Article:

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SUPPLEMENTARY INFORMATION

METHODS

FPOM and invertebrate community analysis

For FPOM, and each community composition metric, we initially examined the response variable distribution from histograms and residuals of linear models to determine the most appropriate family. The normal distribution was specified for taxonomic richness and density of all invertebrates, binomial for all other relative abundance metrics and Poisson for FPOM. FPOM data were modelled as mg/m^2. For binomial models, an observation level random effect was incorporated to account for overdispersion. Analysis was undertaken using R packages nlme and lme4, with model R^2 calculated using MuMIn and effect-size plots created using sjPlot.

Initial assessments were made using a site-level random effect alongside management (burned or unburned) and all site-level covariates, but due to the large number of model terms relative to the number of observations it was not possible to use this mixed approach to model all potential interactions. Datasets comprising a larger number of site observations (cf. Noble et al., 2018) would allow for testing these in more detail. Thus, initial screening of data used fixed effect models with management and covariates, then covariates with $p<0.05$ were retained in a parsimonious model including a random site effect nested in sampling time period, and corrected AIC calculated to confirm that the reduced model provided an enhanced fit. Model outputs showed for the same effect 'direction' for retained variables. In comparison to Brown et al. (2013), our additional analysis here used data for 5 of the 6 sampling periods (i.e. excluding Spring 2010 season) because water temperature and flow data co-variables were not measured prior to the first sampling period which coincided with the initial setting up of datalogger arrays. Time between the five sample periods was calculated as the number of days, and initial examination of models suggested no clear autocorrelation, which we attribute to sample intervals being long (3 months+ in most instances) and invertebrate communities in these rivers respond to habitat change quickly in some circumstances (<1 day to 1 month as shown in some recent sedimentation experiments; Aspray et al., 2018, Brown et al., 2019). The full suite of co-variables incorporated initially included water temperature and catchment size as reported in Brown et al. (2013), and all five river flow event timing variables calculated by Holden et al. (2015) but initial tests showed strong (>0.7) association between flow variables. Thus, we retained only Time from rainfall start to flow peak and Rainfall total before hydrograph rise. Magnitude variables would offer an additional means of incorporating flow into the analysis but these data were unavailable for all study sites. Geology was similar among sites with the exception of limestone which was present in two burned sites and three unburned sites in the North Pennines. As limestone presence/absence can influence aquatic invertebrate communities, we incorporated this as an additional covariate. The geology covariate also distinguishes North Pennines rivers from other sites, complicating its interpretation against other geographical effects (e.g. North Pennine rivers are typically more remote from large urban areas than the South Pennines). Co-variables were centred by mean values prior to analysis.

Contextual literature review

In our initial evaluation of papers, we attempted to extract results which would enable a quantitative way of evaluating the evidence base using comparisons of effect-sizes. Unfortunately, few appropriate datasets have been reported routinely in the evidence base and so it was not possible to undertake such an analysis. This should be considered as a future research aim as more data sources become available for formal meta-analysis. Our focus on published research papers avoided double-counting findings from unpublished reports and journals. We made no assessment of the appropriateness of methods applied in each study although it is notable that this has recently become a wider topic of discussion (Baird et al., 2019; Evans et al., 2019; Young et al., 2019). Such considerations could be used to weight studies in future meta-analyses to aid the decision-making process. Whilst our categorisation approach provides an overview of suggested impacts of burning from multiple studies, we appreciate that it relies on our interpretation of written reports, and the conclusions of those papers may be based on methods or analysis that other researchers consider to be problematic. The analysis also draws on suggested effects of burning from papers that are most often based on conclusions influenced only by p-values of statistical tests (cf. Halsey, 2019). Policy makers could benefit from clearer presentation of summary statistics and effect size analysis in all studies because these could then be utilized in numerical meta-analyses to contrast various potentially influential factors such as geographical location, sponsors, researchers, whether a paper has been subject to genuine critical review or not, length/timing of study, level of replication, methods and approaches. It would also be beneficial in future to have a set of core sites and measures, undertaken using protocols agreed by the peatland research community, so that researchers can work together towards a common goal as we have suggested previously (Brown et al., 2015, p1420). Such a principle, following those often used in the medical research community, would strengthen the evidence base and filter out studies that are inadequate for further long-term analysis.

