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CATCHMENT-SCALE PEATLAND RESTORATION BENEFITS STREAM ECOSYSTEM BIODIVERSITY

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SUMMARY

1. Drainage of peat-dominated catchments across the world has caused widespread degradation of the ecosystem services provided by peat and freshwaters. In the UK, an estimated £500 million has been spent over the last decade blocking drains to reverse these changes. The practice raises water tables to induce rewetting and promote peat aggradation. However, the potential benefits for impacted ecosystems such as streams remain unknown.

2. This study examined stream physicochemistry and benthic macroinvertebrates across peatland catchments with widespread artificial drainage networks, or drains that have recently been blocked, and compared these with intact peatland sites having no history of drainage.

3. Streams in artificially drained catchments were characterised by more benthic fine particulate organic matter (FPOM), higher suspended sediment concentrations and finer bed sediments (D_{50}) than in drain-blocked and intact catchments.

4. Drained sites had higher abundance of Diptera (Simuliidae, Chironomidae) larvae, and lower abundance of Ephemeroptera, Plecoptera and Trichoptera larvae, than drain-blocked sites. In contrast, streams in drain-blocked catchments had macroinvertebrate communities broadly similar to intact sites in terms of taxon richness, overall species composition and community structure. These changes were associated with lower suspended sediment and FPOM concentrations following drain-blocking.

5. *Synthesis and applications.* For the first time, this study has shown changes in the structure of stream benthic macroinvertebrate assemblages linked to increases in benthic POM and suspended sediment following peatland drainage. However, these effects seem to be reversible following catchment-scale restoration by drain-blocking.

Drain-blocking therefore appears to benefit not only peatland soil, vegetation and hydrological ecosystem services but also stream water quality and biodiversity. The numerous agencies undertaking peatland restoration should consider implementing detailed pre- and post-blocking monitoring of streams to: further improve our understanding of the mechanisms through which peatland management affects stream biodiversity and biological recovery dynamics, refine drain-blocking practices, and; inform aquatic conservation and management strategies.

INTRODUCTION

River catchments worldwide have been altered significantly as a consequence of land-use change (e.g. Harding *et al.*, 1998; Paul & Meyer, 2001; Allan, 2004). Peatland covers around 500M ha globally but, historically, many countries such as the Netherlands, Finland, Russia, Ireland, UK, New Zealand, Canada and Ecuador have witnessed broad-scale artificial peatland drainage. However, the full catchment-scale effects of these drainage practices have rarely been examined (Ramchunder *et al.*, 2009). In the UK, which is one of the most extensively drained lands in Europe (Baldock *et al.*, 1984), some drainage predates Roman times (Darby, 1956) although many peatlands were drained during the 1960s and 1970s to increase grouse, sheep and timber production, provide peat for horticulture and fuel, and with the intention of reducing downstream flooding (Holden *et al.*, 2004). At the time it was considered that this management intervention would improve the ecosystem services provided by upland moorland but, as Stewart and Lance (1983) noted, there was no evidence of economic benefits from moorland drainage and little research was undertaken into the wider environmental impacts.

Artificial drainage of peatland is now considered to impose a range of deleterious effects including alterations to baseflow and stormflow runoff regimes with some evidence of increased flow peaks and lower baseflows (e.g. Conway & Millar, 1960; Holden *et al.*, 2006a). Impacts resulting from artificial drainage of peat have also included: changes to soil chemistry leading to changes in solute (e.g. calcium, magnesium, potassium) and nutrient (e.g. carbon-bound nitrogen and sulphur and organically bound phosphorus) concentrations of streams (e.g. Ramchunder *et al.*, 2009), and erosion of exposed peat surfaces resulting in increased sediment delivery to receiving waters (e.g. Holden *et al.*, 2007). In an attempt to restore peatlands following such degradation, the practice of installing peat drain blocks or dams started in the UK in the late 1980s (Armstrong *et al.*, 2009).

Drain-blocking has so far been implemented over >3200 ha of England alone (Holden, 2009), with common practice being a series of dams rather than total infilling. The principal aim of drain-blocking is to increase water table height, encouraging re-vegetation by peat forming species such as *Sphagnum*. The restorative capabilities of drain-blocking on terrestrial vegetation and soil structure/chemistry are reviewed by Ramchunder *et al.* (2009) and the efficacy of different blocking methods is detailed in Armstrong *et al.* (2009). Damming at intervals along each artificial drain has reduced drain discharge in some places by over 70%, while alterations to the hydrological routing of water have also been observed (Worrall *et al.*, 2007). Soil water dissolved organic carbon concentrations have been shown to be less in drain blocked peatland areas compared to where drains are still open (Wallage *et al.*, 2006). However, questions remain as to whether catchment-scale restoration by drain-blocking has any benefits for stream biodiversity. Therefore, further knowledge would be useful given the general need to increase understanding of the ecological benefits of catchment-scale remediation schemes (Boon, 1998; Hillman & Brierley, 2005), which have received comparatively less attention than restoration schemes focused on river sections or reaches (e.g. Bernhardt *et al.*, 2005; Palmer *et al.*, 2005).

This study investigated stream macroinvertebrate communities from nine headwater (2-4 km^2) peatland catchments (three intact, three with widespread artificial drainage networks and three with blocked drains), on five occasions from 2007 to 2008. Based on knowledge from hydrological studies of peatland drainage and drain-blocking, it was hypothesised that: (1) stream systems in catchments that possess artificial drainage networks would have higher suspended sediment concentrations, finer bed sediments and more benthic fine particulate organic matter (FPOM) when compared with stream systems from intact catchments (Prévost *et al.*, 1999; Holden *et al.*, 2007). These changes to the stream environment were expected to (2) lead to macroinvertebrate communities containing taxa associated more with in-stream fine sediment deposition and benthic POM (e.g. Ramchunder *et al.*, 2009) compared with stream systems from intact catchments. In contrast, (3) drain-blocked stream ecosystems were expected to be in a similar condition as those at intact sites, because drain dams effectively reduce sediment flux to the stream network (Holden *et al.*, 2007), and restore hydrological and hydrochemical aspects of the terrestrial ecosystem. The findings of this study are subsequently considered in the context of more general literature discussing the effects of land management and catchment-scale remediation schemes on stream ecosystems.

