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O'Driscoll, C, O'Connor, M, Asam, Z et al. (3 more authors) (2016) Forest clearfelling effects on dissolved oxygen and metabolism in peatland streams. Journal of Environmental Management, 166. 250 - 259. ISSN 0301-4797

https://doi.org/10.1016/j.jenvman.2015.10.031

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.

Keywords: stream metabolism; dissolved oxygen; water quality; temperature; forest clearfelling; blanket peat catchments

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 (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., 2012).

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; Figure 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 clear felling. 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.

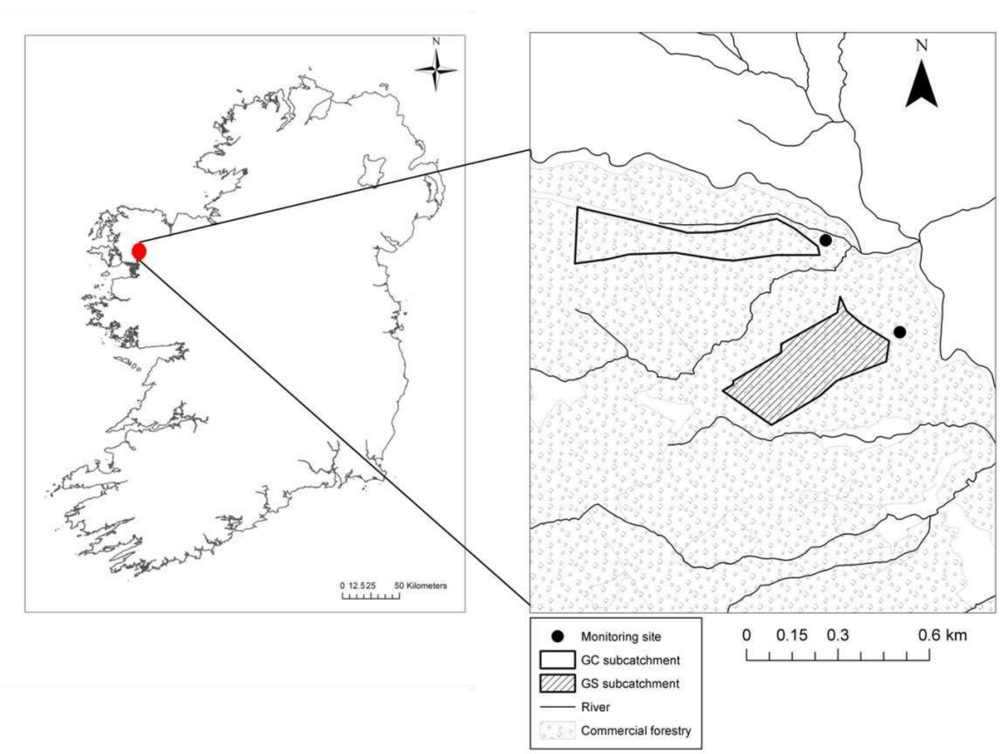

Figure 1.Map of the Glennamong catchment showing locations of the control (GC) and study (GS) areas.

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; Figure 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 quarterly (March 2009 – August 2011) from five cobble surfaces using 100 ml of stream water (Biggs and Kilroy, 2000). Orthogonal measurements of each stone were taken in the field, and converted to stone surface area (Dall, 1979). Samples were stored in the dark and analysed in the laboratory later the same day for chlorophyll a (Chl a) and ash free dry mass (AFDM) (APHA, 2005). Simultaneously, grab water samples were taken and analysed for total reactive phosphorus (TRP), ammonium-N (NH_4-N) and nitrate-N (NO_3-N) colorimetrically using a nutrient analyser (Konelab 20, Thermo Clinical Labsystems, Finland). Suspended sediment (SS) analysis was carried out by passing a known volume of water through a pre-dried and weighed 1.2 µm GF/C filter disc (Whatman, England) under suction. The filter and retained sediment were then dried at 105◦C for 24 h and reweighed to give the total SS (APHA, 2005).

Water samples were also taken as part of two complementary studies on a monthly basis in the pre-felling study period and weekly in the post-felling period using an automated ISCO sampler, the findings of which are published in O'Driscoll et al. (2014c) and Finnegan et al. (2014). Storm events were targeted as

Rodgers et al. (2010) found that nutrient concentrations were low during base flow conditions. The sampling frequency was dependent on the duration of the storm event which was estimated using weather predictions (e.g. hourly samples in a 24 h event; every 2 hours during a 48 hour event and so on). O'Driscoll et al. (2014b) examined the efficiency of a buffer zone on ameliorating nutrient and suspended sediment export arising from clearfelling. Finnegan et al. (2014) investigated the implications of applied best management practice for peatland forest harvesting incorporating phosphorus, nitrogen, suspended sediment, DO, electrical conductivity, pH and stream water temperature. This study examines in more depth the DO and temperature impacts arising from clearfelling by considering a longer pre-felling period and investigating flow regimes and metabolic processes.