We are grateful to A&H for their suggested use of the following search term in Web of Knowledge, which provided additional papers for consideration:
TS=((burn OR "fire") AND (peat* OR heath* OR moor* OR "bog" OR "mire" OR upland*) AND ("habitat management" OR "biodiversity" OR "grouse" OR bird* OR plant* OR "vegetation" OR sphagnum* OR invertebrate* OR insect* OR amphibian* OR reptile* OR mammal* OR "water quality" OR "water colour" OR "flow" OR "saturated" OR "dissolved organic carbon" OR "DOC" OR hydrolog* OR infiltrat* OR "soil" OR carbon budget* OR "carbon cycling" OR carbon flux* OR "carbon sequestration" OR carbon stock* OR "carbon storage" OR ecosystem* OR environment*))*
Settings: language = English; document types = article; timespan = 1945 to 2019.

Analysis for potential sponsor effects

Grouse-shooting industry groups were defined as those which actively promote or support prescribed burning as part of grouse moor management (Game & Wildlife Conservation Trust (GWCT), formerly the Game Conservancy Trust; The Heather Trust, The Moorland Association, plus landowners or estates directly involved in managing grouse shoots by means of prescribed burning). Non-grouse shooting groups were defined as those not actively promoting or supporting prescribed burning as part of grouse moor management (e.g. government agencies, research councils, universities, upland restoration groups such as Moors for the Future/Yorkshire Peat Partnership, water companies). Government agencies were defined as those that shape national-scale environmental policy (e.g. Department of Agriculture for Northern Ireland, Department for Environment, Food and Rural Affairs (Defra) – formerly the Ministry of Agriculture, Fisheries and Food (MAFF), Natural England - formerly English Nature, Scottish Government, Scottish Natural Heritage). Non-government agencies were all other groups.

Due to the relatively small number of observations available across ecosystem properties, Fisher's Exact Test for count data was used to test associations between funding groups and research conclusions for each of the seven ecosystem properties and the combined-effect using R v.3.5.2. Pairwise comparisons were conducted using the Fisher multi-comparison test in the RVAideMemoire package (Hervé, 2019), and applying a Bonferroni correction. The test was unconditioned because rows and column totals varied. The test assumes independence of observations but this might not be the case for some long-term studies such as those reporting vegetation changes over time at the Moor House experimental plots in northern England with some of the same authors (Lee et al., 2013; Marrs et al., 2019b; Milligan et al., 2018). We considered it appropriate to relax this assumption because we were analyzing conclusions being reported in individual publications, and we accepted the judgement of the scientists and journals publishing those papers that there were enough new observations to justify publication.

In addition to BASC funding, A&H cited in their conclusions that there was forthcoming work from Heinemeyer et al. (report now published – Heinemeyer et al. 2020) about a project called Peatland-ES-UK which includes funding from both grouse-shooting industry and non-grouse shooting organisations. We are happy to hear that this provides new evidence to government. However, it undermines the argument that policy makers are unduly influenced by the EMBER work as there is ample evidence, as here, that policy groups collect evidence from multiple sources/research groups, and then evaluate such evidence in an open and balanced manner (Glaves et al., 2013). The commentary paper (Ashby & Heinemeyer, 2019) was listed by Heinemeyer as an apparent output from Peatland-ES-UK at the project's advisory group meeting in March 2019. Phase 2 of that project is currently funded by a range of organisations, including water companies, Yorkshire Wildlife Trust, the BASC and the Moorland Association, the latter also being a body that promotes the management of heather on grouse moors, including on peatlands, through the practice of controlled burning. On 15 October 2018, Heinemeyer published, on social media, a note of thanks to the Heather Trust for funding the Peatland-ES-UK project (https://twitter.com/AndreasHeinem/status/1051819786265616384, last accessed 29 July 2019). More recently, Ashby also revealed that he has undertaken work for the Moorland Association since April 2019. It is again not clear, therefore, why A&H did not declare these additional perceived competing interests in their paper criticising selected examples of our earlier work. Omissions such as this can create confusion when not declared fully, and as there is no way of knowing if this is a wider issue affecting other publications, our analysis was based only on information declared in the original papers.