METHODS

Study areas

The study was undertaken at sites in the north Pennines and Yorkshire Dales, northern England, specifically at Moor House Upper Teesdale National Nature Reserve, Wensleydale, Wharfedale, Geltsdale and Weardale (Table 1). All sites have blanket peat cover, with vegetation dominated by *Sphagnum* spp. (mosses), *Eriophorum* spp. (cotton grasses) and *Calluna vulgaris* (heather) (Radley, 1962; Holden & Burt, 2003a). The climate of all sites is typical of the UK uplands and, although data are not available for all nine locations, mean annual precipitation measurements of 2012 mm (1951-1980 and 1991-2006) at Moor House (Holden & Rose, 2011) and 1817 mm (1981-2008) at Oughtershaw Beck (Wallage *et al.*, 2006) can be considered broadly representative. Mean annual air temperature at Moor House is 5.3°C (1931–2006; Holden and Rose, 2011) and 6°C at Oughtershaw Beck (2007-2009, Brown *et al.*, 2010a).

Potential study sites were identified initially as those having second order streams based on 1:25,000 Ordnance survey maps. Catchment (1.08–3.52 km^2; Table 1) and stream size were standardised as far as possible following Ramchunder *et al.* (2011) who suggested catchment size (and thus stream order) can have a significant effect on peatland stream macroinvertebrate community composition. Candidate drained and drain-blocked (3-11 years post blocking) sites were identified from aerial photographs and following discussions with local land managers. Sites were selected on the basis of having no confounding effects of wildfire, rotational heather burning, mining, major erosion or forest cover. At each catchment outlet, a representative 15 m reach was selected randomly for detailed study with all subsequent sampling undertaken in riffle areas of those reaches.

Field sampling

Streams were sampled on a seasonal basis across two to four days per quarter (2007: Sept. 03-07; Dec. 04–08; 2008: Mar. 05–09; June 04–08; Sept. 02–06). On each sampling date, 16 stream environmental variables were measured (Table 2). Water temperature, pH and electrical conductivity (EC) were measured using Mettler Toledo MP120 and MP126 handheld probes (Mettler-Toledo, Ltd, Leicester, UK). Dissolved oxygen (DO) concentration was measured using a Hanna Instruments HI9412 probe (Hanna Instruments Ltd, Bedfordshire, UK). In addition, 120 ml of stream water were passed through a 0.45 µm filter and subsequently analysed in the laboratory for major anions (Cl, SO_4 and NO_3), dissolved organic carbon (DOC) and dissolved metals (Al and Fe). A further 500 ml of unfiltered stream water was collected for determination of suspended sediment concentration (SSC) by filtration. Streambed sediments were characterised by measuring b-axis length and median grain size (D_{50}) of 100 randomly sampled clasts. To provide a relative indication of flow differences between sites and over time, flow velocity was measured at the time of sampling using a Valeport open channel flow meter (Valeport Ltd, Devon, UK), with the channel cross-section surveyed and stream discharge (Q) calculated using the velocity-area method.

Five replicate 0.05 m^2 benthic macroinvertebrate samples were collected from riffle habitats using a modified Surber sampler with a 250 µm mesh net. Sediment was disturbed to a depth of ~10 cm for approximately three minutes per sample. All samples were preserved immediately in 70% ethanol then transported back to the laboratory for sorting and identification. Where possible, macroinvertebrates were identified to species level under a light microscope (x40 magnification) but other taxa were identified to higher levels (e.g. Diptera [Family/Genus], Oligochaeta [Class]) using standard UK freshwater macroinvertebrate identification keys. Particulate organic matter retained within the Surber samples was sorted into fine (<1 mm) and coarse (>1 mm) fractions and ashed to determine ash free dry weight.

Data Analysis

Principal component analysis (PCA) was used to examine inter-relationships between the 16 environmental variables across all sampling dates. Principal components (PCs) with Eigenvalues >1 were retained and the % variance of each recorded. Repeated Measures ANOVA (with season as the repeated measure) with Bonferroni correction was used to determine whether there were significant differences in stream environmental variables and PC scores as a function of site, land management type and season. Site was included as a random factor in the model while land management and season were fixed.

Prior to the analysis of macroinvertebrate data, the five replicate invertebrate samples collected for each site/date were pooled (e.g. Brown *et al.*, 2007) to enable clearer elucidation of the land management impacts as opposed to patch-scale variability. Macroinvertebrate community structure was summarised using five measures: (1) $\log_{10}$ (total abundance +1) expressed as the total number of individuals per m^2; (2) taxonomic richness; (3) Relative abundance (%) of Ephemeroptera, Plecoptera, Trichoptera, Chironomidae, Simuliidae and Other taxa; (4) 1/Simpson's diversity index (1/*S*):

$$1/(S = \frac{\sum n_i (n_i - 1)}{N(N-1)})$$

where N is the total number of individuals in a sample, and n_i is the number of individuals of taxon i;

(5) Taxonomic dominance or evenness (D) estimated using the Berger-Parker index:

$$D = N_{max} / N$$

where N_{max} is the number of individuals in the most abundant species and N is the total number of individuals collected.

RM-ANOVA was repeated for the macroinvertebrate summary measures using the same methods as outlined above. All environmental and macroinvertebrate data sets were tested for normality and, where necessary, $\log_{10}$, arcsin or square root transformed to improve normality and homogeneity of variance prior to statistical tests. All statistical tests were undertaken in SPSS v17.0 or Minitab v15 and considered significant where $P<0.05$.

Species-habitat relationships were assessed using multivariate ordination in CANOCO v4.5. Macroinvertebrate data were $\log_{10}$ (x+1) transformed prior to analysis. A preliminary Detrended Correspondence Analysis, showed axis 1 gradients were 3.0SD, thus, subsequent analyses used direct gradient analysis (Redundancy Analysis; RDA) to test for linear trends in species compositional change (Lepš & Šmilauer, 2003). Forward selection was used to determine which of the physicochemical variables accounted for a significant proportion of the species variance. An initial RDA included a dummy variable 'Time' (no. days from start of sampling) to determine whether there were significant temporal trends within the stream macroinvertebrate communities. Thereafter, a partial RDA (*p*RDA) was carried out to remove the variance accounted by Time, in an attempt to provide a better indication of the spatial component of the dataset (Borcard *et al.*, 1992). One-way analysis of similarity (ANOSIM) was employed to test the null hypothesis that differences in stream macroinvertebrate taxa abundance between peatland management types were not different to those within types. ANOSIM was undertaken on all samples combined owing to the small number of replicates per quarterly sample collection, and because spatial dynamics (linked to management type) were the key focus of this analysis. ANOSIM was undertaken using both the Bray-Curtis (BC) dissimilarity index (based on taxa relative abundance) and Jaccard's coefficient of similarity (based on taxa presence-absence), with 10,000 permutations and Bonferroni corrections using PAST 2.05 (Hammer *et al.*, 2001).

