Quantifying the impact of wildfires in Northern Ireland

Final Report 2016

Quercus
Biodiversity & Conservation Biology

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Quantifying the impact of wildfires in Northern Ireland: Final Report 2016

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Funders: Northern Ireland Environment Agency (NIEA)

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between the Northern Ireland Environment Agency and
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EXECUTIVE SUMMARY

1. During the unusually warm and dry conditions of spring 2011 numerous wildfires occurred in upland habitats across Northern Ireland resulting in 3,801 ha of damage within Areas of Special Scientific Interest (ASSIs).

2. The aim of this project was to establish the broad ecological impact of such events and the degree of recovery over a short-term (3.5 years) period post-wildfires; on dry heath, wet heath and blanket bog habitats of conservation concern.

3. Surveys of soil chemistry, seed bank, vegetation and bird were conducted at 122 quadrats (2x2m) at six designated sites throughout Northern Ireland during the summer/autumn of 2012, 2013 and 2014; corresponding to one, two and three years post-wildfire. A pilot study of impacts on ground beetles and spiders was conducted in 2012 at two ASSIs, and this was extended to all six study sites in 2013 and 2014.

4. The chemical composition of soils differed between burnt and unburnt control plots; with both phosphorus and calcium levels remaining elevated 3.5 years post-wildfires. Calcium concentrations remained higher in burnt sites across all three habitat types, whilst phosphorus levels remained higher in blanket bog and wet heath but returned to the same level as unburnt plots in dry heath. There were no detectable differences in pH, nitrogen, carbon, potassium, magnesium or sulphur concentrations between burnt and unburnt areas one-and-a-half years after the wildfires.

5. Shrub height was significantly lower in areas where wildfires had occurred and recovery was reduced by grazing pressure in burnt areas. After 3.5 years average shrub height was 11 cm in areas with "high" grazing pressure, 16 cm in areas with "low" grazing pressure and 23 cm in areas with very little or no grazing pressure.

6. Pre-wildfire and unburnt quadrats had a higher cover of shrub species, bryophytes (including Sphagnum) and Cladonia lichens than burnt quadrats. Burning was associated with higher cover of bare peat and a slight increase in graminoid cover. Over the time period 2012-2014 initial recovery of plant functional groups took place in burnt areas, this was characterised by a decrease in bare ground and increasing cover of shrub species and bryophytes.

7. Recovery rates differed between ASSIs. For example the area of bare peat had returned to approximately baseline levels at Mullaghcarn, Slieve Beagh and Slieveanorra after 3.5 years but remained higher at Cuilcagh, Eastern Mournes and Glennasheevar. Large fluctuations were also evident in the recovery of Sphagnum cover with the largest difference between pre-burn Sphagnum cover and 2014 measurements at Cuilcagh and Glennasheevar.

8. Soils collected from burnt areas 1.5 years after wildfires produced a marginally lower species richness of seedlings and a lower abundance of non-*Calluna* seedlings in germination trials. Mean seed density in unburnt plots was 6,994 m^{-2} and in burnt plots was 5,750 m^{-2}.

9. Burning shifted plant communities away from those species typically associated with blanket bog, towards generalists and rapidly colonising species. Specifically, bog specialists including round-leaved sundew (*Drosera rotundifolia*), crowberry (*Empretrum nigrum*), bog asphodel (*Narthecium ossifragum*) and bryophytes including *Sphagnum* spp*., Hylocomium splendens, Rhytidiadelphus loreus* and *Racomitrium lanuginosum* were less abundant in burnt areas. However, common bilberry (*Vaccinium myrtillus*) which is a positive indicator species for blanket bog and wet heath and hare's-tail cotton grass (*Eriophorum vaginatum*) which are both positive indicator species for blanket bog had higher abundances in burnt areas. Generalist species such as wavy hair grass (*Deschampsia flexuosa*) were more abundant in burnt areas, as were pioneer acrocarp mosses including *Ceratodon purpureus* and *Campylopus introflexus*.