RESULTS

Figure S1. Scatterplots of hydrological metrics detailed in Holden et al. (2016) show no clear relationships with altitude or catchment size.

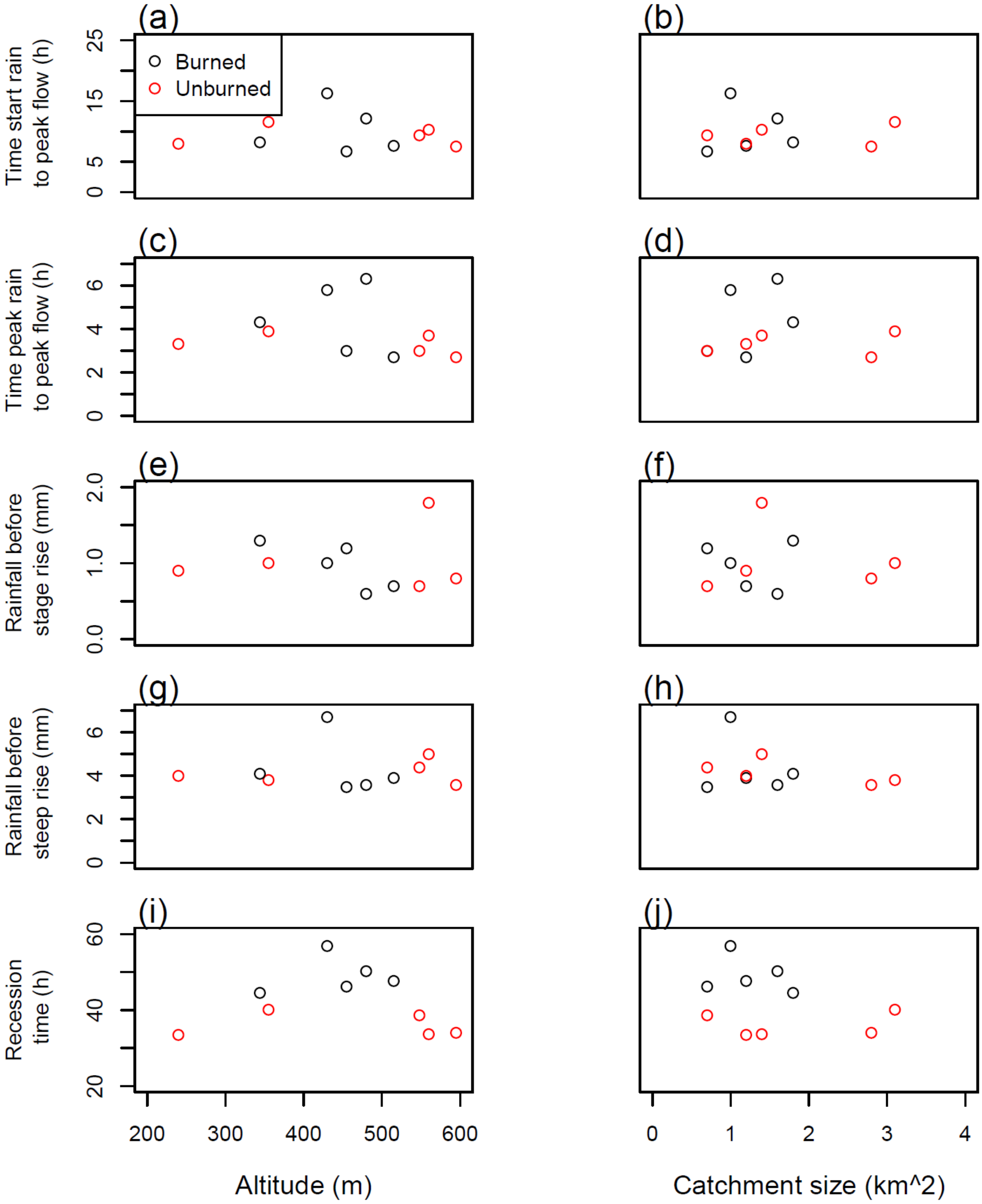


Figure S2. No relationships were evident between fine particulate organic matter (FPOM) densities and (a) catchment size (R^2=0.007, p=0.57), (b) altitude (R^2=0.001, p=0.81) or (c) precipitation during month of sampling (data sources as in A&H) (R^2=0.005, p=0.63) in the EMBER rivers studied by Brown et al. (2013). See Supplementary Table 2 for FPOM and rainfall data; catchment size and altitude data were reported in Brown et al. (2013). Log transformation is used only for clarity of presentation; statistics presented are for untransformed data. For c, data sources from A&H were used for rainfall estimates and so it should be noted that for some sites the values were the same as a result of modelling errors arising from the gridded analysis used by A&H.

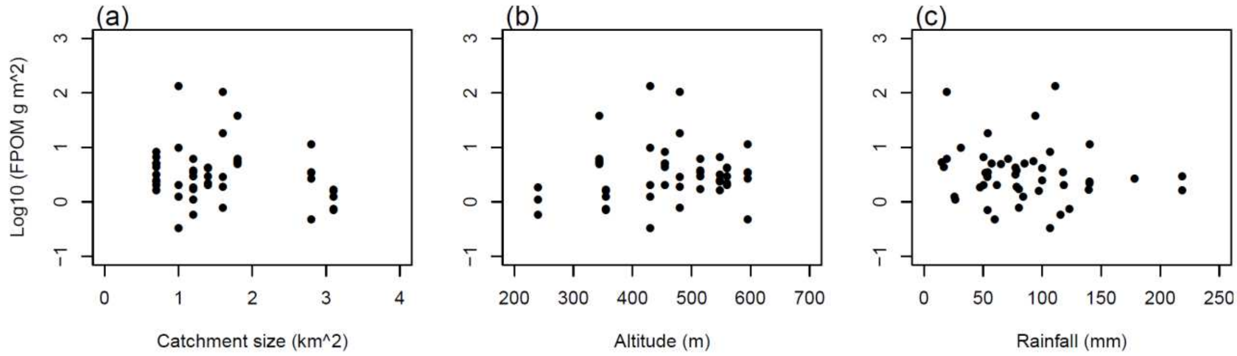


Figure S3. Photographs highlighting examples of vegetation burning alongside or over watercourses

(a) Recent burn patch crossing a watercourse (foreground), Ashop Clough catchment, north Derbyshire

(b) Recent burned patches adjacent to a watercourse (centre left, light grey colour; Bull Clough, South Yorkshire)

