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ROTATIONAL VEGETATION BURNING EFFECTS ON PEATLAND STREAM ECOSYSTEMS

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Running head: Burning impacts on peatland streams

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Summary

1. Rotational vegetation burning in peatlands is undertaken predominantly to increase habitat suitability and food availability for red grouse (*Lagopus lagopus*). Red grouse shooting contributes to the upland economy and is seen as a traditional leisure activity. However, there is concern that burning can have detrimental effects on peatland terrestrial and freshwater ecosystems.

2. This study examined spatial and seasonal dynamics of stream physicochemistry and benthic macroinvertebrates from peatland sites that are managed via rotational vegetation burning and compared these with intact sites with no recent history of burning.

3. Streams draining burned catchments were characterised by higher fine benthic particulate organic matter (FPOM), suspended sediment concentration (SSC), aluminium, iron and dissolved organic carbon than unburnt intact catchments. Anion concentrations were higher in intact catchments.

4. There were significant differences in benthic macroinvertebrate richness, diversity and dominance, and community composition and functional feeding groups between burned and intact catchments, suggesting that land management had an effect on aquatic ecosystems.

5. Higher SSC and FPOM in burned catchments were associated with lower abundance of some mayflies, stoneflies and caddis-flies, and elevated abundance of some Diptera (Chironomidae and Simuliidae) larvae.

6. *Synthesis and applications.* This study suggests that some aspects of peatland stream ecosystems are altered in catchments with rotational vegetation burning. Currently, there is much emphasis on the effects of rotational burning on peat carbon stores, but this study is the first to document the impacts on stream biota. Agencies

with a remit covering upland freshwater ecosystem management might need to consider ways of reducing the extent of rotational vegetation burning to prevent effects on lotic ecosystems, and monitor whether macroinvertebrate assemblages subsequently shift back to a status similar to those in intact peatland streams. Fire occurs commonly on peatlands throughout the world, and our results suggest that trade-offs are needed to satisfy both economic and ecological facets of the combined social-ecological systems in such areas, especially where fire is implemented as a management tool.

Introduction

Controlled burning is used worldwide for vegetation management but there are serious concerns about its environmental implications (Freckleton 2004). In the UK, fire has been used to control upland vegetation since 7700–6300 BC (Goodfellow 1998) but over the last 150 years many upland landscapes have been subjected to controlled rotational burning regimes (Davies 2008). Rotational burning usually occurs on patches of approximately 400 m^2 and burning cycles vary from 8 to 25 years (Davies 2008; Grant *et al.* 2012) depending on productivity, habitat type, grazing level, traditional burning schedules, or government body instigated management prescriptions. Thus, the catchment of an individual stream will have dozens of burning patches of different ages. Typically, burning will take place within the catchment most years, but each year, a different set of patches will be burned so that on average an individual patch will be burned once every 8 to 25 years. Across burned peatland there will therefore be patches which have been very recently burned (i.e. within the last 12 months) and those which have not been burned for many years thereby creating a mosaic. Rotational burning on peatlands is practised to remove ageing dwarf shrubs (e.g. *Molinia caerulea* and *Calluna vulgaris*) and allow regeneration of younger, palatable shoots. This is deemed to be suitable for increasing red grouse populations (Harris *et al.* 2011; Worrall *et al.* 2011). Annually, in England and Wales alone, grouse shooting is worth more than £10 million to land owners (Ward *et al.* 2007) and contributes some £192 million to the UK upland economy indirectly (P.A.C.E.C. 2006).

Open upland moors consist of a variety of vegetation and soil types including deep blanket bog, wet heath and dry heath. In England and Wales there is a Code (Defra

2007) that anyone burning vegetation is expected to follow. This burning code includes a presumption against burning on blanket bog. Undoubtedly, however, a large amount of burning takes place on blanket bog, often with permission of regulatory authorities. Previous work from Yallop *et al.* (2006a) has suggested that there was an increase in approximately 20% of upland heath and bog that had been burnt recently, implying an increase in rotation frequency. Defra (2010) estimated that 18% of UK peatlands have been subjected to managed burning, which is approximately 3150 km^2. Although there are large economic benefits with sport shooting (see report by P.A.C.E.C. 2006), more research is needed to understand fully the environmental impacts of rotational vegetation burning (Sutherland *et al*. 2006).

A conservation status assessment carried out by English Nature (2003) reported that 24% of the area of upland Sites of Special Scientific Interest (SSSI) in England was in an unfavourable condition due to rotational burning. Rotational burning can cause alterations to the terrestrial environment (e.g. vegetation, soil structural, physical and chemical alterations, Maltby *et al.* 1990; Laubhan 1995), increase sediment erosion and transfer to stream systems (e.g. Imeson 1971; Arnett 1980), increase saturation-excess overland flow through higher water tables as there is less plant transpiration (e.g. Clay *et al.* 2009a) and perhaps induce changes to stream chemistry (e.g. DOC, Mitchell & McDonald 1992; Clay *et al.* 2009; Clay *et al.* 2010). While there are multiple drivers of increased water discolouration (associated with DOC production) in peatland streams over the past 40 years (Worrall *et al.* 2004; Evans *et al.* 2006a; Chapman *et al.* 2010), there is evidence to suggest that prescribed burning is an additional factor, although further work is required to establish causal mechanisms (Holden *et al.* 2012).

Despite the recent increase in attention on the effects of rotational vegetation burning on aquatic systems, there remains a lack of knowledge about impacts on stream biota (Ramchunder *et al.* 2009; Worrall *et al.* 2010). Ramchunder *et al.* (2012) documented that increases in fine particulate organic matter (FPOM) and suspended sediment concentrations (SSC) following peatland drainage were associated with decreased abundance of some mayfly and stonefly species but increases in *Ephemera danica* (Ephemeroptera), Chironomidae and Simuliidae abundances. Comparable responses of the stream ecosystem can be hypothesised for systems affected by vegetation burning because the alterations caused to the terrestrial environment could potentially deliver elevated sediment loads to nearby water courses (Ramchunder *et al.* 2009). Similar effects have been observed in stream ecosystems affected by forest fires (e.g. Minshall *et al.* 1997; Vieira *et al.* 2004).

Macroinvertebrates constitute an important part of animal production within freshwaters and are integral to the structure and functioning of these ecosystems (Allan & Castillo 2007). The categorisation of stream macroinvertebrates into functional feeding groups (FFG) is a reliable tool for assessing the dynamics of lotic communities (Allan & Castillo 2007). Post-wildfire studies in US forests have shown shredder biomass decreases due to the loss of riparian vegetation inputs, whilst algal biomass increases following the opening of the canopy and nutrient release led to more scrapers (Minshall 2003). To date, there have been no studies investigating macroinvertebrate community responses following rotational vegetation burning on UK peatland ecosystems or elsewhere.

This study investigated stream macroinvertebrate communities from ten headwater peatland catchments (five intact, five burned). The aim was to provide a detailed evaluation of how controlled vegetation burning on peatland influences stream macroinvertebrate communities. Based on knowledge from previous studies of peatland drainage and burning, it was hypothesised that (H_1) streams in burned catchments would have higher SSC and benthic FPOM compared with intact catchments (Maltby *et al.* 1990; Tucker 2003). Previous work by Ramchunder *et al.* (2012) suggested that increases in FPOM and SSC in artificially drained catchments altered individual species abundance but had no discernible effect on community richness, Simpson's diversity, dominance and total abundance. Therefore, (H_2) similar biological responses were expected in burned catchments. However, (H_3) alterations in the stream environment due to burning were expected to result in macroinvertebrate communities containing higher abundance of taxa associated with in-stream fine sediment deposition and benthic particulate organic matter, with increases in filtering-collectors (linked to FPOM supply from burned catchments), but negative effects on herbivore and predator abundance (e.g. Mihuc & Minshall 1995; Vieira *et al.* 2004). The findings of this study are considered subsequently in the context of more general literature on rotational vegetation burning effects on peatland stream ecosystems, and some implications for upland policy makers and landowners are discussed.