RESULTS

Mean SO_4, POM (fine and coarse), SSC and Al concentrations were all highest in drained streams but mean DOC concentrations and D_{50} were lowest (Table 2). EC and pH were highest on average in the drain-blocked sites, although the highest recorded pH was from an intact peatland stream. Cl, NO_3 and SO_4 concentrations were all lowest on average at intact sites. The repeated measures ANOVA showed significant differences in SSC, FPOM, D_{50}, and NO_3 (with seasonal interaction) between peatland management types (Table 2).

The PCA generated five PCs with Eigenvalues >1 (Table 3 & Fig. 1). PC1 had strong positive loadings (>0.5) of NO_3, EC, FPOM and POM and a strong negative loading of Al. PC1 scores were significantly different between management types ($F_{(2, 44)}$ = 6.3; P = 0.004). The Tukey's post-hoc test showed that PC1 scores were significantly different between streams in intact and artificially drained catchments (P = 0.003). Streams with artificial drainage were characterised by more negative PC1 scores but they were not significantly different to those for drain-blocked streams (P > 0.05). PC2 had strong positive loadings of coarse particulate organic matter (CPOM), FPOM, POM and Fe, and strong negative loadings of pH and D_{50}. PC2 scores were also significantly different between management types ($F_{(2, 44)}$ = 15.8; P <0.001). The

Tukey's post-hoc test showed that sites with artificial drainage were characterised by more positive PC2 scores (Fig. 1), and there were significant differences between intact and drain-blocked (P = 0.022), intact and artificially drained (P = 0.018), as well as between drain-blocked and artificially drained sites (P <0.001).

Mean abundance and richness were highest in drain-blocked and intact sites respectively, while mean 1/S and dominance were highest in intact and drain-blocked sites respectively. Lowest mean abundance, richness, 1/S and dominance were documented in the drained sites (Table 4). The RM ANOVA revealed a borderline significant difference in taxonomic richness between peatland management types (Table 3 & Fig. 1). Ephemeroptera and Trichoptera relative abundance were typically higher in the intact and drain-blocked sites when compared with the drained sites (Fig. 3), whereas Plecoptera and other taxa relative abundance were similar across the three management types. The relative abundance of Chironomidae and Simuliidae were typically highest in artificially drained systems (Fig. 3).

Axis 1 of the RDA accounted for a total of 16.2% of the total variance while the second axis accounted for 7%. Taxa-environment correlations were 0.790 and 0.876 for axis 1 and 2, respectively. Time accounted for 10.1% of the species variance, thus a *p*RDA was undertaken. Axis 1 and 2 accounted for a total of 17.0% and 7.2% of the total variance while taxa-environment correlations were 0.791 and 0.853 for axis 1 and 2, respectively. Forward selection showed pH, SSC, Al, Fe, FPOM, SO_4 and water temperature were associated with a significant proportion of the variance.

The analysis showed that the intact and drain-blocked sites were associated with low suspended sediment, FPOM and SO_4 concentrations (Fig. 4a). The taxon-environmental variables biplot showed some Ephemeroptera species (e.g. *Baetis rhodani*, *Ecdyonurus torrentis*, *Ecdyonurus dispar* and *Rhithrogena semicolorata*), Plecoptera (e.g. *Perlodes microcephala* and *Isoperla grammatica*) and caseless Trichoptera larvae (e.g. *Polycentropus flavomaculatus* and *Hydropsyche pellucidula*) were more abundant in streams from intact and drain-blocked catchments. In contrast, some dipterans (e.g. Simuliidae and Chironomidae), the ephemeropteran *Ephemera danica* and the cased Trichoptera larvae *Sericostoma personatum* were more strongly associated with drained sites with higher SSC, FPOM and SO_4 concentrations (Fig. 4b). A diverse assemblage of Ephemeroptera species was found in the intact and drain-blocked sites, while only *E. danica* was documented in the drained sites (Fig. 4b).

ANOSIM based on macroinvertebrate relative abundance data revealed significant differences in community composition between land management types (R^2 = 0.39; P <0.001), with pair-wise comparisons highlighting drained site macroinvertebrate assemblages were different to those at intact and drain-blocked sites (R^2 = 0.43 and 0.65, respectively; P <0.001 in both cases). Similarly, community composition at the intact and drain-blocked streams was significantly different (R^2 = 0.10; P = 0.032). ANOSIM based on presence-absence data also showed significant differences between land management types (R^2 = 0.36; P <0.001), with pair-wise comparisons showing macroinvertebrate communities were different at drained sites compared with intact and drain-blocked sites (R^2 = 0.43 and 0.59, respectively; P <0.001 in both cases). In contrast to the relative abundance analysis, the intact and drain-blocked streams were not significantly different ($R^2 = 0.07$; $P = 0.105$).

DISCUSSION

To the authors' knowledge, this study has provided the first detailed insight into the spatial and temporal dynamics of stream physicochemical environmental variables and macroinvertebrate communities in blanket peatland river systems influenced by artificial drainage and drain-blocking. This discussion considers reasons for the differences in: (1) physicochemical variables, and; (2) macroinvertebrate communities, between streams influenced by different peatland catchment management strategies. It then compares the results of this UK upland restoration case study with catchment-scale restoration schemes from other parts of the world, to identify commonalities that may be of relevance to future management interventions.