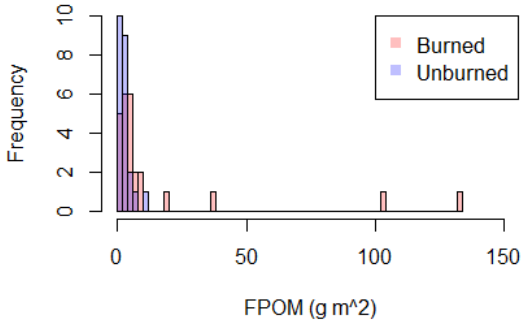
(c) Recent burn patch (left) adjacent to a watercourse, with older burn patches in the near distance crossing the watercourse (Walshaw Moor, West Yorkshire)

(d) Recent burn patch adjacent to two watercourses (Walshaw Moor, West Yorkshire)

Invertebrate community metrics

No covariates were associated with FPOM densities. Burning was associated with more riverbed FPOM (2.4x, 95% range -0.3 to 5.1, Supplementary Figure 4) supporting suggestions in Brown et al. (2013). For taxonomic richness, whilst burn was not a ‘significant’ predictor in terms of its p-value, it was still associated with a general tendency for reduced taxonomic richness (-2.1, 95% range -5.6 to +1.4) as suggested previously (Brown et al., 2013). The effect was less certain due to estimates showing that burn sites occasionally hosted up to 1 extra taxon. Richness was associated positively with limestone presence in North Pennine catchments. More replication of sites would be needed to allow for a detailed consideration of burn effects *within* region/geology, but these results appear similar to Noble et al.’s (2018) vegetation analysis whereby burning effects might still be evident despite regional geographical differences.