2.3 Stream metabolism

Metabolism was calculated from measurements of diel temperature and DO concentration using the one-station open-channel method (Odum, 1956). Metabolism was estimated by quantifying changes in DO concentration at a single site over time and adjusting for DO exchange with the atmosphere (Bott, 2006). Flow duration curves were developed to distinguish between peak (Q_5) and low flow (Q_{95}), and metabolism data collected during peak flows were omitted due to the potential for inaccuracies in the estimated reaeration coefficient (DaSilva et al., 2013). Likewise for very low flows the following rationale was adopted. Open channel flow nozzles (H Flumes) with shaft encoders were employed and were rated under theoretical conditions to be operated at a minimum head of 0.005 m. However, to set the head for the device being used to record water level, a prerequisite is the need for water to be present in the flume, posing a potential source of error due to turbulence. For this reason water level of 0.01 m was chosen as a cut off point for limit of detection (negligible or no flows), to increase confidence in the flow data.

Ecosystem respiration (ER) and the reaeration coefficient (K) were estimated for each 24-h cycle using the night-time regression method (Hornberger and Kelly, 1975), for which the changes in DO concentration per hour during the dark period of each 24-h cycle (0:00 hours to sunrise and sunset to 0:00 hours) were regressed against the DO deficit so that ER (mg O^2 m^{-2} h^{-1}) is equal to the intercept and K (h^{-1}) is equal to the slope. Night-time periods that yielded regressions with significant slopes ($P < 0.05$) were used for the calculation of ecosystem metabolism. Additionally, the diel patterns of DO were inspected visually for outliers and 24-h periods were omitted where such data was present (Izagirre et al., 2007). Unlike empirical equations that are commonly used to estimate K, the night-time regression method allowed determination of K independently of discharge (Q), and thereby did not violate the assumption of independence for the later examination of metabolism variability as a function of Q using linear regression. $ER_{20°}$ and $K_{20°}$ were calculated from mean night-time temperature and then adjusted for specific temperature at each time interval following the methods of Erlandsen & Thyssen (1983) and Thyssen et al. (1983) respectively. Daily rates were determined from the sum of hourly rates. Gross primary production (GPP) for each time interval (dt) was calculated from the following equation:

$$GPP\ (dt) = dC / dt - K\ (Cs - C) + ER + A \tag{1}$$

where C is the concentration of DO at a given time, Cs is the saturating oxygen concentration, and ER is expressed as absolute values. Accrual of ground water (A) was judged to be negligible (Izagirre et al., 2007). Daily GPP (mg m^{-2} day^{-1}) was calculated from the sum of GPP rates for each time interval during daylight. The daylight period was calculated using times from a sunrise ⁄ sunset calculator (http://www.timeanddate.com/) for Dublin, Ireland, and 20 minutes added for the west of Ireland.

A caveat with all studies of whole stream metabolism is the accuracy of the estimations of the metabolic parameters with potential for error occurring throughout the stages of calculations (Demars et al., 2011). Indeed, methods of assessment for ecosystem functioning are still undergoing development (Aristegi et al., 2009; Demars et al., 2011) and improvement. One of the main issues is the quantification of the reaeration rate, but in a study comparing several methods (Aristegi et al., 2009) the night-time method proved the most robust and reliable among those tested. While the reaeration rates estimated in our study were derived from this method rather than being measured directly in situ (e.g. using gas additions), a consistent calculation and approach was applied to the two sites pre- and post-felling data; thus, the relative magnitude of effect between before-after and control-impact was considered reliable.

The volumetric rates of ER and GPP were converted to areal units (mg m^{-2} day^{-1}) by multiplying by the mean reach depth (m). Net ecosystem production (NEP) was the difference between GPP and ER, and P⁄R was GPP divided by ER. Negative ER is used to show oxygen consumption and a larger negative number indicates increased respiration. All metabolism calculations were performed using the RIVERMET© spreadsheet package (Izagirre et al., 2007). This analysis resulted in 198 estimates of open-water metabolism per year at GS and 224 at GC. There were a total of 83 concurrent values for the two streams, 55 before clearfelling and 28 after.

2.4 Statistical analyses

Comparisons of daily precipitation and air temperature in the years before and after clearfelling were made using repeated measures (RM) analysis of variance (ANOVA) models. Flow and DO measurements were averaged by day to reduce the number of observations and eliminate a falsely enhanced P value. Significance was determined at the 1% level unless otherwise noted. Repeated measures (RM) ANOVA models were used to test the significance of site (GS and GC) and treatment (before and after clearfelling) in DO concentration, and flow. Datasets were ‘matched’ by retaining only those days where data were available for both GC and GS. The data were tested for normality and homogeneity of variance to ensure the assumptions of linear modelling were met.

To assess whether clearfelling influenced stream thermal regime, a generalised regression approach was used to predict daily mean and maximum stream temperature at GS from measurements at GC following the methods of Gomi et al. (2006) and Dickson et al. (2012). Pre-clearfelling data was used to create models, and subsequently to assess the impacts of stem abstraction during clearfelling and in the immediate post-clearfelling phase.

Primary exploratory ordinary least squares (OLS) regression, autocorrelation analyses and Durbin–Watson statistics emphasised significant residual autocorrelation, thus generalised least squares (GLS) regression was used (Pinheiro et al., 2006). Models took the form

$$TGS = a + bTGC + ß2 \sin (2\pi j/D) + ß3 \cos (2\pi j/D) + e \quad (2)$$

where TGS = water temperature at GS, a = regression intercept, b = regression coefficient, TGC = water temperature at GC, j = calendar day of year, D = number of days in year (i.e. 365), and e = error term. Error terms were modelled as first order autoregressive processes based on a priori examination of autocorrelation and partial autocorrelation functions.