Materials and Methods

Study areas

This study comprised of: (a) a seasonal study of three burned sites and three unburned sites (hereafter 3v3 survey) located in Upper Teesdale, Wensleydale and Geltsdale in northern England, and (b) a broader, single occasion survey, comparing five burned sites and five unburned sites (hereafter 5v5 survey), with the datasets from (a) augmented by sampling at additional sites in the north Peak District (Table 1).

Potential study catchments were identified as those having second order streams based on 1:25000 Ordnance Survey maps, and candidate burned sites were identified from aerial photographs. Sites were selected randomly with no confounding effects of recent wildfire, mining, major erosion or forest cover. At each catchment outlet, a representative 15-m reach was selected randomly for study with subsequent sampling undertaken in riffle areas of those reaches.

All sites had blanket peat cover, with vegetation dominated by *Eriophorum* spp. and *C. vulgaris* and there was *Sphagnum* spp. cover at all sites but this was less abundant in the Peak District. Although data were not available for all sites, mean annual precipitation of 2012 mm (1951–1980; 1991–2006) occurs at Moor House, Teesdale (Holden & Rose 2011). Mean annual air temperature at Moor House is 5.3°C (1931–2006; Holden & Rose, 2011). Annual rainfall varies considerably across the Peak District, ranging from 1000–1584 mm (Evans *et al.* 2006b; Shotbolt *et al.* 2008). The climate is cool with mean monthly temperatures ranging from 2–14°C (Evans 2005).

Field sampling

For the 3v3 survey, streams were sampled seasonally across 3–4 days per quarter (2007: September 11–13, December 19–21; 2008: March 4–7, June 10–13, September 16–18). The 5v5 survey was concurrent with the September 2008 survey. During each site visit, 16 stream environmental variables were measured to provide contextual habitat information (Table 2). Water temperature, pH and electrical conductivity (EC) were measured using MP120 and MP126 handheld probes (Mettler-Toledo Ltd, Leicester, UK). Dissolved oxygen (DO) concentration was measured using a HI9412 probe (Hanna Instruments Ltd, Bedfordshire, UK). Additionally, 120 mL of stream water was passed through a 0.45-μm filter and subsequently analysed in the laboratory for chloride (Cl), sulphate (SO_4) and nitrate (NO_3), dissolved organic carbon (DOC), aluminium (Al) and iron (Fe). A further 500 mL of unfiltered stream water was collected for the determination of SSC by filtration. Streambed sediments were characterised by sampling 100 clasts randomly, measuring *b*-axis lengths and calculating the median grain size (D_{50}). To provide a relative indication of flow differences between sites and over time, stream discharge (Q) was measured at the time of sampling using an open channel flow meter (Valeport, Devon, UK) and the velocity–area method.

Five replicate benthic macroinvertebrate samples were collected randomly on each site visit from riffle habitats using a modified 0.05-m^2 Surber sampler (250 μm mesh) and were preserved immediately in 70% ethanol. After sorting in the laboratory, macroinvertebrates were identified to species level (where possible) under a light microscope (x40 magnification) but some taxa were identified to higher levels (e.g. Diptera [Family/Genus], Oligochaeta [Class]) using standard keys (see Pawley *et al.* 2011 and references therein). Particulate organic matter (POM) retained in each sample was sorted into fine (<1mm; FPOM) and coarse fractions (>1mm; CPOM), then ashed to determine ash-free dry mass.

Data analysis

Repeated Measures ANOVA (season as repeated measure) with Bonferroni correction was used to ascertain if there were significant differences in stream environmental variables as a function of land management. Land management was fixed and season was random. Sites were selected randomly as a 'representative reach' for each treatment type and because the focus of the study was on effects of burning, inter-site comparisons were not considered in detail. One-way ANOVA was used for the single occasion 5v5 survey to determine if there were differences in stream environmental variables as a function of management type.

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 FFGs assigned following Hynes (1977), Elliott *et al.* (1988), Edington and Hildrew (1995) and Wallace *et al.* (2003); (4) 1/Simpson's diversity index (*1/S*): (Simpson 1949) and (5) taxonomic dominance (*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 total abundance.

RM-ANOVA and one-way ANOVA were repeated for the macroinvertebrate community metrics for the 3v3 and 5v5 survey, respectively, using the same methods outlined above for environmental variables. All environmental and macroinvertebrate data sets were tested for normality and, where necessary, $1\log_{10}$, arcsin or square root transformed to improve normality and homogeneity of variance prior to statistical tests. All tests were undertaken in SPSS v17.0 or Minitab v15.0 and considered significant where $P<0.05$. Mauchly's Test of Sphericity was not violated throughout the RM-ANOVA analyses.

Taxon–habitat relationships were assessed for both the 3v3 and 5v5 surveys separately using redundancy analysis (RDA) in CANOCO v4.5 (Lepš & Šmilauer 2003). Invertebrate abundance data were Hellinger-transformed following Legendre & Gallagher (2001). Forward selection was used to determine which of the stream environmental variables accounted for a significant proportion of the species variance. An initial RDA on the 3v3 survey included a dummy variable 'Time' (no. days from start of sampling) to determine whether there were significant seasonal dynamics within the stream macroinvertebrate communities. Following this, a partial RDA (*p*RDA) was carried out to remove the variance accounted by Time, providing a better indication of the land management and between stream components of the data set (Borcard *et al.* 1992). A standard RDA was conducted on the 5v5 survey as samples were collected only in September 2008.

One-way analysis of similarity (ANOSIM) tested the null hypothesis that differences in stream macroinvertebrate taxa abundance between burned and unburned peatlands were not different to those within the two land management types. ANOSIM was not undertaken to test for seasonal effects in the 3v3 survey owing to the small number of replicates per quarterly sample collection, and because spatial dynamics (linked to management type) were the central focus of this study. ANOSIM was undertaken using both the Bray-Curtis (BC) dissimilarity index (based on taxa relative abundance) and the Jaccard's coefficient of similarity (based on taxa presence-absence), with 10,000 permutations and Bonferroni corrections using PAST v2.05 (Hammer *et al.* 2001).

Results

3v3 survey

Stream environmental variables

Mean concentrations of Cl, NO_3, Al, pH, SSC and DOC, benthic FPOM and POM, and water temperature were all higher in the burned streams. Mean SO_4, EC, CPOM, DO, Q, Fe and D_{50} were lower in burned streams (Table 2). The RM-ANOVA showed significant differences in Cl, SO_4, NO_3, Al, Fe, DOC, SSC, D_{50}, CPOM, FPOM and POM between land management (Table 2).

Macroinvertebrate community structure

Mean total abundance, community richness and *1/S* were higher in the intact sites while mean dominance was higher in the burned sites. The lowest observed richness was documented at New Water (burned), whereas the lowest total abundance, *1/S* and dominance were documented across the intact sites (Table 3; Fig. 1). RM-ANOVA showed significant differences between peatland management types and macroinvertebrate community richness and Simpson's diversity (Table 3). The relative abundances of Ephemeroptera, Trichoptera and Other were significantly higher at intact sites, while Chironomidae relative abundance was significantly higher at burned sites (Table 3). Except for March 2008, relative abundance of 'Other' (taxa composed of adult and larva Coleoptera, molluscs and Megaloptera) was often higher in the intact sites, while relative abundance of Chironomidae was consistently higher in the burned sites (Fig. 2). Significantly higher abundance of herbivores and predators were observed in the intact sites (Table 3).

Macroinvertebrate species–environment relationships

Axes 1 and 2 of the initial 3v3 RDA accounted for a total of 19.9% and 6.8% of the total variance, respectively. Taxa-environment correlations were 0.746 and 0.878 for axis 1 and 2 respectively. Time accounted for 10.1% of the species variance; therefore, a partial RDA (*p*RDA) was undertaken to extract the variance accounted by Time. Axes 1 and 2 of the *p*RDA analysis accounted for a total of 19.1% and 3.8% of the total variance with taxa-environment correlations for axes 1 and 2 being 0.746 and 0.721 respectively. Forward selection showed EC, FPOM and SSC were associated with a significant proportion of the variance. The analysis showed that the intact sites were associated with lower SSC and FPOM and higher EC (Fig. 4a).