Figure S4. Altered distribution and extreme FPOM densities in burned catchments, and mixed model statistics.



	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	7.6469	0.2299	33.264	< 2e-16
Burn	0.8781	0.313	2.805	0.00503
$R^2m = 0.14$, $R^2c = 0.99$				

Mixed model statistics for taxonomic richness

	Value	Std.Error	DF	t-value	p-value
(Intercept)	12.29044	1.705159	38	7.207799	0
Burn	-2.0815	1.799477	7	-1.15673	0.2853
Geology	5.020917	1.799818	7	2.789681	0.0269
$R^2m = 0.39$, $R^2c = 0.74$					

Burning was associated with a strong negative effect on Ephemeroptera relative abundance (ratio x10.8, 95% CI from -5.6 to -15.9) even when controlling for other site-based variables, as noted in our previous study. Catchment size was associated with a positive effect (ratio x3.9±3.5) but less than the effect of burning. There was also an association between geology and Ephemeroptera (x9.3, 95% CI from 4.4 to 14.2) but this was in the opposite direction to burning (similar to taxonomic richness), suggesting higher abundances in North Pennine limestone influenced rivers. Burning was associated with a strong positive effect on Chironomidae relative abundance (x4.7, 95% CI from 1.6 to 7.8) as suggested previously. Water temperature was also associated with a small positive effect (ratio x1.3, 95% CI from -0.8 to 3.4), although less than the effect of burning, which might reflect a seasonal dynamic or slightly higher relative abundance at lower altitude sites. Effect size estimates suggest a slight increase of Simpson’s diversity linked to burning (ratio x1.6, 95% CI -2.5 to 5.6) after accounting for co-variables, but R^2 for both fixed effects and including random suggested a poor fit model. For total invertebrate density, no strong effect was detected for any variable, although there was a slight tendency towards more invertebrates in burn sites as previously suggested (Brown et al., 2013). There was a small effect of the time from rainfall start to flow peak on total density (46±35) which may reflect between site differences in flashiness as well as burning (Holden et al., 2015).

Mixed model statistics for Ephemeroptera relative abundance

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-3.6744	0.8783	-4.184	2.87E-05
Burn	-2.3752	0.9729	-2.441	0.0146
Size	1.3523	0.5872	2.303	0.0213
Time rain start to flow peak	0.338	0.1759	1.921	0.0547
Geology	2.2283	0.9218	2.417	0.0156

$R^2m = 0.39$, $R^2c = 0.66$

Mixed model statistics for Chironomidae relative abundance.

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-2.9103	0.3356	-8.672	< 2e-16
Burn	1.5416	0.4603	3.349	0.00081
Temperature	0.2981	0.0699	4.265	2.00E-05

$R^2m = 0.21$, $R^2c = 0.55$

Mixed model statistics for Simpson's diversity

	Estimate	Std.Error	z-value	Pr(>|z|)
(Intercept)	1.1233	0.6824	1.646	0.0997
Burn	-0.4412	0.7335	-0.602	0.5475
Geology	0.7274	0.73	0.996	0.3191

$R^2m = 0.06$, $R^2c = 0.06$

Mixed model statistics for total invertebrate density

	Value	Std.Error	DF	t-value	p-value
(Intercept)	739.2256	71.93715	38	10.27599	0
Burn	66.4468	100.077	7	0.663957	0.528
Time rain start to flow peak	46.6306	17.89036	7	2.606468	0.0351

$R^2m = 0.14$, $R^2c = 0.14$

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