10. Burning was negatively associated with the abundance of all frequently observed *Sphagnum* spp.. This negative association with burning was strongest for *S. capillifolium* and *S. papillosum* which are key peat building species. Ecosystem services of carbon sequestration, water

purification and soil and water retention are likely to be negatively affected by the increased bare peat surface and the decreased cover of key peat forming *Sphagnum* spp. and other vegetation.

11. 37 species of ground beetles were recorded including the Northern Ireland Priority Species *Carabus clatratus* (found at Cuilcagh) as well as other species deemed "nationally scarce" in Great Britain such as *Carabus nitens* (Cuilcagh and Slievebeagh) and *Cymindis vaporariorum* (Glennasheevar). To our knowledge *Carabus nitens* had not been previously recorded at Slievebeagh ASSI nor *Cymindis vaporariorum* at Glennasheevar ASSI.

12. There was no difference in ground beetle species richness between burnt and unburnt areas. However, shifts in community composition were evident. Species associated with unburnt areas included the widespread common upland species *Abax parallelepipedus*, *Cychrus carabidoides* and *Agonum fuliginosum* and the common but more localised peat species *Carabus glabratus*. Burnt areas were associated with higher abundances of the widespread and common upland species *Carabus problematicus*, *Nebria salina*, *Nebria brevicollis* and *Pterostichus diligens*. Burnt areas were also associated with the much less common species *Carabus nitens*.

13. A total of 75 species of spider were recorded. Species richness of spiders was significantly higher in unburnt areas: however, spider richness increased in burnt areas in 2014 indicating some recovery. Unburnt communities were characterised by a higher abundance of some indicator species of good peat bog condition, namely *Pirata uliginosus* and *Trochosa spinipalis,* but also with some more less specialist species including *Trochosa terricola*, *Gonatium rubens*, and *Agroeca proxima.* Burnt areas were characterised by a higher abundance of *Antistea elegans* (family Hahnidae) which is also considered to be a good indicator for peat bogs in western Britain, and by more widespread species such as *Robertus lividus, Centromerita concinna* and *Xysticus cristatus.*

14. A total of 17 bird species were noted during bird counts. The most commonly observed species were meadow pipits (*Anthus pratensis*) and swallows (*Hirundo rustica*) which accounted for 77% and 12% of all observations respectively. There was no difference in the abundance of birds observed in burnt and unburnt areas. However, bird species richness was significantly lower at burnt quadrats than at unburnt quadrats and this was consistent across the three years of survey.

15. In this study we have focused on differences between burnt and unburnt areas over a short timeframe (1.0 - 3.5 years post-wildfire). Over this period recovery of habitats was observed in terms of vegetation height, functional plant groups, vascular plant community composition and the species richness of the spider fauna. Furthermore, there was no detectable difference between burnt and unburnt areas in a number of other parameters including the species richness of vascular plants, bryophytes and ground beetles or the total abundance of spiders or birds.

16. However, of particular concern are changes to bryophyte, ground beetle and spider communities where no recovery was evident over the time period of this study. Furthermore, bryophyte communities became more divergent from baseline communities over the course of this study, and were mainly characterised the increasing abundance of the alien species *Campylopus introflexus.*

17. Further research, over longer time-frames is required in order to determine whether these species communities will recover in the longer term. However, given the detrimental impacts on ecosystem services and species communities evident immediately post wildfires a precautionary approach is advisable to prevent wildfires and protect these key upland habitats.

CONTENTS

INTRODUCTION

The impact of wildfires on upland habitats is of increasing concern both from a scientific and conservation perspective. Climate change is likely to increase the risk of wildfires in temperate regions and alter the capacity of natural habitats to cope such events. In particular, increased temperatures and decreased precipitation in summer months in the UK and Ireland (Blenkinsop & Fowler 2007; Murphy & Fealy 2010) may contribute to a greater frequency and intensity of uncontrolled fires (Albertson et al. 2011). European, including UK, wildfires are predominantly anthropogenic in origin with more than 95% of wildfires started by people (Birot et al. 2009; McMorrow et al. 2009). Thus, wildfires in the UK and Ireland represent an interaction between people, landscape and climate which may be mitigated by management actions.