Regression models were used subsequently to predict water temperature in the post- clearfelling phase. The approximate statistical significance of clearfelling impacts was assessed by calculating a measure of random disturbance ($\hat{u}t$) (Gomi et al., 2006; Watson et al., 2001):

$$\hat{u}t = (yt - \hat{y}t) - \rho 1 \ (yt\text{-}1 - \hat{y}t\text{-}1) \quad (3)$$

where y is the observed water temperature and $\hat{y}$ is the predicted water temperature on day t, and $\rho 1$ is the lag1 autocorrelation coefficient from the GLS regression. 95% confidence intervals of disturbance estimates were calculated as 1.96 (σ $\hat{u}t$). If there was no effect of forest clearfelling on water temperature, $\hat{u}t$ in the post-felling period would be similar to the pre- clearfelling period; this hypothesis was tested with a two sample Kolmogorov–Smirnov test (Gomi et al., 2006). Linear mixed-effects (LME) models were fitted using the library nlme (Pinheiro and Bates, 2000) to test the significance of forest clearfelling on instream metabolism and primary productivity. This approach was adopted as there were repeated measures taken at the same sites over time. The model included Site (GS and GC) as a fixed effect and Treatment (before and after clearfelling) as a random effect. All statistical tests were performed with the R software package (R Development Core Team, 2008).

3. Results

3.1 Meteorological observations

The annual mean air temperature was 10.2 °C during the study period, varying from a daily minimum of -7.5 °C to a daily maximum of 18.8 °C. There was a significant difference between the pre- and post-clearfelled daily air temperature with higher temperatures observed in the pre-felling period (RM ANOVA, P < 0.0001) (Figure 2a).