The taxa–environmental variables biplot showed some Ephemeroptera species (e.g. *Baetis rhodani*, *Ecdyonurus torrentis*, *Ecdyonurus dispar* and *Rhithrogena semicolorata*), Plecoptera (e.g. *Perla bipunctata* and *Isoperla grammatica*), caseless Trichoptera larvae (e.g. *Rhyacophila septentrionis*, *Polycentropus flavomaculatus* and *Hydropsyche pellucidula*) were associated more with intact sites. Alternatively, the dipterans (e.g. Simuliidae and Chironomidae), the Ephemeropteran, *Ephemera danica* and Plecoptera (e.g. *Protonemura meyeri*, *Amphinemura sulcicollis* and *Leuctra inermis*), were more common in burned sites (Fig. 4b). A diverse assemblage of Ephemeroptera species was found in the intact sites while only *E. danica* and *Siphlonurus lacustris* were documented in the burned sites (Fig. 4b). ANOSIM based on macroinvertebrate relative abundance data from the 3v3 and 5v5 survey showed significant differences in community composition between land management types (R^2=0.31; P<0.001 and R^2=0.62; P<0.05, respectively), as did the analysis based on presence/absence data (R^2=0.19; P<0.005 and R^2=0.528; P<0.05, respectively).

5v5 survey

Stream environmental variables

The burned sites, on average, had higher Cl, NO_3, SO_4, Al, Fe, DOC, pH, SSC, FPOM and CPOM. Alternatively, EC, D_{50}, POM and water temperature were on average higher in the intact sites. ANOVA showed significant differences in Al, DOC, SSC and D_{50} between burned and intact sites (Table 2).

Macroinvertebrate community structure

Intact sites had higher mean abundance, richness, dominance and *1/S* compared with the burned sites. In contrast, average dominance was higher in the burned sites whilst abundance in the burned sites was similar to the intact sites (Table 3; Fig. 5). ANOVA showed significant differences in richness, *1/S* and dominance between land management (Table 3).

Trichoptera and Other relative abundances were significantly higher in the intact sites compared with the burned sites (Table 3). In contrast, Chironomidae relative abundance was significantly higher in the burned sites (Table 3 and Fig. 6). Higher abundance of shredders, herbivores and predators were observed in the intact sites. Burned sites had a greater abundance of gathering-collectors and filtering-collectors (Table 3 and Fig 3b). ANOVA showed significant differences in herbivore abundance between land management types (Table 3).

Macroinvertebrate species–environment relationships

Axes 1 and 2 of the RDA accounted for a total of 39.5% and 8.2% of the total variance respectively. Taxa-environment correlations were 0.964 and 0.817 for axis 1 and 2, respectively. The analysis showed that the burned sites (except Ashop Clough) were associated with higher DOC concentrations and lower EC (Fig. 7a).

The taxa-environmental variables biplot of the sites showed a division between the majority of burned and unburned streams in terms of community composition. Plecoptera (e.g. *P. bipunctata*, *Perlodes microcephala*), Ephemeroptera (e.g. *R. semicolorata*, *E. torrentis* and *B. rhodani*) and Trichoptera (e.g. *H. pellucidula*, *Hydroptila* spp. and *Rhyacophila dorsalis*) were associated with unburned sites. Chironomidae, the stoneflies within the genus, *Amphinemura* and the cased-caddis *Drusus annulatus* and *Limnephilidae* spp. were associated more with the burned sites (Fig. 7b).

Discussion

Rotational vegetation burning effects on stream environmental variables

This study has provided a detailed insight into the spatial and seasonal dynamics of stream environmental variables and macroinvertebrate communities in UK upland rivers influenced by rotational vegetation burning. Both the 3v3 and the 5v5 surveys showed burning was linked to changes in several stream environmental variables (e.g. increases in SSC, FPOM, Al, SO_4, NO_3, DOC and smaller D_{50}) allowing H_1 to be upheld. These findings are supported in part by evidence from other studies, where the removal of the vegetation cover and litter layer by fire, coupled with wind and rain can increase vulnerability of the soil to physical erosion, resulting in higher sediment yields being deposited into streams (Tallis 1987; Tucker 2003). Charred peat after burning can also form loose crusts which are broken down easily and washed into streams in overland flow (Tucker 2003).

Higher concentrations of SO_4 were found in burned catchment streams compared with the intact sites. Burning removes ‘blocks’ of vegetation, and thus the exposed peat can be subjected to enhanced drying and oxidation (Maltby *et al.* 1990; Tucker 2003). The oxidation of reduced sulphur stored in the peat and the mineralisation of organic sulphur to dissociated sulphuric acid may explain the observed higher levels of SO_4 in this study (e.g. Bottrell *et al.* 2004; Clark *et al.* 2005). These findings of increased SO_4 in this study were similar to those from artificially drained peatland catchments (Ramchunder *et al.* 2012).

In this study, significantly higher concentrations of DOC were observed in catchments managed via burning compared with intact catchments. Although, numerous drivers of increased DOC production have been proposed (e.g. water table drawdown via drainage (Wallage *et al.* 2006), warmer temperatures (Tranvik & Jansson 2002) or a reduction in SO_4 deposition (Evans *et al.* 2006a)), this study adds weight to the mounting (but not entirely unequivocal) evidence that burning may be a local driving factor in DOC production operating alongside larger scale factors. While it should be recognised that we only conducted seasonal spot sampling, intensive sampling by Yallop & Clutterbuck (2009) also documented an increase in DOC concentrations with the greater exposure of peat surface following burning. Furthermore, this relationship was observed for both ‘microscale’ (< 3 km^2) catchments and in larger catchments. Additionally, Yallop *et al.* (2011) working in three South Pennine catchments documented elevated humic DOC in catchments with a high proportion of new burns. However, further work is required as data from plot-scale studies to date are not able to account for these catchment-scale patterns (Holden *et al.* 2012).

Rotational vegetation burning effects on stream macroinvertebrate communities

Both the 3v3 survey and the 5v5 surveys revealed significant differences in community richness, *1/S* and dominance, and therefore we rejected H_2. This was in contrast to the findings of Ramchunder *et al.* (2012) where artificial drainage had no discernible effect on stream macroinvertebrate community metrics, and from previous forest wildfire research by Minshall *et al.* (1997) and Minshall (2003). Nevertheless, similar findings have been documented by Minshall *et al.* (2001) and by Viera *et al.* (2004) where the authors documented less resistance and resilience to post-fire spates. Indeed, the loss of terrestrial vegetation and post-fire flooding could have altered the physical properties in the stream channels of the burned catchments in this study. However, studies across a larger number of burned and unburned streams may be necessary to provide a more conclusive insight into burning effects on stream macroinvertebrate community structure.

Stream ecosystem functional group responses following rotational burning are poorly understood but our results show lower abundance of herbivores and predators in the burned sites partly supporting H_3. Furthermore, the ordination analysis demonstrated a shift in the stream macroinvertebrate community from one dominated by mayflies and large predatory stoneflies at the intact sites, to a community dominated by dipterans and smaller stoneflies at burned sites. Individual taxa respond differently to the various physical changes and shifts in food resource, and opportunistic species appear to favour streams impacted by fire (Mihuc & Minshall 1995; Minshall *et al.* 2001; Minshall 2003). The increase in Chironomidae relative abundance following rotational burning could be related to the elevated organic SSC (e.g. Vieira *et al.* 2004), or it could be a response to the reduction in predator abundance. Vuori & Joensuu (1996) and Ramchunder *et al.* (2012) found artificial drainage of peatlands encouraged increased Chironomidae and Simuliidae abundance, suggesting synergies between the stress imparted on stream ecosystems by seemingly disparate artificial drainage and vegetation burning management techniques.