Peatland management effects on stream environmental variables

Artificial drainage of peatlands was linked to changes in several stream physicochemical variables (e.g. increases in SSC and FPOM) allowing hypothesis 1 to be supported. These findings are supported in part by evidence from other studies, where 50-fold (Robinson & Blyth, 1982) and 12-fold (Ahtiainen & Huttunen, 1999) increases in average stream sediment loads have been observed due to peat drainage. The erosion of artificial drains can result in greater quantities of fine particulates being deposited into nearby aquatic systems (Holden, 2006b; Holden *et al.*, 2007) and this study showed significantly higher concentrations of FPOM in sites that were artificially drained when compared with intact sites.

Stream benthic FPOM concentrations recorded from the drain-blocked catchment streams were lower than those documented from artificially drained catchment streams most likely because, even in poorly dammed drains, fine particulate erosion and transport can be markedly lower when compared with unblocked drains (Holden *et al.*, 2007). However, these previous studies investigated FPOM concentrations in the drains as opposed to our study of receiving stream systems. Following drain-blocking, the area of exposed bare peat, and the connectivity of bare areas with the stream, is reduced (Holden *et al.*, 2007). Thus, peat erosion declines and less FPOM reaches the stream and is deposited. Following drain-blocking, excess FPOM in streambed sediments is likely to be removed during high flows at a greater rate than it is re-supplied (e.g. Bilby & Likens, 1979), leading to a significantly lower concentration over time. Further study is necessary to document the rapidity of this likely recovery. However, we recognise that our samples were taken mainly during baseflow conditions and further sampling is required across a range of flows to properly characterise the suspended sediment and FPOM within the fluvial system. Nevertheless, peat streams spend the majority of time at low flow (Holden & Burt, 2003b) and so baseflow conditions are an important component of stream habitat.

Peatland management effects on stream macroinvertebrate communities

The findings of this study supported the second hypothesis that there would be significant differences in stream macroinvertebrate communities between intact, drain-blocked and artificially drained catchments. The third hypothesis, that macroinvertebrate communities in intact and drain-blocked sites would be similar reflecting successful catchment-scale restoration, was also supported by the results. Taxonomic richness was higher in intact and drain-blocked sites compared with the drained sites, and the higher suspended sediment concentrations in the latter could have led to greater deposition on benthic habitats and reduced macroinvertebrate diversity. These inferences are supported by previous work by Vuori & Joensuu (1996) and Vuori *et al.* (1998), who investigated forestry drainage in Finland and observed a reduction in species richness linked to suspended and deposited sediment. Deposited fine sediments can alter substrate composition and suitability (Richards & Bacon, 1994), affect respiration by deposition on respiratory surfaces/appendages/nets (Lemly, 1982), prevent feeding by grazers (Aldridge *et al.*, 1987) and reduce the nutritional value and abundance of periphyton (Vuori *et al.*, 1998). Additionally, reduced diversity and richness could be related to higher levels of benthic FPOM. The usual consequence of increased FPOM is an increase in a few detritivorous species with a decrease in overall richness and diversity (Moss, 1973; Perry & Sheldon, 1986) because excess FPOM can alter the physical environment in a manner similar to fine inorganic sediment.

The low relative abundances of Plecoptera and Trichoptera observed at the drain-blocked sites could be due to prolonged effects of stream ecosystem stressors linked to earlier artificial drainage. Historic land-use can continue to affect stream diversity and communities over time scales spanning decades, thus long-term studies are often required to quantify recovery (e.g. Harding *et al.*, 1998; Foster *et al.*, 2003). In this study, stream biota were collected at sites where drain-blocking had been implemented between three and 11 years prior to sampling, and therefore exact recovery times/rates were not measured. Nonetheless, the low relative abundance of Plecoptera and Trichoptera in drain-blocked sites suggests earlier drainage impacts may continue. In contrast, relative abundance of Ephemeroptera was similar in drain-blocked sites to intact streams. Ephemeroptera are often among the first macroinvertebrates to recolonise stream habitats following disturbance, because adults are relatively strong fliers and they have a propensity for drift as nymphs from any undisturbed upstream populations (Williams, 1980). Recolonisation by drift seems unlikely in these artificially drained peatland streams because drains covered the entire catchment and so all parts of the stream network would likely have been affected. Further research investigating pre-and post-blocking would help to establish the role of dispersal constraints in the temporal sequence of drain-blocked peatland stream recolonisation.

The ordination biplot indicated that several macroinvertebrate species found in the intact sites were also documented in drain-blocked sites (e.g. *R. semicolorata*, *B. rhodani*, *E. dispar*, *P. microcephala*, *I. grammatica*, *H. pellucidula* and *Hydroptila* spp.) but not in artificially drained sites. These taxonomic differences were associated with higher concentrations of suspended sediment and benthic FPOM. Most species within the families Baetidae and Heptageniidae are algal scrapers/grazers, and so their feeding could be quickly impaired on sediment smothered periphyton (Larsen & Ormerod, 2010). Hydropsychiids (Trichoptera) are sensitive to increases in sediment loads as their retreats and nets can become embedded under excess particles, thus reducing oxygen levels in interstitial spaces (Runde & Hellenthal, 2000). Nets can also clog, rip or be buried, resulting in decreased food acquisition (Strand & Merritt, 1997), interference with respiration (Lemly, 1982) and increased energy expenditure due to net cleaning (Runde & Hellenthal, 2000). Furthermore, many predatory Plecoptera such as Perlidae and Perlodidae are normally found living in coarse sediments. Fine sediment can limit oxygen availability by reducing flow velocities in clogged interstices, reduce interstitial water exchange and constrict the movement of these invertebrates in the substrata (e.g. Beauger *et al.*, 2006; Bo *et al.*, 2007).

Higher abundance of the mayfly *E. danica* and the dipterans Chironomidae and Simuliidae was recorded in drained peatland streams compared with intact and drain-blocked sites. Previous work by Beisel *et al.* (2000) in northeast France found higher abundance of *E. danica* in habitats with excess fine sediment, while Vuori & Joensuu (1996) found Finnish forest drainage encouraged increased Chironomidae and Simuliidae abundance. Simuliidae larvae filter FPOM effectively from the stream water column (e.g. Vuori & Joensuu, 1996; Ciborowski *et al.*, 1997) which would explain their increased abundance in streams from drained peatland catchments characterised by higher suspended and deposited organic particulates.

