophication generally promotes excessive plant growth and decay, eventually causing a severe reduction in DO. Peatland soils are characterised by low density and can be easily eroded in the absence of vegetation cover leading to increased suspended sediment export. Organic matter when present in suspended sediment is biologically active and as a consequence contributes to the oxygen consumption in streams during decomposition (Paavilainen and Päivänen, 1995). Moreover, suspended sediment might also reduce photosynthesis via reduced light penetration and bed smothering (Davies-Colley et al., 1992; Van Nieuwenhuyse and LaPerriere, 1986). An additional strong driver of changes in DO concentrations is likely to be altered stream thermal regime, after canopy removal leads to significant increases in net radiation (Hannah et al., 2008; Brown et al., 2010).

The overarching aim of this study was to increase understanding of the effects of land use change due to forest clearfelling on peatland stream ecosystems. DO concentrations were monitoring continuously over a two-year period in two first-order forested headwater streams in Ireland, both with high gradient channels and a bedrock/peat substrate. Commercial, non-native coniferous forestry was clearfelled from the catchment of one of these streams, while the forestry was left intact in the catchment of the control stream. Based on the findings of earlier studies of forestry clearfelling effects on receiving streams (Drinan et al., 2013; Finnegan et al., 2014) we hypothesised that following clearfelling: (H_1) there would be significant decreases in DO concentrations; (H_2) periphytic biomass would increase in the stream due to increased light, temperature and nutrients, (H_3) both water temperatures and stream flow rates would increase; and; (H_4) metabolic rates would rise post-clearfelling due to increased nutrients, light and temperature in the study stream. From a global perspective this study provides a unique opportunity to develop our understanding of the impacts of forest clearfelling on stream functional ecology. This need for understanding is essential for enhancing the conservation of freshwater ecology in these habitats, informing management practices and underpinning conservation schemes.

2. Methods

2.1. Study sites

The study was carried out from March 2009 to January 2012 in Glennamong, a 17.9 km^2 sub-catchment of the Burrishoole catchment, Co. Mayo (53°58′N, −9°37′E, 69 m a.s.l; Fig. 1). Catchment topography is mountainous with a maximum elevation of 716 m. The Burrishoole has a temperate oceanic climate due to its proximity to the Atlantic coast with mean annual rainfall of 1560 mm year^{-1} (McGinnity et al., 2009). The Glennamong catchment was planted in 1972 with a combination of Lodgepole pine (*Pinus contorta*) (86%) and Sitka spruce (*Picea sitchensis*) (13%) using spaced-furrow ploughing, creating furrows and ribbons (overturned turf ridges). The trees were planted on ribbons at 1.5 m intervals, giving an approximate soil area of 3 m^2/tree. Ground mineral phosphate was applied at a rate of 28.3 g per tree and was spot-applied manually immediately after planting. Stand density was reduced to ~2800 trees ha^{-1} by natural die off before clearfelling. The basal area for the stand was ~56 m^2 ha^{-1}. The remainder of the subcatchment is commonage (i.e. land that is owned by more than one person) and is grazed extensively by sheep. Peat soil depth at the sites is > 1 m and overlies mainly quartzite and schist bedrock.

Small catchments drained by two first order streams that flow directly into the Glennamong stream were studied. Both catchments are approximately 0.1 km^2 in area and the study streams had a mean width of approximately 50 cm. Both streams largely flow over bedrock although some sections have a peat substratum. One catchment was clearfelled during the study period and is herein referred to as Glennamong Study (GS). The second catchment received no management intervention during the study period and is herein referred to as Glennamong Control (GC; Fig. 1). Clearfelling commenced in GS on February 8th 2011 and finished at the end of March 2011. A harvester machine was used to clearfell the 9.4 ha GS catchment. Clearfelling at the GS was carried out in accordance with best management practices (BMPs) (Forest Service, 2000a, b), as far as practicable (see Finnegan et al., 2014 for more detail). The crown of the tree and associated residues (i.e. needles, twigs and branches) were collected to form windrows and brash mats which were used for machine travel thus protecting the soil surface and reducing erosion. The windrows/brash mats (~4 m wide) were laid parallel to the study stream and furrows on the harvested site, which were at right angles to the contours. Surface water flowed along the furrows and into collector drains that discharged into the study stream.

2.2. Instrumentation and sampling

GS and GC were instrumented at stable channel sections downstream of the forested areas in January 2010. H-flumes, a water level recorder (OTT SE200, Germany), a Datasonde (Hydrolab, USA) measuring water temperature and DO and an OTT® LogoSens 2 data logger (5 min resolution) were installed at both stations. Data loggers were downloaded and Datasondes recalibrated every four weeks. Periphyton samples were collected