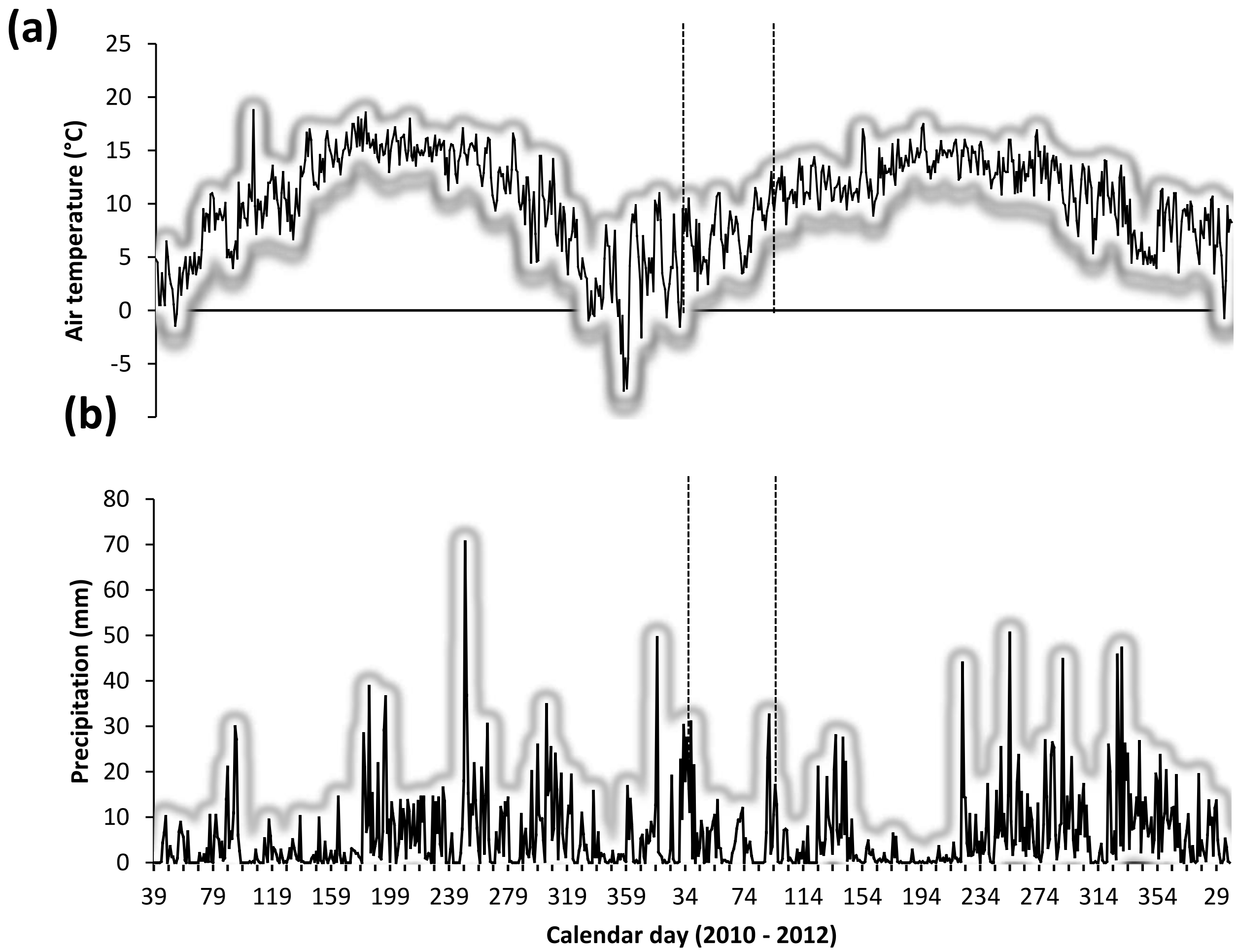

Figure 2 Time series of (a) daily air temperature, (b) total daily rainfall recorded at the Met Eireann weather station (circa 5.5 km south of the study site). Vertical dashed lines indicate the timing of forest clearfelling.

Total precipitation from a nearby Met Eireann weather station (circa 5.5 km south of the study site) was 1398 mm pre-felling (February 2010 – January 2011) and 1981 mm post-felling (February 2011 –

January 2012) (Marine Institute, unpublished data) being significantly higher after clearfelling (RM ANOVA, $P < 0.0001$) (Figure 2b). The number of wet days (>1 mm of precipitation/ day (Hundecha and Bárdossy, 2005)) recorded was 174 before clearfelling and 244 after clearfelling. The mean daily precipitation before clearfelling was 3.8 mm and 5.4 mm after clearfelling.

3.2 Streamwater observations

Nutrient concentrations in GS were low pre-felling with mean concentrations of 12.6 ± 2.2 µg L^{-1}, 75.1 ± 45.4 µg L^{-1}, <50.0 µg L^{-1} for TRP, NH_4^+-N and NO_3-N respectively (Table 1). Similarly, pre-felling mean concentrations at GC were 14.6 ± 3.9 µg L^{-1}, 71.4 ± 49.6 µg L^{-1} and 67.2 ± 24.1µg L^{-1} for TRP, NH_4^+-N and NO_3-N respectively (Table 1). The GS TRP concentrations increased two-fold from 12.6 ± 2.2 µg L^{-1} to 38.1 ± 23.6 µg L^{-1} and the GC TRP concentrations remained the same (Table 1). At GC the NH_4^+-N concentrations almost halved from pre- to post-felling periods (47.5 ± 30.9 µg L^{-1}) whereas the GS pre- and post-felling periods remained the same. The N: P ratio at GS shifted from 6.2 to 2.1after the onset of clearfelling (Table 1).

Table 1. Nutrient and suspended sediment concentrations in the GC (control) and GS (clearfelled) reaches, in the pre- and post-felling periods. (The summary statistics as indicated by ± represent the standard deviation. Nine storm events are represented in the pre- felling period and 18 in the post-felling period, with 24 water samples collected in each event).

	GC Pre-felling	GC Post-felling	GS Pre-felling	GS Post-felling
TRP (µg L^{-1})	14.6 ± 3.9	13.5 ± 5.30	12.6 ± 2.20	38.1 ± 23.6
N-NH4 (µg L^{-1})	71.4 ± 49.6	47.5 ± 30.9	75.1 ± 45.4	69.3 ± 26.8
N-NO3 (µg L^{-1})	67.2 ± 24.1	78.4 ± 63.7	<LOD	72.5 ± 51.4
SS (mg L^{-1})	116 ± 258	9.00 ± 8.60	75.1 ± 52.0	34.3 ± 31.7
DO (mg L^{-1})	11.3 ± 1.26	11.0 ± 0.94	10.5 ± 2.86	9.24 ± 2.84
N/P	5.00	5.50	6.20	2.10