The ANOSIM results further supported the finding that there were significant community and taxonomic differences between the artificially drained catchment streams and the intact and drain-blocked streams. The analysis based on Jaccard's co-efficient of similarity indicated that taxa presence-absence was similar in drain-blocked and intact streams, whereas there were significant differences between these peatland management types in the analysis based upon BC distances. These results suggest that drain-blocked sites have similar taxonomic composition but that there are differences in relative abundance among constituent taxa. This could indicate that drain-blocked streams have not completely recovered to an 'intact' state, or it may represent the development of an alternative endpoint (e.g. Bradshaw, 1996; Ormerod, 2003) in the restored peatland streams.

Catchment-scale restoration effects on stream ecosystems

In many regions of the world, stream ecosystem services and biodiversity have been compromised due to catchment degradation (Harding *et al.*, 1998; Paul & Meyer, 2001; Allan, 2004). In response, catchment-scale rehabilitation programmes have become more common (see Hillman & Brierley, 2005) but there is growing recognition that many schemes lack adequate pre- and post-restoration monitoring, a problem similar to that for reach-scale river rehabilitation schemes (e.g. Bernhardt *et al.*, 2005; Palmer *et al.*, 2005; Woolsey *et al.*, 2007). This study of peatland catchment remediation and stream ecosystem response is one of only a few studies to consider stream ecosystem responses to catchment-scale restoration. Stream ecosystems of catchments where there was a recent drain-blocking intervention appeared to have improved water quality, thus sustaining broadly similar macroinvertebrate communities to those in catchments with no history of peat drainage. These findings illustrate that such intervention may promote positive effects for in-stream biodiversity. Importantly though, this progress is undoubtedly being missed by high profile and costly monitoring schemes, such as those tracking attempts to improve ecological status under the EU Water Framework Directive, because catchments $<10\text{km}^2$ are rarely appraised (Logan & Furse, 2002). More detailed consideration of small headwater systems that have been restored may be fruitful with regards to improving estimates of the number of watercourses in ‘good’ or ‘high’ ecological condition.

Field Code Changed

Conservation of stream biodiversity has received increasing recent attention as human modification and disturbance of ecosystems increase (Harding *et al.*, 1998; Sponseller *et al.*, 2008). Although, rapid recovery of biotic communities following short-term catastrophic disturbances (e.g. short duration pulse disturbances) can often happen due to immigration from nearby unimpacted streams (e.g. Doeg & Koehn, 1994), impacts of sustained catchment-scale disturbances may profoundly affect all streams in a catchment, eliminating local refugia and meaning that recovery following any subsequent restoration may take a long time due to dispersal constraints (Harding *et al.*, 1998). Although artificial drains studied here have only been blocked for a short period of time (3 - 11 years), recolonisation of streams appears to have been relatively rapid, perhaps as a result of the close proximity to intact peatland streams (cf. Mackay, 1992). To ensure the quickest possible recovery of peatland stream ecosystems following drain-blocking, land managers should consider the importance of hydrological connectivity and proximity to potential sources of recolonisers when planning future restoration schemes.

It is acknowledged that this study examined the spatiotemporal interactions between physiochemical habitat variables and macroinvertebrate communities over only a short period of time using a space-for-time substitution approach. This design was necessitated by the lack of any pre- or post-restoration stream ecosystem data for drain-blocked catchments, a common problem afflicting river rehabilitation schemes more generally (Bernhardt *et al.*, 2005). In order to measure the efficacy of drain-blocking and in-stream ecological recovery, detailed pre- and post-blocking stream ecosystem monitoring is sorely needed. The importance of, and need for, long-term monitoring to investigate impacts of past river system alterations cannot be understated (Palmer *et al.*, 2005); such information would help deduce the level of disturbance, post-restoration recovery times, and re-assembly mechanisms of biotic communities following drain-blocking. To achieve such an aim there is a need to improve knowledge exchange between upland stakeholders/government agencies managers and freshwater scientists (e.g. Brown *et al.*, 2010b), particularly when drain-blocking schemes are at the planning stage. Overcoming these issues will be necessary to establish true baselines against which to judge the response of stream ecosystems to future peatland restoration.

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Table 1. Summary information for the nine stream study sites

Stream name	Management type	Soil types	Catchment area (km^2) [1]	Grid reference
Moss Burn	Intact	Blanket peat	2.15	54°41’1’’N 2°27’0’’W
Snaizeholme Beck	Intact	Blanket peat, stagnogley, stagnohumic gley, humic gley, fine loam, alluvial gley	2.23	54°15’7’’N 2°16’1’’W
Unnamed 2nd order tributary of the River Tees	Intact	Blanket peat, alluvial floodplain	1.12	54°41’8’’N 2°26’8’’W
Crook Burn	Drain-blocked	Blanket peat, alluvial floodplain	2.98	54°42’2’’N 2°20’5’’W
South Tyne	Drain-blocked	Blanket peat, alluvial floodplain	3.52	54°43’6’’N 2°22’8’’W
Blea Gill	Drain-blocked	Blanket peat, humic gley, humic loam,	1.61	54°14’2’’N 2°13’9’’W
Cam Beck	Drained	Blanket peat, stagnohumic gley, seasonally waterlogged loam with peaty surface	2.91	54°13’9’’N 2°16’1’’
Old Water	Drained	Blanket peat, stagnohumic gley, seasonally waterlogged loam with peaty surface	1.29	54°52’4’’N 2°37’9’’W
Unnamed 2nd order tributary of Killhope Burn	Drained	Blanket peat, slowly permeable loam over clay, stagnohumic gley	1.08	54°46’7’’N 2°17’2’’W

[1] Catchment area was measured using the hydrology tool in ArcGIS

Table 2. Descriptive statistics and RM-ANOVA results for the stream environmental variables