The greater abundance of *Amphinemura* spp. in the burned catchments from both the 3v3 and the 5v5 surveys suggests nemourids are more resilient to the effects of rotational burning. These findings are supported by wildfire and post-wildfire work by Viera *et al.* (2011) and Mihuc & Minshall (1995) in the Guaje Canyon, New Mexico and Yellowstone, respectively. Dietary flexibility, life-history strategy (univoltine) and small-body size (therefore able to utilise refugia in microhabitats) may explain the higher abundance of nemourids at the rotationally burned catchments in our study. Both the 3v3 and the 5v5 surveys showed a lower abundance of herbivores, while the 3v3 survey showed a lower abundance of predators in the burned sites, suggesting a strong influence of land use on FFGs. The 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 (Bo *et al.* 2007). At present it is unclear whether burning altered producer biomass, thus depressing herbivore abundance (Vieira *et al.* 2004), or whether changes in the stream environment were more important for influencing herbivores directly. There is some evidence for the latter because scraper/grazer feeding can be quickly impaired on sediment smothered surfaces (Larsen & Ormerod 2010).

Implications for peatland and moorland management

In many regions of the world, the biodiversity and ecosystem services of headwater streams have been compromised due to catchment degradation (Harding *et al.* 1998; Allan 2004). This study suggests that rotational vegetation burning leads to alterations to peatland stream ecosystems, perhaps necessitating focused efforts to restore impacted systems. Although the catchments investigated in this study were <10 km^2, and therefore 'under the radar' of major management efforts being undertaken as part of the EU Water Framework Directive, the results suggest that a lack of detailed consideration of small headwater systems could be providing inaccurate estimates of the number of watercourses in the different ecological status classes. Structural alterations of macroinvertebrate communities can also influence ecosystem functional processes, and this study suggests that upland managers need to consider ways of reducing the extent or rotation frequency of burning to reduce effects on river ecosystems. There also needs to be more routine monitoring of upland systems such as those that we studied, both to characterise effects of contemporary land management and to monitor whether streams will recover if or when upland management changes are implemented.

Currently, there is a growing focus on the effects of peatland vegetation burning on peat carbon stores and DOC release (Worrall *et al.* 2007; Clay *et al.* 2009) whilst the impacts of burning on stream ecosystems have hitherto remained unknown. This is the first study to document the impacts of peatland vegetation burning on the relationships between physical, chemical and biological communities in river ecosystems, and has therefore added significantly to the current knowledge and understanding of rotational burning. It may be that prescribed burning also affects other aquatic organism groups (e.g. algae, microbes, fish) and there is a clear need for more work in this area, particularly given the apparent recent increase in burn frequency and encroachment of prescribed burning onto larger areas of blanket bog (Yallop *et al.* 2006b). We focused solely on headwater second-order streams and therefore need to examine the effects of upland prescribed burning further downstream to determine the spatial extent of burning impacts (Meyer & Wallace 2001). The generality of the results is difficult to determine at this stage because there have been no other published studies into stream ecosystem responses to heather burning, but ongoing research at different study sites across northern England appear to confirm the findings of this work. The similarities to findings from studies of wildfire in other locations suggests some common effects of vegetation burning and catchment disturbance for stream ecosystems (e.g. Minshall *et al.* 1997; Minshall 2003; Vieira *et al.* 2004; Mihuc 2005).

The enactment of recommendations and regulations surrounding burning needs to be done with sensitivity to the views of both grouse moor owners and managers and the wider array of groups with interests in upland ecosystems. In particular we need to improve knowledge exchange between government agencies, managers or upland stakeholders and scientists (Brown *et al.* 2010). Such exchanges will be important in developing appropriate moorland management regimes to deliver multiple ecosystem services and not just burning heather in rotation to maximise red grouse yields. Peatland fires occur at a global scale (Kuhry & Turunen 2006) and our results suggest that trade-offs are needed to satisfy both economic and ecological facets of the combined social-ecological systems in such areas, especially if fire is implemented as a management tool.

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Table 1. Catchment information for the ten stream study sites

Site	**Management**	**Soil types**	**Catchment area (km^2)[a]**	**Grid reference**
Moss Burn (Teesdale)	Intact	Blanket peat	2.15	54°41’1’’N 2°27’0’’W
Unnamed 2nd order tributary of River Tees (Teesdale)	Intact	Blanket peat, stagnogley, stagnohumic gley, humic gley, fine loam, alluvial gley	2.23	54°15’7’’N 21°6’1’’W
Snaizehope (Wensleydale)	Intact	Blanket peat, alluvial floodplain	1.12	54°41’8’’N 2°26’8’’W
Crowden Little Brook (Peak District)[b]	Intact	Blanket peat, fine sandy loam	2.11	53°30’8’’N 2°53’4’’W
Short Grain (Peak District)[b]	Intact	Blanket peat, fine sandy loam	1.49	53°34’2’’N 2°55’9’’W
Great Eggleshope Beck (Teesdale)	Burned	Blanket peat, stagnogley	4.10	54°40’5’’N 2°3’8’’W
Eller Beck (Teesdale)	Burned	Blanket peat, stagnogley, fine loam	1.67	54°29’2’’N 2°0’9’’W
New Water (Geltsdale)	Burned	Blanket peat, stagnogley	2.18	54°50’8’’N 2°37’1’’W
Ashop Clough (Peak District)[b]	Burned	Blanket peat, stagnogley, fine sandy loam	1.82	53°24’8’’N 2°53’0’’W
Thickwoods Brook (Peak District)[b]	Burned	Blanket peat, fine sandy loam	1.61	53°29’2’’N 2°41’5’’W

[a]Measured using the hydrology tool in ArcGIS (Esri, Redlands, CA, USA)

[b] Streams sampled only as part of the 5v5 survey

Table 2. Descriptive statistics and RM-ANOVA and One-way ANOVA results for the physicochemical variables measured during the 3v3 and 5v5 surveys respectively.

3v3	**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 m^{-2})**	**FPOM (mg m^{-2})**	**POM (mg m^{-2})**	**Water temperature (°C)**	**Discharge (m^{3}s^{-1})**
Intact																
Mean	3.75	0.36	2.29	0.05	0.49	14.67	11.07	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	0.79	5.80	18.00	4.29	1.00	4.0	0.02	0.04	0.07	0.5	0.01
Max	9.35	1.26	5.65	0.18	1.33	67.31	19.30	191.40	8.65	12.80	6.9	1.52	3.48	4.07	18.5	0.25
Stdev	2.64	0.40	1.58	0.06	0.39	17.11	3.46	60.04	1.54	3.65	1.4	0.38	0.87	1.02	5.9	0.09
Burned																
Mean	5.90	0.79	2.04	0.15	0.47	29.93	10.54	59.76	5.92	13.57	2.5	0.16	1.00	1.16	9.4	0.04
Min	2.33	< 0.01	2.54	< 0.01	0.01	4.56	4.90	36.80	4.86	1.00	2.0	0.02	0.21	0.25	1.5	0.01
Max	10.25	1.76	11.32	0.51	1.78	87.20	18.40	112.40	8.35	28.40	3.1	0.59	2.56	2.58	16.4	0.20
Stdev	1.92	0.59	2.20	0.16	0.44	17.91	3.85	21.47	0.89	8.83	0.5	0.15	0.67	0.72	5.1	0.05
Season ($F_{4,29}$)	F=8.16 ***P*=0.033**	F=14.87 ***P*=0.011**	F=2.18 *P*=0.234	F=1.40 *P*=0.375	F=1.20 *P*=0.431	F=0.85 *P*=0.562	F=1.89 *P*=0.276	F=0.16 *P*=0.949	F=1.17 *P*=0.442	F=0.82 *P*=0.575	No replicates	F=2.79 *P*=0.172	F=6.25 *P*=0.052	F=0.89 *P*=0.544	F=1.04 *P*=0.484	F=2.99 *P*=0.157
Land management ($F_{1,29}$)	F=21.00 ***P*=0.010**	F= 14.41 ***P*=0.019**	F=25.41 ***P*=0.007**	F=14.87 ***P*=0.018**	F=968.60 ***P*<0.001**	F=45.87 ***P*=0.002**	F=0.08 *P*=0.791	F=4.55 *P*=0.100	F=7.18 *P*=0.055	F=146.71 ***P*<0.001**	F=19.88 ***P*=0.011**	F=27.28 ***P*=0.006**	F =50.71 ***P*=0.002**	F=25.36 ***P*=0.007**	F=0.21 *P*=0.671	F=1.94 *P*=0.236
Season*Land management ($F_{4,29}$)	F=0.47 *P*=0.754	F=1.68 *P*=0.194	F=0.86 *P*=0.506	F=1.86 *P*=0.156	F=2.09 *P*=0.120	F=0.87 *P*=0.500	F=2.35 *P*=0.090	F=2.29 *P*=0.096	F=1.41 *P*=0.266	F=0.35 *P*=0.838	No replicates	F=0.66 *P*=0.629	F=0.26 *P*=0.901	F=0.44 *P*=0.778	F=2.92 ***P*=0.047**	F=2.86 ***P*=0.050**
5v5																
Intact																
Mean	2.97	0.52	3.99	0.10	0.39	6.63	9.00	75.34	6.27	3.00	5.1	0.13	0.13	0.23	10.61	0.12
Min	0.38	<0.01	0.55	0.03	0.13	0.09	8.00	18.00	4.37	0.60	3.6	0.04	0.04	0.09	8.21	0.03
Max	6.56	2.00	8.77	0.25	0.65	17.89	9.70	191.40	8.33	8.80	6.9	0.31	0.31	0.48	12.90	0.25
Stdev	2.72	0.87	3.86	0.05	0.24	6.81	0.73	71.37	1.59	3.37	1.1	0.13	0.12	0.17	1.84	0.09
Burned																
Mean	3.99	0.76	4.39	0.30	26.13	7.90	12.36	33.13	7.96	19.60	2.3	0.52	0.52	0.16	8.17	0.30
Min	2.33	<0.01	2.79	0.02	0.60	0.04	7.10	11.35	4.18	8.00	1.4	0.06	0.06	0.01	6.01	0.02
Max	5.28	2.42	6.24	0.51	51.85	29.89	17.80	79.10	9.90	32.61	3.1	1.65	1.65	0.39	13.23	0.51
Stdev	1.29	1.11	1.72	0.64	23.83	12.94	4.87	30.21	2.30	10.30	0.6	0.67	0.68	0.15	3.11	0.20
Land management ($F_{1,9}$)	F=0.58 *P*=0.469	F=0.15 *P*=0.710	F=0.05 *P*=0.838	F=17.41 ***P*=0.003**	F=4.09 *P*=0.078	F=9.91 ***P*=0.014**	F=0.55 *P*=0.480	F=0.53 *P*=0.487	F=0.08 *P*=0.784	F=19.38 ***P*=0.002**	F=22.40 ***P*=0.001**	F=0.85 *P*=0.386	F=2.51 *P*=0.152	F=1.90 *P*=0.206	F=1.37 *P*=0.275	F=0.02 *P*=0.886