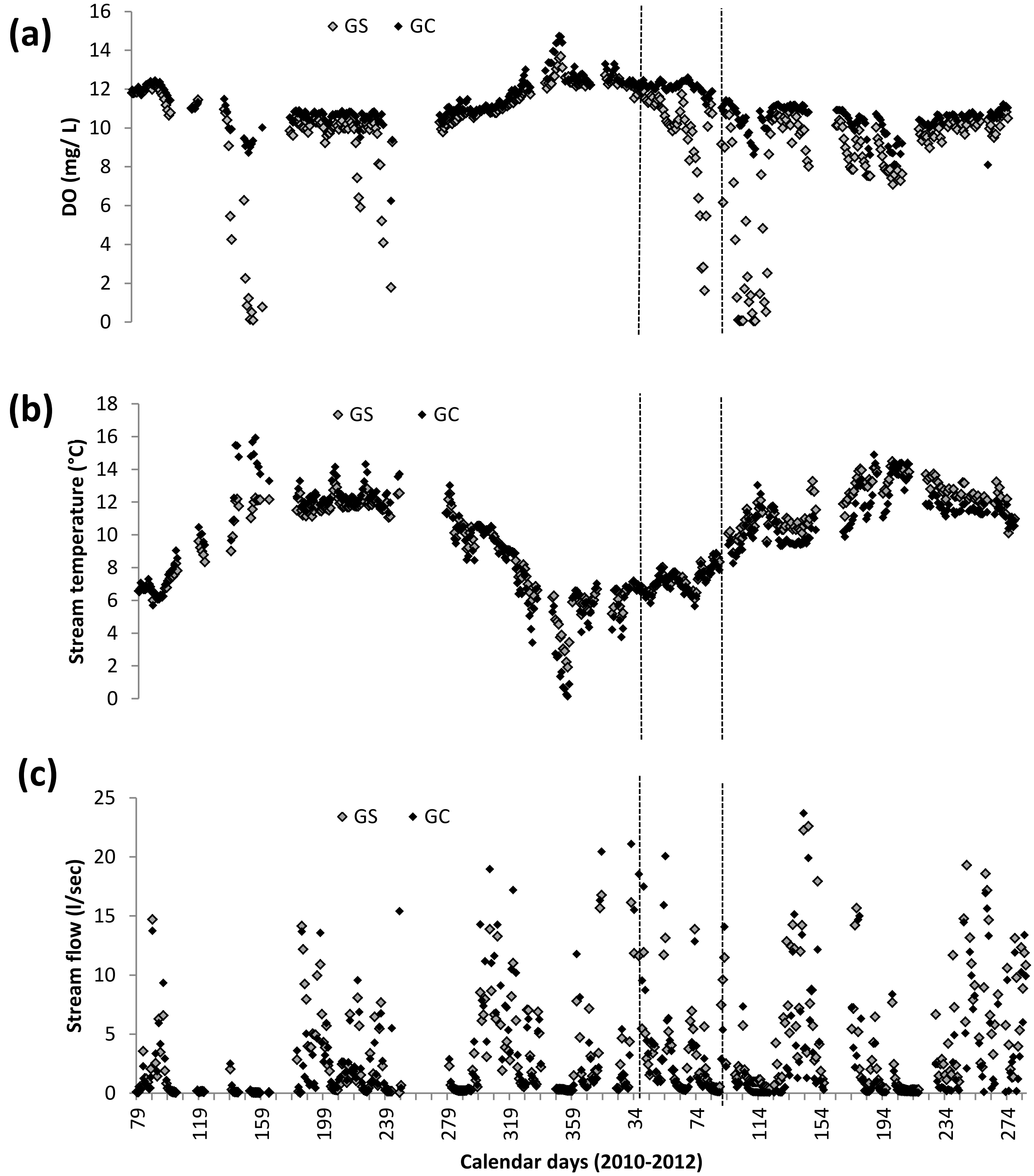

Figure 3 Trend of daily averages of (a) DO concentration (mg L^{-1}), (b) stream temperature (°C) and (c) stream flow (l s^{-1}) at control (GC) and clearfelled (GS) sites. Vertical dashed lines indicate the timing of forest clearfelling. Low DO occurred in the study catchment before clearfelling as well as afterwards. Gaps represent missing data.

Daily mean DO concentrations ranged from 6.24 to 14.7 mg L^{-1} at the GC and 0.05 to 14.8 mg L^{-1} at the GS over the duration of the study period (Figure 3a; Table 1). ANOVA results indicated that clearfelling significantly impacted DO (RM ANOVA, $P < 0.0001$) with lower values of DO concentrations observed in the GS post-felling than the GC. The scatterplot of daily mean DO concentration from GC to GS for both pre- and post clearfelling showed that more days had lower average DO concentrations at both sites post-felling compared with pre-felling (Figure 4). Of all the DO measurements pre-clearfelling, 1% recorded (5-minute increments) at GC and 15% at GS were less than 80% saturation. Post-felling, 2% of recorded DO measurements at GC and 23% at GS were less than 80% saturation.

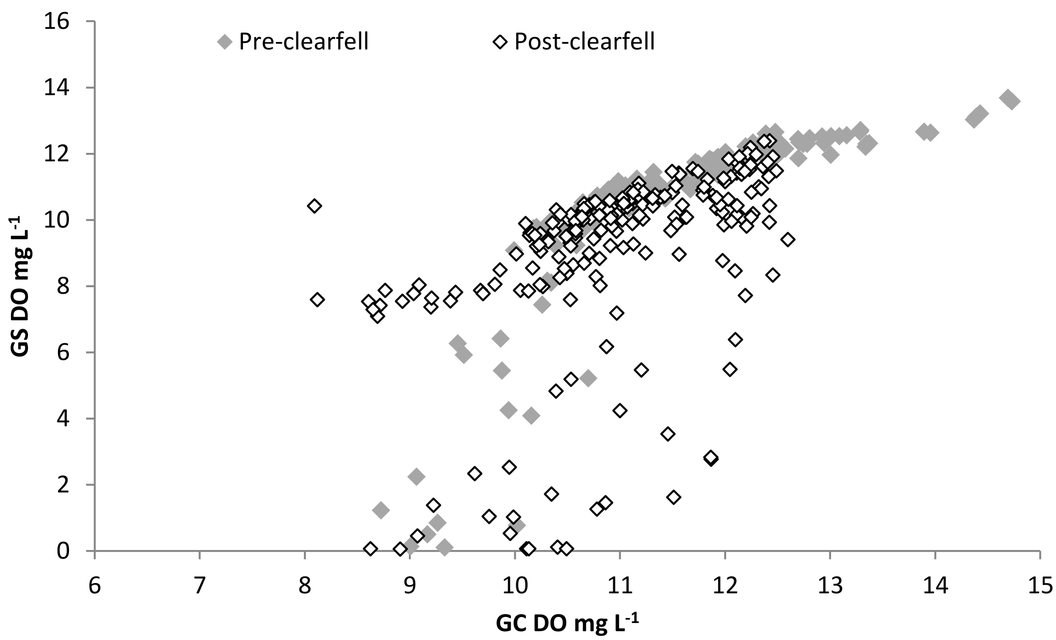

Figure 4. Scatterplots of daily-averaged DO concentration (mg L^{-1}) at GS vs GC during both pre- and post-felling periods.