	Cl (mg l^{-1})	**NO_3 (mg l^{-1})**	**SO_4 (mg l^{-1})**	**Al (mg l^{-1})**	**Fe (mg l^{-1})**	**DOC (mg l^{-1})**	**DO (mg l^{-1})**	**EC (µS cm^{-1})**	**pH**	**SSC (mg l^{-1})**	**D_{50} (cm)**	**CPOM (mg l^{-1})**	**FPOM (mg l^{-1})**	**POM (mg l^{-1})**	**Water temperature (°C)**	**Discharge (m^3s^{-1})**
All streams																
Mean	8.33	0.56	3.70	0.05	0.47	13.28	11.3	116.64	5.43	6.95	3.9	0.50	0.72	1.21	9.0	0.09
Min	0.11	< 0.01	0.53	< 0.01	< 0.01	0.09	5.8	18.00	4.29	0.20	1.0	0.02	0.03	0.07	0.6	0.01
Max	53.13	1.69	12.94	0.31	1.84	67.31	19.3	390.00	8.65	28.60	6.9	2.31	6.75	7.62	18.5	0.47
Intact																
Mean	3.75	0.36	2.29	0.05	0.49	14.01	11.1	76.72	4.99	4.61	5.0	0.31	0.41	0.70	8.8	0.08
Min	0.11	< 0.01	0.53	< 0.01	0.06	0.09	5.8	18.00	4.29	1.00	4.0	0.02	0.04	0.07	0.6	0.01
Max	9.35	1.26	5.65	0.18	1.33	67.31	19.3	191.40	8.65	12.80	6.9	1.52	3.48	4.07	18.5	0.25
Drain-blocked																
Mean	12.19	0.66	4.37	0.04	0.36	13.65	11.9	148.06	7.67	3.57	5.4	0.40	0.12	0.51	9.9	0.16
Min	0.84	< 0.01	0.83	< 0.01	< 0.01	1.92	6.20	23.25	6.98	0.20	5.3	0.02	0.03	0.07	1.1	0.01
Max	53.13	1.34	12.94	0.12	0.99	55.15	19.0	390.00	8.53	16.60	5.5	2.31	0.28	2.54	17.5	0.47
Artificially drained																
Mean	9.04	0.66	4.45	0.07	0.56	12.19	11.0	120.05	5.89	12.65	1.2	0.77	1.63	2.40	8.4	0.04
Min	2.27	< 0.01	2.22	< 0.01	< 0.01	1.76	6.3	30.70	4.77	0.80	1.0	0.02	0.18	0.31	0.9	0.01
Max	27.19	1.69	8.62	0.31	1.84	38.31	19.2	248.00	7.78	28.60	1.5	2.11	6.75	7.62	14.0	0.16
Stream ($F_{6,44}$)	F = 3.62	F = 8.96	F = 10.23	F = 2.38	F = 1.33	F = 3.44	F = 0.74	F = 6.35	F = 15.42	F = 5.17	No	F = 4.37	F = 11.87	F = 6.50	F =1.45	F = 4.61
	P = 0.011	P < 0.001	P < 0.001	P = 0.081	P = 0.283	P = 0.014	P = 0.622	P = 0.001	P < 0.001	P = 0.002	replicates	P = 0.004	P < 0.001	P < 0.001	P = 0.239	P = 0.004
Season ($F_{4,44}$)	F =5.18	F = 54.51	F = 16.46	F = 6.74	F = 3.75	F = 2.34	F = 18.41	F = 6.10	F = 4.83	F = 2.21	No	F = 1.07	F = 5.09	F = 2.93	F = 21.45	F = 5.02
	P = 0.004	P < 0.001	P = < 0.001	P = 0.002	P = 0.017	P = 0.084	P < 0.001	P = 0.002	P = 0.005	P = 0.098	replicates	P = 0.395	P = 0.004	P = 0.042	P < 0.001	P = 0.005
Land management ($F_{2,44}$)	F =2.12	F = 1.69	F = 2.79	F = 0.15	F = 1.10	F = 0.18	F = 1.12	F = 0.64	F = 1.00	F = 5.73	F = 47.05	F = 0.28	F = 5.23	F = 2.75	F = 0.94	F = 1.70
	P = 0.201	P = 0.261	P = 0.139	P = 0.868	P = 0.392	P = 0.836	P = 0.384	P = 0.560	P = 0.423	P = 0.041	P < 0.001	P = 0.768	P = 0.048	P = 0.142	P = 0.443	P = 0.260
Season*Land management	F =0.79	F = 4.20	F = 2.19	F = 1.19	F = 1.78	F = 1.43	F = 0.23	F = 1.13	F = 0.52	F = 1.65	No	F = 1.07	F = 1.89	F = 1.03	F = 0.91	F = 0.42
($F_{8,44}$)	P = 0.614	P = 0.003	P = 0.066	P = 0.366	P = 0.131	P = 0.236	P = 0.982	P = 0.384	P = 0.829	P = 0.164	replicates	P = 0.420	P = 0.110	P = 0.439	P = 0.522	P = 0.896

Table 3. Loading scores, Eigenvalues and % variance explained for the five principle components produced from the environmental variables dataset. Values greater than 0.5 or less than -0.5 are highlighted in bold font to aid interpretation

	Principle component				
Variable	**1**	**2**	**3**	**4**	**5**
Cl	0.284	-0.141	0.102	**0.577**	0.392
NO_3	**0.795**	-0.310	-0.251	0.165	-0.017
SO_4	0.365	-0.451	**0.535**	0.023	-0.397
EC	**0.529**	-0.416	0.377	0.089	0.283
Water temperature	-0.338	-0.080	**0.822**	-0.262	0.005
pH	0.293	**-0.636**	0.404	-0.134	0.311
DO	0.308	-0.180	**-0.723**	0.338	0.194
SSC	0.453	0.409	0.336	0.381	-0.076
DOC	-0.377	0.147	-0.061	0.379	-0.062
CPOM	0.422	**0.526**	0.064	-0.342	0.399
FPOM	**0.641**	**0.534**	-0.046	-0.225	-0.130
POM	**0.671**	**0.618**	-0.014	-0.305	0.040
Q	-0.131	-0.146	**-0.580**	-0.494	0.053
Al	**-0.625**	0.484	0.071	0.322	0.105
Fe	-0.343	**0.524**	0.274	-0.028	**0.520**
D_{50}	-0.341	**-0.726**	-0.187	-0.274	0.301
Eigenvalue	3.43	3.13	2.44	1.54	1.09
% variance explained	21.4	19.5	15.2	6.9	6.8

Table 4. Descriptive statistics and RM-ANOVA results for the macroinvertebrate community metrics