Cl – Chloride; NO_3 – Nitrate; SO_4 – Sulphate; Al – Aluminium; Fe – Iron; DOC – Dissolved organic carbon; DO – Dissolved oxygen; EC – Electical conductivity; SSC – Suspended sediment concentration; D_{50} – median clast size; CPOM – Coarse Particulate Organic Matter; FPOM – Fine Particulate Organic Matter; and POM – Particulate Organic Matter

Table 3. Descriptive statistics and RM-ANOVA and One-way ANOVA results for the macroinvertebrate community metrics and FFGs measured during the 3v3 and 5v5 surveys respectively.

3v3	**Total abundance (# per m^2)**	**Richness**	**Simpson's Diversity (*1/S*)**	**Dominance (D)**	**Shredders**	**Predators**	**Herbivores**	**Gathering-collectors**	**Filtering-collectors**	**Ephemeroptera**	**Plecoptera**	**Trichoptera**	**Chironomidae**	**Simuliidae**	**Other**
Intact															
Mean	2665	30	6.10	37.7	610	109	372	1525	42	1061	564	85	568	16	308
Min	972	16	1.74	18.1	64	4	0	720	0	0	144	32	112	0	0
Max	4592	41	11.05	75.3	1984	240	1552	2764	136	3480	2016	184	1208	56	1004
Stdev	990	8	3.08	17.5	562	77	464	595	33	1099	508	45	378	19	323
Burned															
Mean	2344	23	3.76	45.3	105	6	4	298	50	271	509	25	1075	190	122
Min	1137	11	2.01	29.3	12	1	0	134	3	4	60	4	501	12	4
Max	4540	39	5.97	70.0	230	18	34	781	254	1116	1112	56	3176	1016	628
Stdev	924	7	1.18	11.3	72	6	9	188	69	345	339	16	708	278	183
Season ($F_{4,29}$)	F=5.36	F=0.82	F=0.96	F=1.07	F=3.97	F=2.01	F=0.62	F=3.87	F=0.26	F=2.55	F=2.39	F=0.45	F=9.91	F=0.90	F=0.190
	P=0.066	*P*=0.573	*P*=0.515	*P*=0.473	*P*=0.105	*P*=0.257	*P*=0.675	*P*=0.109	*P*=0.892	*P*=0.193	*P*=0.210	*P*=0.772	***P*=0.024**	*P*=0.540	*P*=0.258
Land management ($F_{1,29}$)	F=2.33	F=10.85	F=8.50	F=6.73	F=0.12	F=8.53	F=23.43	F=0.84	F=2.34	F=40.87	F=0.36	F=11.80	F=76.17	F=4.06	F=26.11
	P=0.202	***P*=0.030**	***P*=0.043**	*P*=0.060	*P*=0.751	***P*=0.043**	***P*=0.008**	*P*=0.410	*P*=0.201	***P*=0.003**	*P*=0.582	***P*=0.026**	***P*=0.001**	*P*=0.114	***P*=0.007**
Season*Land management ($F_{4,29}$)	F=0.55	F=0.53	F=0.61	F=0.39	F=0.59	F=0.49	F=0.46	F=0.39	F=5.33	F=0.15	F=0.91	F=1.71	F=0.07	F=1.51	F=0.18
	P=0.701	*P*=0.714	*P*=0.662	*P*=0.811	*P*=0.671	*P*=0.740	*P*=0.762	*P*=0.811	***P*=0.004**	*P*=0.963	*P*=0.476	*P*=0.188	*P*=0.992	*P*=0.237	*P*=0.948
5v5															
Intact															
Mean	2296	32	8.83	30.33	766	58	209	1191	56	790	666	94	358	24	312
Min	1156	25	3.78	16.26	256	16	4	412	32	4	248	40	88	0	32
Max	3560	40	13.70	48.79	1684	116	368	2120	100	2044	1328	144	1052	48	848
Stdev	955.45	6	4.59	16.42	577	38	148	636	25	776	776	44	396	21	326
Burned															
Mean	2182	20	2.98	52.58	618	50	6	1438	62	245	598	17	1121	46	25
Min	1350	16	1.76	37.32	204	12	0	672	24	16	188	4	620	24	4
Max	2804	23	3.93	74.18	1148	140	16	2512	136	696	1112	38	2080	132	44
Stdev	582.96	3	0.86	13.91	430	52	7	714	46	290	405	13	648	48	14
Land management ($F_{1,9}$)	F=0.01	F=19.31	F=9.73	F=5.80	F=0.21	F=0.43	F=10.82	F=0.38	F=0.54	F=0.66	F=0.14	F=17.64	F=8.26	F=1.69	F=9.57
	P=0.973	***P*=0.002**	***P*=0.014**	***P*=0.043**	*P*=0.662	*P*=0.532	***P*=0.011**	*P*=0.556	*P*=0.484	*P*=0.441	*P*=0.719	***P*=0.003**	***P*=0.021**	*P*=0.230	***P*=0.015**