Mean stream discharges over the duration of the study period were 3.60 l s^{-1} and 3.16 l s^{-1}at the GS and GC, respectively (Figure 3b). During the winter period, the mean low flow at GS increased 2-fold (from 2.29 – 4.68 l s^{-1}) whereas the GC observed a 1.5-fold increase (from 2.13 – 3.21 l s^{-1}). During the summer period, the mean low flow at GS also increased 2-fold (from 2.26 – 4.71 l s^{-1}) whereas the GC observed an 1.5-fold increase (from 2.15 – 3.12 l s^{-1}), a difference which was statistically significant (RM ANOVA, $P < 0.0001$).

Daily mean stream water temperatures in the pre-felling period ranged from 0.14 to 15.9 °C in the GC and 1.92 to 13.1 °C in the GS (Figure 3c). Post-felling stream temperatures ranged from 5.65 to 14.9 °C in the GC and 6.09 to 14.5 °C in the GS. Mean (±St. dev) ût values for pre-felling were -0.007 (±0.32), but post-felling these rose to 0.32 (±0.31), with the increase in ût post-felling being statistically significant (KS: D=0.69; P < 0.00001) (Figure 5).

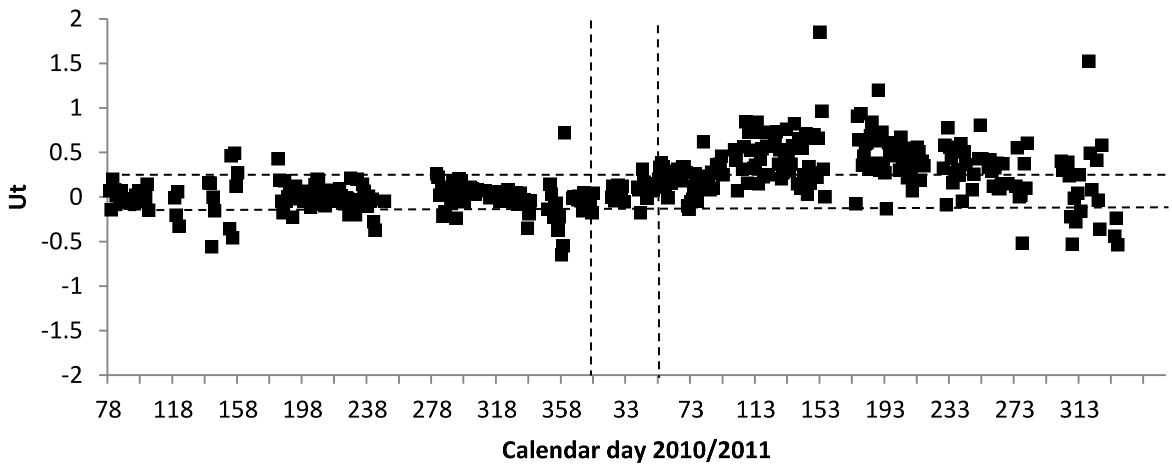

Figure 5 Mean (±St. dev) random disturbance ($\hat{u}_t$ values) pre-and post-felling over the duration of the study period. Vertical dash lines indicate the felling period. Random disturbance measured the approximate statistical significance of clearfelling impacts determined using regression models. Dashed horizontal lines in plots of treatment effects denote 95% confidence intervals of the pre-felling model.

3.3 Functional response to clearfelling

Ecosystem respiration (ER) was greater than GPP for all readings recorded indicating both the GS and GC streams were heterotrophic over the duration of the study period. Gross Primary Productivity (GPP) rates were not significantly different at the GS site before and after clearfelling (P = 0.7169) (Table 2). Ecosystem respiration (ER) was significantly different at the GS following clearfelling (P < 0.001) (Table 2). Net ecosystem production (NEP) was significantly different at the GS following clearfelling (P < 0.001) (Table 2). Chlorophyll a (Chl a) concentrations ranged from 0.1 mg m^{-2} to 42.9 mg m^{-2} at the GS and 0.4 mg m^{-2} to 11.7 mg m^{-2} at the GC and while the maximum Chl a values were observed in the post-felling period at the GS site the interaction term site x treatment was not significant (RM ANOVA, P>0.05). Ash Free Dry Mass (AFDM) ranged from 0.86 g m^{-2} to 11.6 g m^{-2} at the GS and 1.02 g m^{-2} to 5.91 g m^{-2} at the GC and the interaction term site x treatment was significant (RM ANOVA, P=0.04).