Upland heather moorland is of significant international importance and contains 13 vegetative communities which are recognised in the EC Habitats Directive [92/43/EEC], with six communities occurring in the Great Britain and Ireland only (Thompson *et al.,* 1995). Such plant communities provide important habitat for fauna, including a high diversity of invertebrate species (Usher, 1992), mammal species including the pygmy shrew (*Sorex minutus*) and Irish hare (*Lepus timidus hibernicus*), and bird species including a high density of skylarks (*Alauda arvensis*) and meadow pipits (*Anthus pratensis*) along with 8 other bird species protected under the EC Birds Directive [2009/147/EC] including red grouse (*Lagopus lagopus scoticus*), hen harrier (*Circus cyaneus*) and golden plover (*Pluvialis apricaria*).

Previous studies on the impacts of fire in the UK have focused mainly on prescribed burning on heathland sites which have a history of human influence through grazing, burning and cutting (e.g. Harris *et al.,* 2011; Davies *et al.,* 2010). Hence, there is little information available on the potential impact of wildfires on less intensively managed sites, particularly blanket bog. Additionally, previous work has focused mainly on vascular plant species and there is a lack of information on the potential impacts on bryophytes or on the invertebrate fauna which characterise upland habitats.

In the unusually dry spring and early summer of 2011, a total of 3,801 hectares of designated habitats within nine Areas of Special Scientific Interest (ASSIs) were burnt in Northern Ireland (Table 1). This report details the findings of a 3-year study carried out by Quercus, School of Biological Sciences, Queen's University Belfast, in conjunction with the Northern Ireland Environment Agency (NIEA) and the National Museum of Northern Ireland (NMNI). We take a broad-scale approach to the issue, examining impacts and recovery across 6 ASSIs (Fig 1) and multiple trophic levels (plants, invertebrate and bird species). In addition, we detail the impacts of burning on multiple soil chemistry parameters.
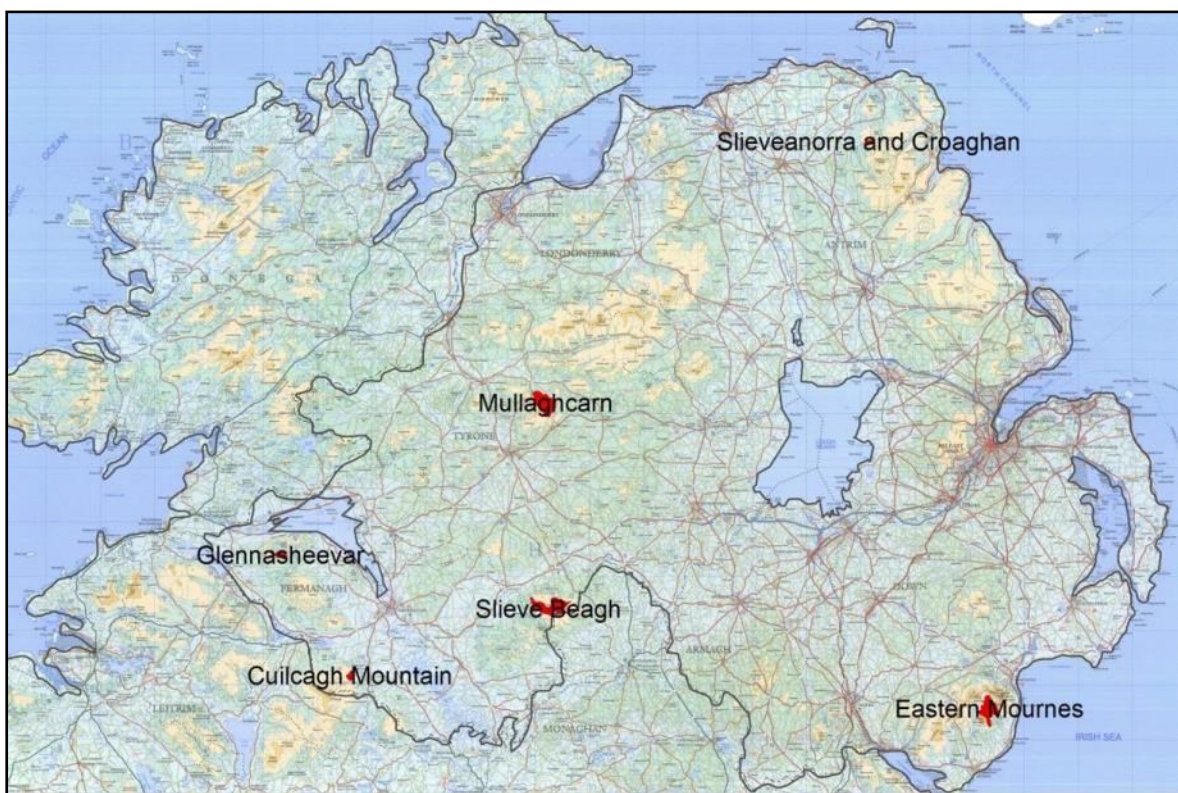