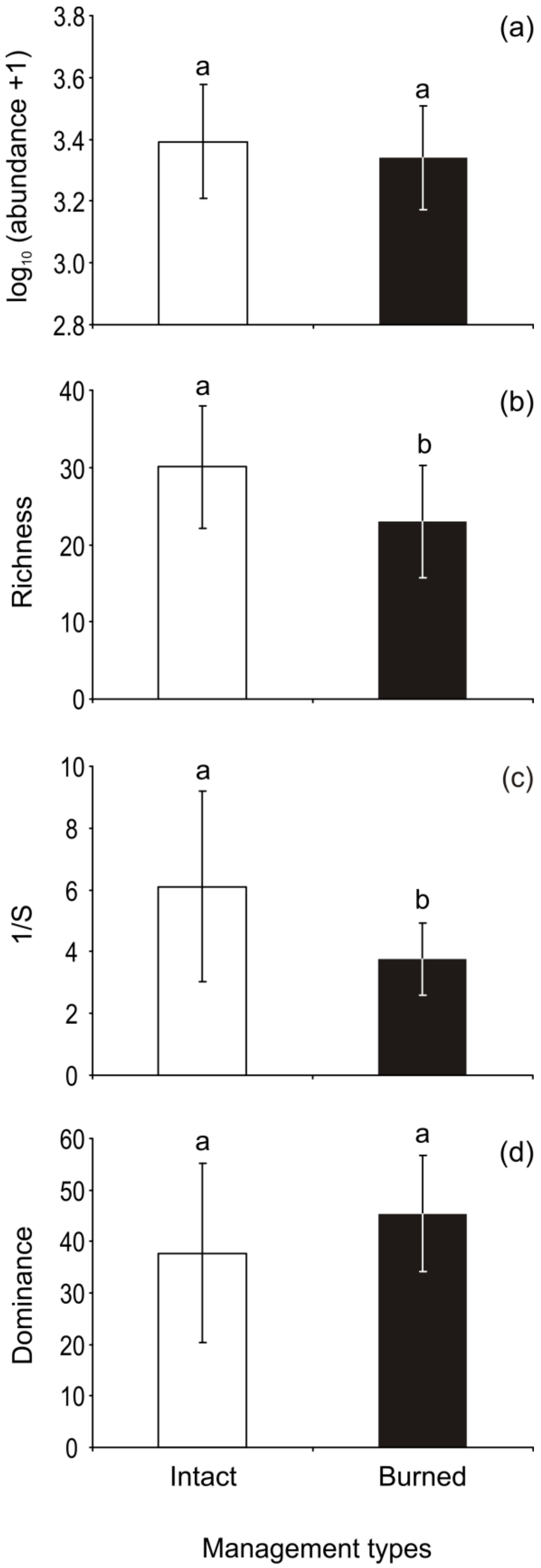

Fig 1. Effects of land management type on **(a)** $\log_{10}$(abundance +1); **(b)** Richness; **(c)** *1/S* (Simpson's Diversity); and **(d)** Dominance for the 3v3 survey (Error bars shows $\pm$ 1 SD from the mean).

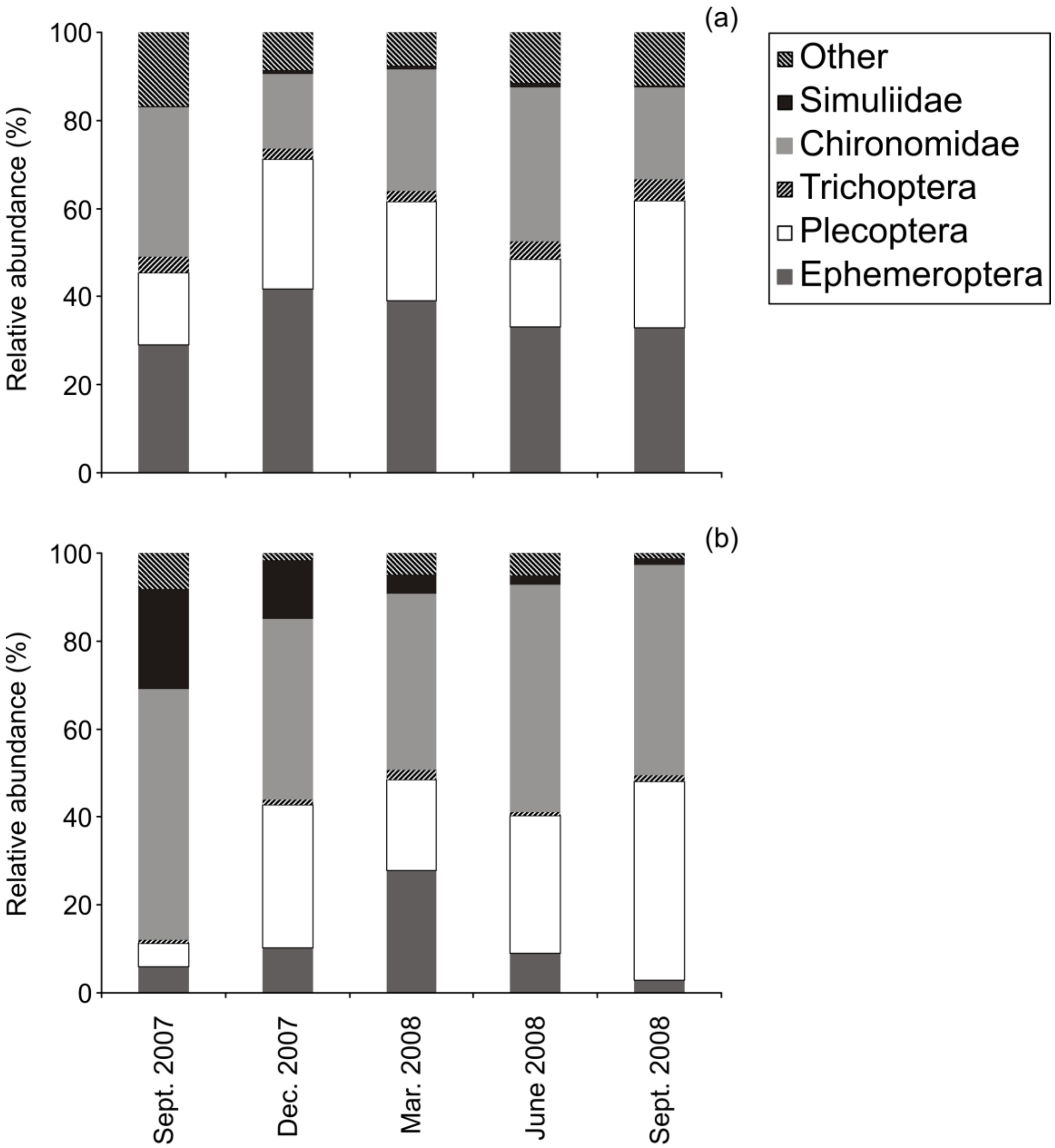

Fig 2. Seasonal effects on relative abundance of EPT (Ephemeroptera, Plecoptera and Trichoptera), Chironomidae, Simuliidae and Other taxa from **(a)** intact and **(b)** rotationally burned sites.