Table 2. Gross primary production (GPP), ecosystem respiration (ER), and net ecosystem respiration (NEP) rates, and chlorophyll a (Chl a) and ash free dry mass (AFDM) concentrations at GC and GS in the pre- and post-felling periods. (The summary statistics as indicated by ± represent the standard deviation. (There were a total of 83 concurrent values for the two streams, 55 before clearfelling and 28 after).* indicates significant differences pre- and post-felling for the GS catchment.

	GC Pre-felling	GC Post-felling	GS Pre-felling	GS Post-felling
GPP (mg O^2 m^{-2} d^{-1})	0.13 ± 0.17	0.23 ± 0.33	0.20 ± 0.25	0.21 ± 0.35
ER (mg O^2 m^{-2} d^{-1})*	-0.56 ± 0.97	-0.84 ± 0.91	-1.39 ± 1.26	-3.46 ± 5.85
NEP (mg O^2 m^{-2} d^{-1})*	-0.44 ± 0.89	-0.61 ± 0.66	-1.19 ± 1.17	-3.36 ± 5.63
Chl a (mg m^{-2})	3.50 ± 6.78	6.02 ± 6.33	7.55 ± 10.6	17.2 ± 27.1
AFDM (mg m^{-2})*	2.38 ± 3.14	2.61 ± 1.49	4.22 ± 3.95	9.45 ± 7.81

4. Discussion

The results of this study advance previous findings from peatland forest clearfelling research (Drinan et al., 2013; Finnegan et al., 2014, O'Driscoll et al., 2013) by increasing our understanding of the impacts on DO concentrations and associated drivers. Water quality impacts arising from peatland management have received increased attention over the past decade (Holden et al., 2007). Studies have focused on sediment, and nutrient export and bio-indicators such as macroinvertebrates (e.g. Brown et al., 2013; Drinan et al., 2013; Ramchunder et al., 2011; Rodgers et al., 2010, 2011). This study has provided the first detailed insight into how instream metabolic rates respond to forest clearfelling on blanket peat.

Periods of low DO occurred in both the GS and GC in the pre-felling period. Pre-clearfelling, 1% (5-minute increments) of the DO measurements recorded at GC and 15% at GS were less than 80% saturation.This is consistent with the findings of other studies (e.g. Da Silva et al., 2013; Ice and Sugden, 2003). Headwater forested peatland streams are further subjected to conditions that naturally limit DO such as the peat substrata on the stream bed (Drinan et al., 2013; Finnegan et al., 2014, O'Driscoll et al., 2013). Post-felling, 2% of recorded DO measurements at GC and 23% at GS were less than 80% saturation. DO saturation below 80% is considered to pose environmental risk for aquatic life by the Irish EPA (Bowman, 2009). The before-after-control-impact (BACI) comparison of daily mean DO concentration over the entire study period shows that although there were notable periods of low DO in the GS pre-felling, there was a significant increase in the number of occurrences post-felling thus supporting H_1, that there would be significant decreases in DO concentrations post-felling. The reductions in DO concentration could be attributed to higher concentrations of organic suspended sediment post-felling. Finnegan et al. (2014) reported higher organic suspended sediment at the GS following clearfelling, and as the organic component is biologically active oxygen could be utilised during decomposition (Greig et al., 2007; Paavilainen and Päivänen, 1995; Rodgers et al., 2011). This theory is supported by Drinan et al. (2013) where elevated readings of BOD and COD were observed in runoff water from the clearfelled GS catchment with the BOD exceeding guidelines for good status under the European Community Environmental Objectives (Surface Waters) Regulations 2009 (EC, 2009).

Removal of the canopy provides increased light to the stream which could directly enhance algal periphyton biomass (Lam, 1981; Holopainen and Huttunen, 1998). Moreover, increases in phosphorus concentrations to greater than 30 μg l^{-1} can trigger eutrophication in freshwaters (Boesch et al. 2001). However, while TRP concentrations did increase significantly (> 30 μg l^{-1}) in the impact stream after clearfelling, the increase was not reflected in Chl a biomass therefore H_2 was partly rejected. Due to the large export of terrestrial organic C in peatland streams (Ryder et al., 2014), in addition to the increased export following clearfelling (Drinan et al., 2013), aquatic primary production can be somewhat constrained by decreased light penetration (Arvola, 1984). However, benthic AFDM biomass did significantly increase following clearfelling supporting H_2. Increases in benthic AFDM were also noted by Brown et al. (2013) in peatland catchments subjected to burning and this was attributed to the increased vulnerability of organic soils to physical erosion. The emerging consensus is the effect of disturbance increases benthic organic matter in peatland stream systems (Brown et al., in press).

Clearfelling was shown to influence the thermal regime of the study stream with stream temperatures increasing significantly post-felling and supporting H_3. These findings align with previous studies (Gomi et al., 2006; Moore et al., 2005). Water temperatures in the post-felling winter period were warmer than the pre-felling winter period at both the GS and GC; however, temperatures at the GC exceeded the GS in the summer period pre-felling with the reverse observed in the post-felling summer period. Canopy removal eliminated the shading effect of the trees naturally implying a change in lighting conditions in the open stretches of the impact stream following clearfelling. Solar radiation is the predominant contributor of energy for summer warming in streams with no canopy (Bowler, 2012). Gomi et al. (2006) suggest riparian areas along streams protect the stream from increased thermal variability, with effects varying to some degree with buffer width. However, many of the earlier afforested blanket peat catchments in Ireland and the UK were established without any buffer areas, and trees were planted and harvested to the stream edge (Broadmeadow and Nisbet, 2004).