Fig 1 Map showing the burnt areas of six ASSIs which were selected for survey. Red areas indicate burning. Data extracted from the European Forest Fire Information System (EU Joint Research Centre 2012).

Table 1 Areas of Special Scientific Interest (ASSIs) on which wildfires occurred in the spring or early summer of 2011; showing area of fire and percentage damaged. Habitat and species selection features are also listed. Burnt area data was extracted from the European Forest Fire Information System (EFFIS).

ASSI	County	Area burnt (ha)	Percentage (%) of ASSI burnt	Designated habitat feature	Species feature
Camlough Quarry	Armagh	0.1	36		
Cuilcagh Mountain	Fermanagh	143.6	5	Blanket bog, Dry heath, Inland rock, Montane heath, Dystrophic lakes, Wet heath	Golden Plover, Higher plant assemblage, Invertebrate assemblage
Eastern Mournes	Down	931.1	12	Blanket bog, Dry heath, Inland rock, Montane heath, Oligotrophic lakes, Wet heath	Fungi assemblage, Higher plant assemblage, Invertebrate assemblage
Glennasheevar	Fermanagh	172.9	63	Blanket bog, Wet heath	Marsh Fritillary, Invertebrate assemblage
Lough Corry	Fermanagh	3.3	29	Oligotrophic lakes	
Mullaghcarn	Tyrone	1,119.9	54	Blanket bog, Wet heath, Dry heath, Oakwood, Dystrophic lakes	
Slieve Beagh	Fermanagh & Tyrone	1,166.6	61	Blanket bog, Dry heath, Dystrophic lakes	Invertebrate assemblage
Slieve Gullion	Armagh	160.4	26	Dry heath, Fens	Invertebrate assemblage, Breeding bird assemblage
Slieveanorra / Croaghan	Antrim	102.6	6	Blanket bog, Wet heath, Dry heath	Merlin, Hen Harrier
TOTAL / MEAN		**3,800.5**	**32**		