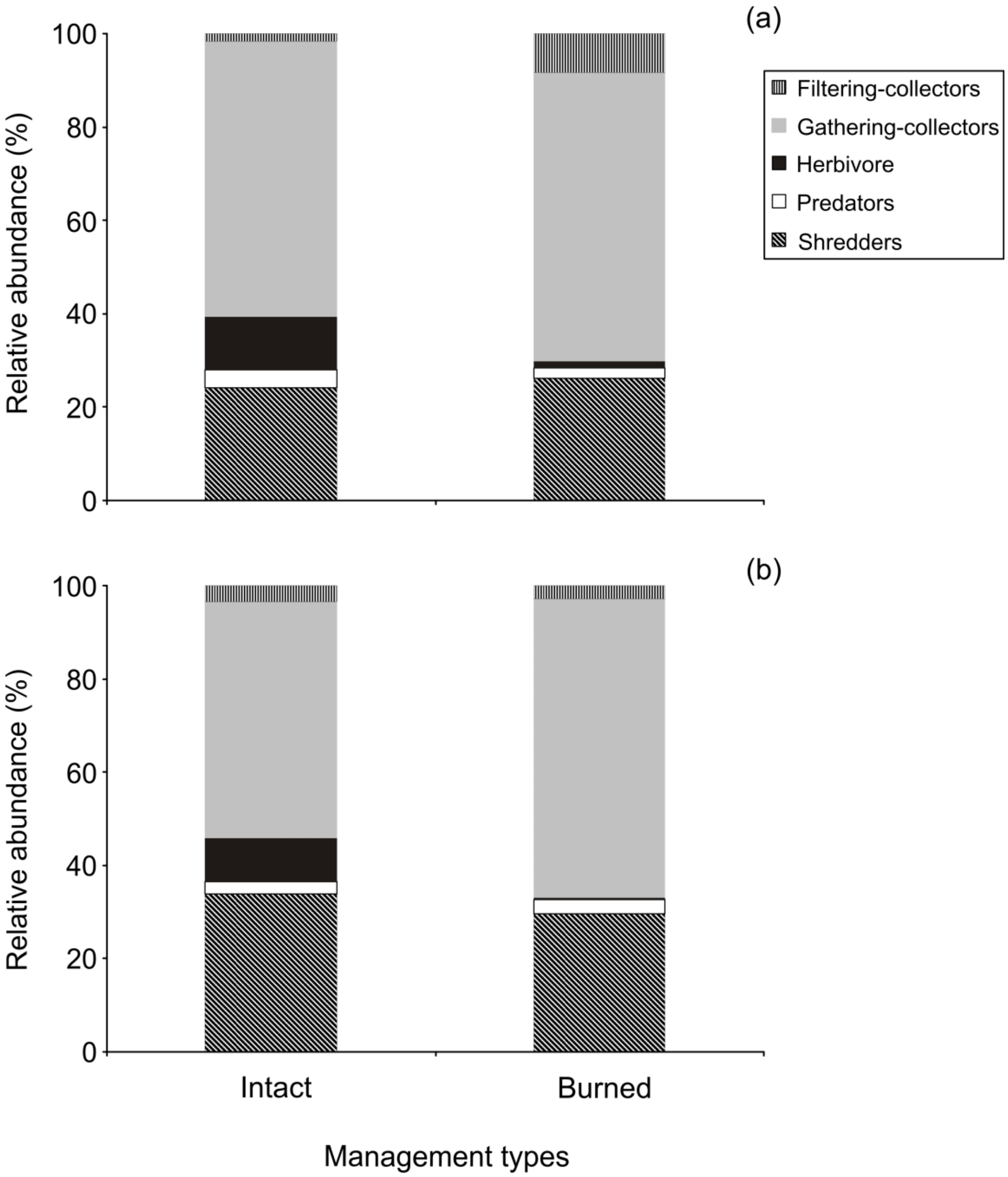

Fig. 3 Comparison of relative abundances of Functional Feeding Groups (FFGs) between intact and burned sites from the **(a)** 3v3 survey (amalgamation of the sites from every quarter from Sept. 2007 to Sept. 2008) and **(b)** 5v5 survey.

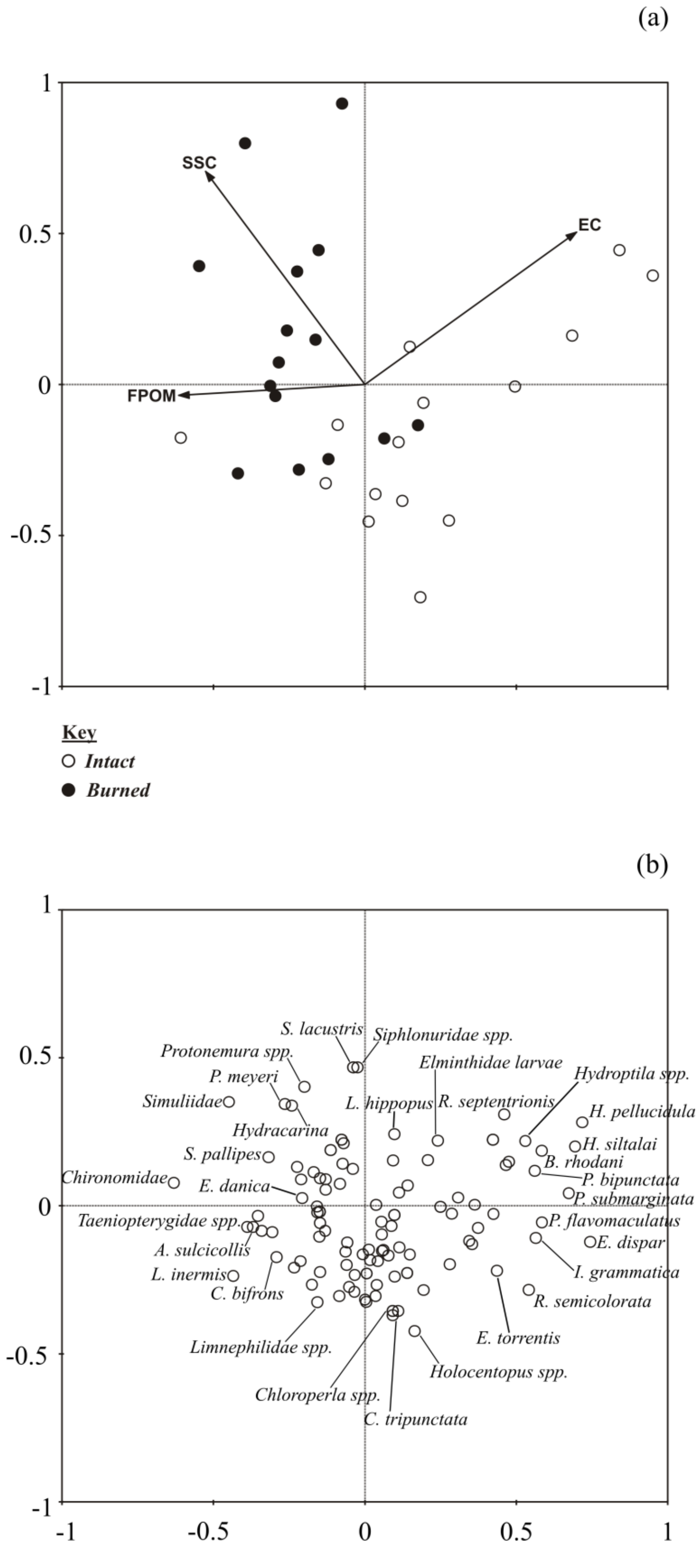

Fig. 4(a) Site-physicochemical variable biplot and **(b)** species-physicochemical variable biplot from the partial Redundancy Analysis (*p*RDA) for the 3v3 survey. Ordinations are based on *p*RDA using Time as a covariable. Only significant (Electrical conductivity [EC], $p = 0.005$ (% variance = 19.5); fine particulate organic matter [FPOM], $p = 0.015$ (% var. = 13.2); suspended sediment concentration [SSC], $p = 0.018$ (% var. = 11.2)) (forward selection) in the constrained ordination are shown.