In addition to canopy removal, alteration of stream discharge could also impact on the stream thermal regime (Gomi et al., 2006). Headwater streams can be shallow and experience low discharge enhancing the opportunity for warming. The GS discharges increased two-fold following clearfelling thus supporting H_2. The increase was significant. This finding is similar to previous studies which attributed the increase to a reduction in evapotranspiration following tree removal (Robinson et al., 2003). Stream water temperature is known to have a clear impact on the bio-physico-chemical integrity of streams (Schlosser, 1991; Stott and Marks, 2000). This impact is expressed primarily through its regulation of DO solubility in water (Horne and Goldman, 1994) and the growth, metabolism, and respiration of aquatic organisms (Eckert, 1988). It has been previously reported that it takes several years for upland blanket peat sites to revegetate following clearfelling (O'Driscoll et al., 2011); however, it is not clear how the thermal regime recovers with recovering growth in vegetation in the ensuing years.

The stream metabolism rates observed for this study were within the range reported across numerous other streams (e.g. Clapcott and Barmuta, 2010; Da Silva et al., 2013) with GPP consistently below ER (e.g. Marzolf et al., 1994; Young and Huryn, 1996; Demars et al., 2011). These findings provide some confidence in the use of the single-station open-channel method, and complement earlier studies which report that headwater streams have higher rates of respiration than primary production and a larger export of terrestrial organic C can be expected from forested peatland streams (Birkel et al., 2013). No significant impact of clearfelling was observed on GPP, suggesting that heterotrophic processes dominated prior to felling and the increased light and nutrients made available by the clearfelling did not alter this state. ER, however, increased significantly following clearfelling. This finding contrasts with Da Silva et al. (2013) who reported no impact on stream metabolism following forest clearfelling. Clapcott and Barmuta (2010) found significant increases in the mean values of all functional variables with clearfelled streams in comparison to the control catchments but reported that degree of response depended on the underlying geology. The decrease observed in NEP indicates an increase in the net rate of organic carbon consumed (Young et al., 2008). Forest clearfelling can cause inputs of fresh brash into receiving waters (Campbell and Doeg, 1989; Lockaby et al., 1997), stimulating heterotrophic processes (Clapcottt and Barmuta, 2010), and Drinan et al. (2013) reported elevated BOD in the GS following clearfelling supporting this theory.

Ponce (1974) reported on the high demand for oxygen exerted by microbes associated with fresh brash and mentioned it can rapidly deplete DO concentrations in receiving water. The rate of chemical release and oxygen demand is highest during the first two weeks post-clearfelling, which coincides with the time in this study when a sharp decline in DO occurred in the study stream. Drinan et al. (2013) also noted increased COD in the study stream immediately following clearfelling. Clearfelling was carried out in accordance with BMPs with respect to harvesting plans, coupe size, timber landing areas, use of brash mats, as far as practicable (Finnegan et al., 2014). However, a site inspection highlighted that a brash mat used by the forwarder for carrying timber off the site, was laid over the study stream in dry conditions. It is likely that the oxidation of organic matter and nitrification of the decomposing brash was the mostly likely cause of the observed DO minima post-felling. The immediate reduction in DO could have been caused by the increase in BOD and heterotrophic processes; however, the open station method does not allow discrimination between BOD and COD therefore further work is necessary to unpick these drivers within peatland streams. Heterotrophic bacteria are the primary mediators in the process of terrestrially derived carbon degradation (Battin et al., 2003; Meyer, 1994) but limited work has been carried out elucidating these responses. Finnegan et al. (2014) recommended that site inspections for harvesting plans are carried out during or immediately after a period of prolonged rainfall. Further research is warranted into the use of metabolism as an indicator of land management impacts in blanket peat catchments as it is one of the most integrative ecosystem functions.

5. Conclusions

Peatland management operations in many northern European locations are, on the whole, environmentally challenging due to high annual precipitation, high soil water content, low ground-bearing capacity and the typically ecologically sensitive nature of the receiving waters. This study has provided a detailed insight into the effects of blanket peat clearfelling on instream DO and associated drivers and processes. Clearfelling significantly reduced the DO concentrations in the stream and this was most likely linked to changes in respiration owing to increases in stream temperature and nutrient export to the stream following clearfelling. The findings of this study highlight the need to develop our understanding of whether natural revegetation establishment will enable recovery of the stream thermal regime and associated biogeochemical processes to pre-clearfelling levels. Furthermore, quantifying riverine GPP and ER, and linking these to nutrient cycling under different flow, temperature and nutrient concentration conditions, is needed to provide baseline data to underpin assessment of management interventions and environmental change effects in peatland systems.

Acknowledgements

This study was funded by the SANIFAC and For Site projects which were funded by the Department of Agriculture, Fisheries and the Marine under the STIMULUS Programme 2007–2013. The authors wish to acknowledge the input of Coillte in allowing access to state forestry for this project, and for providing logistical support and advice. The authors gratefully acknowledge the assistance of the Marine Institute. Two anonymous reviewers provided insightful comments on the manuscript.

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