METHODS

Site and quadrat selection

GIS mapping was used to identify six Areas of Special Scientific Interest (ASSIs) within which large wildfires had occurred for inclusion in this study. These ASSIs were selected on the basis of containing areas of both upland heath and blanket bog habitats which had been burnt during 2011, namely: Cuilcagh, Eastern Mournes, Glennasheevar, Mullaghcarn, Slieveanorra/Croghan, and Slieve Beagh. The location and size of these fires was derived from satellite data using the European Forest Fire Information System (EFFIS, http://effis.jrc.ec.europa.eu; Fig 1 and Table 1). Information from pre-fire condition assessments, conducted by the Northern Ireland Environment Agency (NIEA), Department of Environment (DOE) was used to classify quadrats into three EU Annex I habitat classes using the Joint Nature Conservation Committee (JNCC) National Vegetation Classification (NVC) system for UK habitats (following Averis et al. 2004). Annex I habitats included were "blanket bog" (n=75), "Northern Atlantic wet heaths with *Erica tetralix*" (n=25) and "European dry heaths" (n=22). A total of 122 quadrats (2 x 2 m) were selected randomly from the quadrats which had been previously surveyed by the NIEA for inclusion in this study. Seventy-one (71) quadrats were in areas burnt during 2011 and 51 in nearby unburnt areas, such that burnt and unburnt areas were sampled within each site. The number of quadrats within each habitat was determined in proportion to the occurrence of that habitat at each site. The mean distance between quadrats within sites was 1.77 km (min = 0.07 km, max = 6.07 km). Details of site locations, habitats and dates of burning are shown in Table 2.

Survey methods

Soil chemistry

Five soil samples were taken using a cylindrical soil sampler (5 cm in depth and 6 cm in diameter) were taken at a minimum distance of 1m apart for soil chemistry analysis within the same quadrats as those used for vegetation surveys between October and November 2012. Calcium (mg/l), carbon (%), magnesium (mg/l), total nitrogen (%), available phosphorus (mg/l), potassium (mg/l), pH and sulphur (mg/l) content.

Table 2 Details of study site locations, characteristics and management. Data on dates of wildfires were supplied by the European Forest Fire Information System – EFFIS (http://effis.jrc.ec.europa.eu).

Site/Area of Special Interest (ASSIs)	Altitude range (m)	pH range	Habitats surveyed	Management	Dates of fire
Cuilcagh	198 - 396	3.9 – 5.8	Blanket bog Dry Heath Wet Heath	Low intensity cattle and sheep grazing at 100% of site	02/05/2011
Eastern Mournes	202 - 554	4.0 – 4.8	Blanket bog Dry Heath Wet Heath	Low intensity sheep grazing at 100% of site / recreational use (hill-walking)	10/04/2011 01/05/2011
Glennasheevar	184 - 245	4.1 - 4.3	Blanket bog Wet Heath	Low intensity sheep grazing at 100% of site	02/05/2011
Mullaghcarn	218 - 505	3.9 – 4.2	Blanket bog Dry Heath	Low intensity sheep grazing at 70% of site	23/04/2011 & 02/05/2011
Slieve Beagh	241 - 352	3.9 – 4.2	Blanket bog	Low intensity sheep grazing at 66% of site	01/05/2011
Slieveanorra	232 - 345	3.9 – 4.2	Blanket bog	Low intensity cattle grazing at 33% of site	22/04/2011

The first survey identified differences between burnt areas in terms of Calcium (mg/l) and available Phosphorus (mg/l). Therefore, repeat soil sampling to assess changes in these two elements was conducted between September and October 2014. In this second sampling period a subset of the original quadrats were used due to financial constraints (32 unburnt and 30 burnt). All soil chemistry analysis was conducted at the Agri-Food and Biosciences Institute (AFBI), Agriculture Food and Environmental Science Division, Newforge Lane, Belfast, Northern Ireland. Specific laboratory methods for chemistry analysis were as follows: Calcium and Magnesium were analysed on the same extract using Atomic Absorption Spectrophotometer, carbon and total nitrogen using aTruMac CN analyser (LECO Corporation, Michigan, USA), pH potassium and available phosphorus were using a Skalar method (Skalar San Plus Auto Analyser; Skalar Analytical, Breda, NL). Extractable Sulphur was analysed using Varian liberty ICP (Varian Inc., California, USA) (for full protocols see Murphy and Riley, 1986; Sinclair A. G. 1973).

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Plants

Quadrats were surveyed between June and October in 2012, 2013 and 2014, corresponding to approximately one, two and three years post-wildfires. Two additional quadrats were surveyed at Cuilcagh in 2013 to replace two previously unburnt quadrats at that site which were burnt in April 2013. Botanical surveys included more detailed information including a full plant species inventory of vascular plants and bryophytes (to species-level), and *Cladonia* lichens (to genus level) in addition to the reassessment of condition assessment variables.