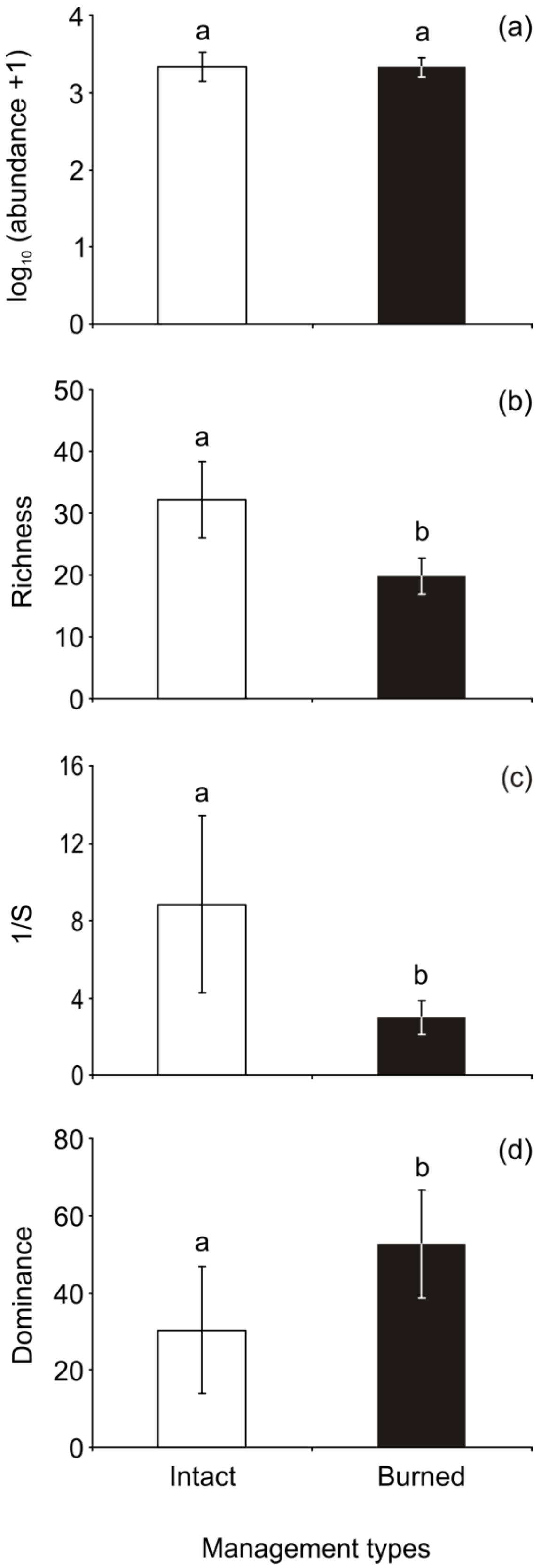

Fig 5. Effects of land management type on **(a)** $\log_{10}$(abundance +1); **(b)** Richness; **(c)** *1/S* (Simpson's Diversity); and **(d)** Dominance for the 5v5 survey (Error bars shows $\pm$ 1 SD from the mean).

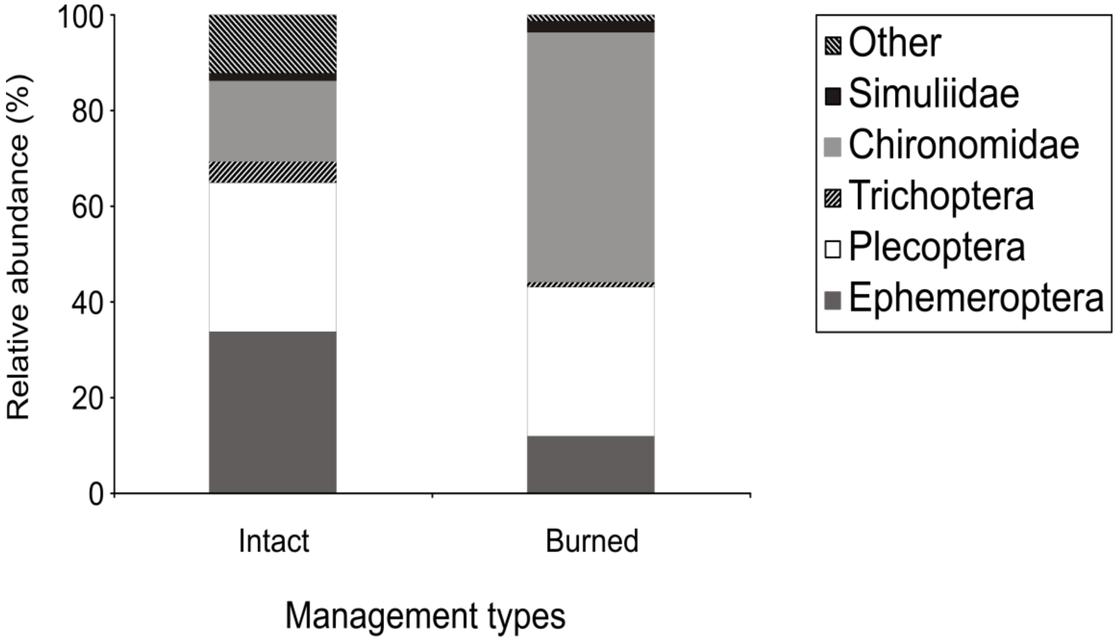

Fig 6. Effects of land management type on relative abundances of EPT (Ephemeroptera, Plecoptera and Trichoptera), Chironomidae, Simuliidae and Other taxa for the 5v5 survey.

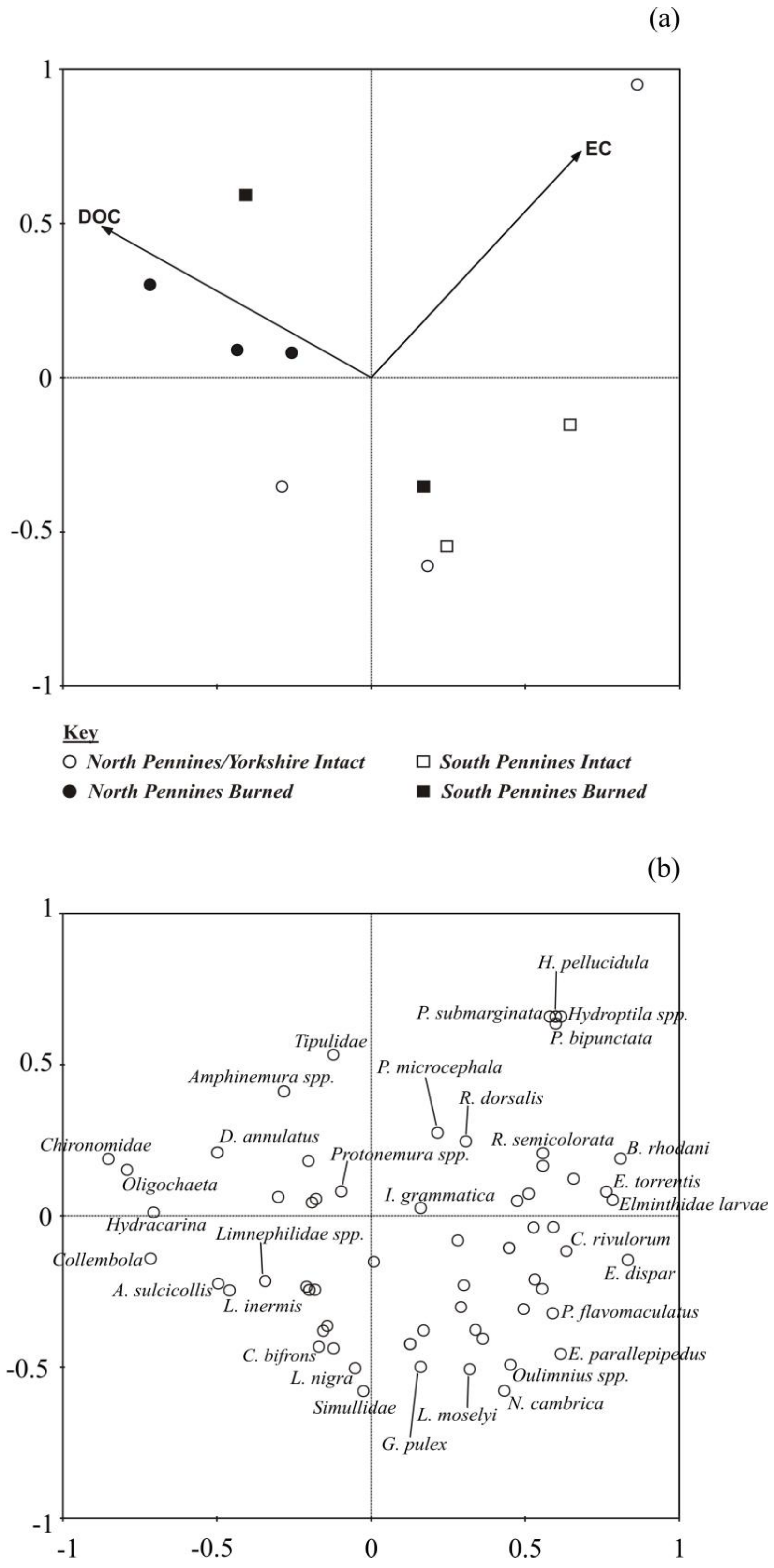

Fig. 7(a) Site-physicochemical variable RDA biplot and **(b)** species-physicochemical variable RDA biplot from the 5v5 survey. Ordinations are based on partial Redundancy Analysis (*p*RDA) using Time as a covariable. Only significant (Dissolved organic carbon [DOC], p = 0.005 (% variance = 32.00); electrical conductivity [EC], p = 0.017 (% variance = 15.70)) (forward selection) in the constrained ordination are shown.