	Log_{10} (total abundance +1) (per m^2)	**Richness**	**Simpson's Diversity (1/*S*)**	**Dominance (*D*)**
All streams				
Mean	3.38	25	5.01	41.4
Min	2.54	10	1.54	18.1
Max	4.11	42	11.05	79.9
Intact				
Mean	3.39	30	6.10	37.7
Min	2.99	16	1.74	18.1
Max	3.66	41	11.05	75.3
Drain-blocked				
Mean	3.45	29	5.43	38.5
Min	2.98	15	1.54	19.1
Max	3.86	42	9.15	79.9
Artificially drained				
Mean	3.30	17	3.59	47.3
Min	2.54	10	1.60	26.5
Max	4.11	39	6.88	78.0
Stream ($F_{6,44}$)	$F = 6.67$ $P < 0.001$	$F = 9.92$ $P < 0.001$	$F = 7.34$ $P < 0.001$	$F = 6.78$ $P < 0.001$
Season ($F_{4,44}$)	$F = 4.61$ $P = 0.007$	$F = 3.42$ $P = 0.024$	$F = 1.17$ $P = 0.351$	$F = 1.40$ $P = 0.264$
Land management($F_{2,44}$)	$F = 0.42$ $P = 0.676$	$F = 5.03$ $P = 0.05$	$F = 1.11$ $P = 0.390$	$F = 0.71$ $P = 0.530$
Season*Land management ($F_{8,44}$)	$F = 1.15$ $P = 0.365$	$F = 1.66$ $P = 0.159$	$F = 2.02$ $P = 0.088$	$F = 2.01$ $P = 0.089$

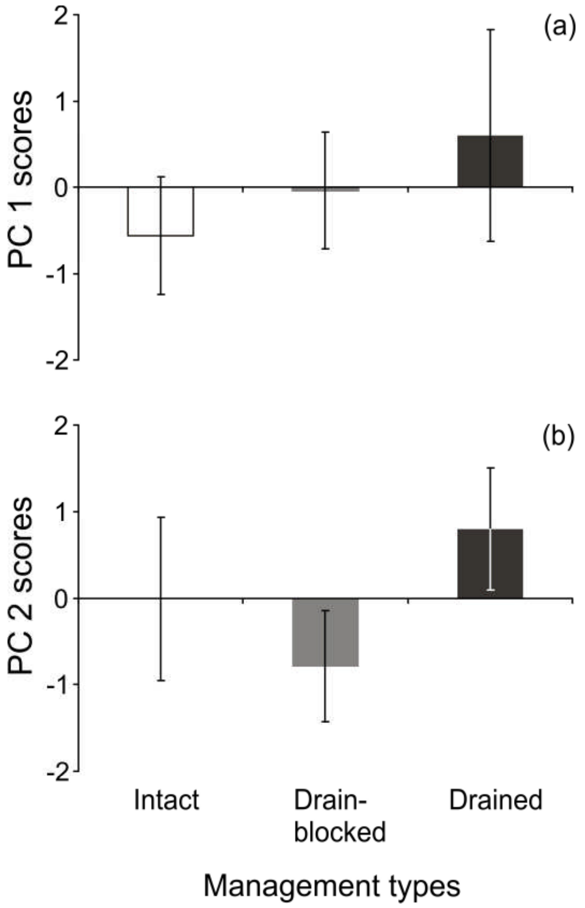

Fig. 1. PC scores for (a) axis 1 and (b) axis 2, for the three peatland management types [Error bars show ±1SD from the mean]. See Table 1 for individual variable loadings, Eigenvalues and % variance of each PC

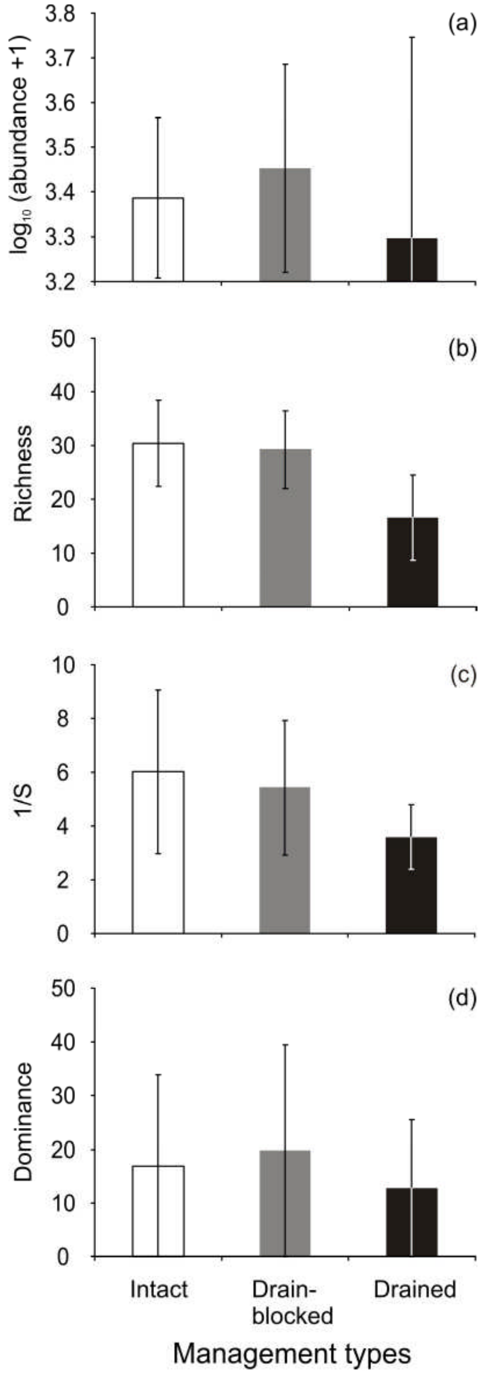

Fig. 2. Effects of management for (a) $\log_{10}$ (abundance +1); (b) Richness; (c) 1/S; and (d) Dominance. [Error bars show ±1SD from the mean]