Seed-bank

Seed-bank samples were taken using a cylindrical soil sampler (5 cm in depth and 6 cm in diameter) at three locations 1 m apart within each quadrat. This sampling regime accounted for potential aggregation of seeds on small spatial scales by taking multiple samples within quadrats and optimised efficiency by only sampling the top 5 cm of soil where most viable seeds occur (Putwain & Gillham 1990; Pywell et al. 1997). Soil samples were refrigerated between (2-5 °C) over-winter until the following April. Individual soil samples for seed-bank germination were not aggregated within quadrats.

To maximise germination, seed-bank samples were concentrated by passing soil through 4 mm and 0.250 mm mesh width sieves to remove large material and fine particulate matter (following Ter Heerdt et al. 1996). Seedling germination trials were initiated between the 8th and 16th of April 2013 by spreading individual soil samples on top of ericaceous compost (70% peat). An additional 60 control samples of pure compost were potted making a total of 426 samples (i.e. 60 control samples + 366 experimental samples; 3 samples from each of 122 quadrats). Sample pots were 7.5 cm in depth and sufficient compost was used such that the soil samples were level with the top of the pots. Pots were placed in trays with 1 cm depth of water to maintain adequate water supply. These were checked daily and manually watered from above as required. The positions of burnt, unburnt and control samples was randomly mixed,

and changed twice weekly to reduce potentially confounding position effects on germination. Seed-bank samples were germinated and grown under ambient light conditions in an unheated greenhouse for a period of 15 weeks after which all seedlings were identified, enumerated and removed. Soil samples were then vernalized by freezing at -20 °C for 7 days and returned to the greenhouse for germination. Seedlings which emerged following vernalisation were identified and enumerated after a further 10 week growing period.

Invertebrates

In 2012, a preliminary study of invertebrates was conducted at two ASSIs (Eastern Mournes and Slieveanorra/Croaghan). A total of 20 pitfall traps were set at each site and were open from August to October (40 days at Slieveanorra and for 33 days in Eastern Mournes). In the Eastern Mournes, two sites were used on the southern side of Slieve Binnian and 5 traps were set in burnt and unburnt ground within each. At Slieveanorra, two sites were also selected on either side of a road where one side had been burnt in 2010 and the other in 2011. Again, 5 traps were set in burnt and unburnt ground. Traps were filled with antifreeze (>95% ethlyene glycol, <5% diethlyene gylcol, <1% water) and covered with chicken wire mesh and plastic rain covers to reduce the chances of accidentally capturing pygmy shrews and to exclude rain. Ground beetles (Carabidae) and spiders (Arachnidae) were identified to species-level by Damian McFerran and Rachel Hamill (Cedar, Ulster Museum).

In 2013 and 2014, invertebrate pitfall trapping was extended to 6 ASSIs. These were the same as those on which the plant and bird sampling were conducted, namely: Cuilcagh Mountain, Eastern Mournes, Glennasheevar, Mullaghcarn, Slieveanorra / Croaghan, and Slieve Beagh. 20 pitfall traps were placed at each site (10 in burnt and 10 in unburnt areas). These were collected on a 4 weekly basis from late May to October, at all sites except Slieveanorra where trapping began in July as access permission was not granted at that site until after the Red Grouse (*Lagopus lagopus scoticus*) nesting season. The total number of traps emptied was *n*=560. Traps were

filled with non-toxic antifreeze (40-67% monopropylene glycol, 33-60% water) and emptied at 4 weekly intervals. Identification of invertebrate samples from 2013 and 2014 was conducted primarily by Ruth Kelly and Anna Hart, with assistance from

Damian Mc Ferran, Adam Mantell, Gillian Riddell, Claire McVeigh, Alice McPherson, Jeremy Adelard, Marion Chapalain and Amber Woods. Ground beetles were identified to species level for the survey months June, July, August and September of both years, other beetle species were also identified where possible but are not included in the analysis. Spider species were identified for the survey months June and September of both years. Verification of beetle specimens of each species was conducted by Roy Anderson.