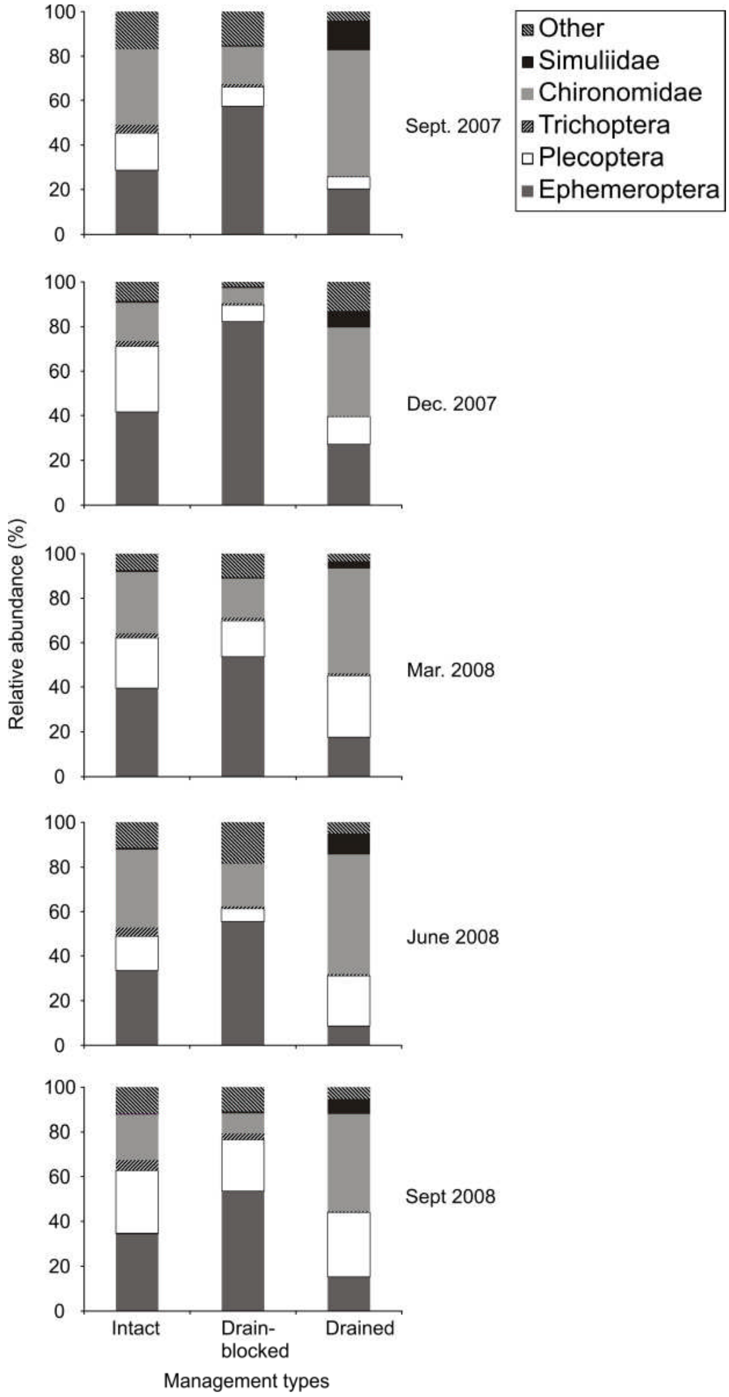

Fig. 3. Seasonal changes in the mean relative abundance of Ephemeroptera, Plecoptera, Trichoptera, Chironomidae, Simuliidae and Other taxa in relation to management types

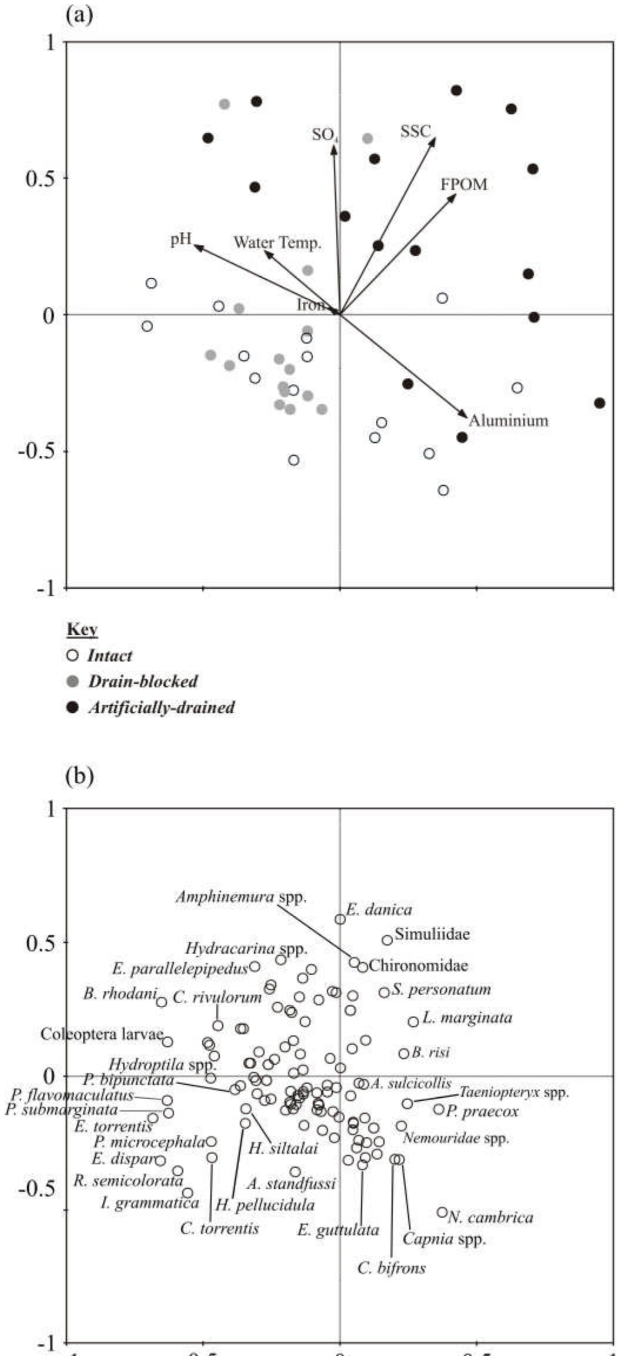

Fig. 4. Redundancy Analysis (RDA) ordination diagrams of **(a)** management types and environmental variables, and; **(b)** macroinvertebrate taxa (with selected taxa highlighted). Species abbreviations follow those provided in the main text