Birds

Bird surveys were conducted at the same time as the vegetation surveys (i.e. between June and October). A 10 minute bird count was conducted at each quadrat recording species observed visually or heard singing within a 50m buffer.

Statistical analyses

Soil chemistry

The effect of burning on soil chemistry was examined using Generalised Linear Mixed Models (GLMMs). In 2012 (1.0 to 1.5 years post wildfires) differences in Calcium (mg/l), carbon (%), magnesium (mg/l), total nitrogen (%), available phosphorus (mg/l), potassium (mg/l), pH and sulphur (mg/l) were assessed. The results of this analysis showed differences in Calcium (mg/l) and available Phosphorus (mg/l) between burnt and unburnt areas. Therefore, these two chemistry parameters were reassessed using the same statistical method based on resampling conducted in 2014 (3 to 3.5 years post-wildfires). Modelling procedure was as follows; initially "Burning" (i.e. burnt/unburnt in 2011) and "Habitat" (i.e. blanket bog, dry heath, wet heath) and their interaction were fitted as fixed factors by Maximum Likelihood Estimation (MLE). "Site" was fitted as a random factor. Where residuals were not normally distributed, (based

on a Shapiro-Wilk test), new models were fitted using a gamma response distribution and a log link function. Models were then optimised by selecting the best subset of explanatory factors based on the model Akaike Information Criterion (AIC) values where the lowest value represented the most parsimonious model.

Vegetation height

Prior to analysis of vegetation six quadrats were removed from the dataset. Three were removed due to evidence of burning prior to the 2011 wildfires in the area and three were removed due to severe heather beetle damage in 2013 and 2014 (at Slieveanorra ASSI). This left 116 quadrats which were included in analyses (72 blanket bog, 20 dry heath and 24 wet heath).

Shrub height

Shrub height was compared between burnt and unburnt quadrats using two sets of Generalised Linear Mixed Model (GLMM). The first model assessed the 'initial impact' of wildfires and compared data collected in the 2012 survey (ca. 1 year post wildfires) to data collected prior to wildfires during NIEA condition assessment surveys (max 5 year pre-wildfires), data from unburnt quadrats was also included to provide information on changes in heather height that occurred in the absence of wildfires at the same sites. The second set of models "recovery", examined changes in heather height wildfires, using data from surveys conducted in 2012,2013 and 2014 (i.e. 1, 2, and 3 years post-wildfires).

Modelling procedure was as follows for both models. The response variable was vegetation height (cm). Initially, a global model was constructed including the explanatory fixed factors of "Burning" (burnt or unburnt) and "Habitat" (blanket bog, dry heath or wet heath) and covariates of "Year", "Altitude" (m), "Slope" (º from horizontal), "Solar heat load Index", and "Grazing". The interaction between "burning and grazing", "burning and habitat" and "burning and slope", were also fitted to account for potentially different effects of wildfires in difference contexts (i.e. on more or less grazed sites,

different habitats, or steeper slopes). In 'recovery' models the interaction between "burning" and "year" was also fitted to account for in differences in growth rates between burnt and unburnt quadrats. Topographical variables (i.e. Altitude and Slope) were extracted from GIS raster files at a 25m resolution. Heat Load was a measure of local temperature resulting from solar radiation, slope and aspect specifically designed for use in vegetation science (McCune & Dylan, 2002). Here,

the Heat Load Index was calculated according to Equation 3 in McCune & Dylan (2002) based on raster data at a 25 m resolution. This equation was chosen as it was most suitable for areas with slopes of less than 60º and latitudes of between 30-60º. Whilst this metric does not account for small-scale variation in temperature caused by, for example, local shading or surface reflectance, it provides a useful measure of differences in thermal environments at a landscape scale (e.g. between north and south facing, or shallow and steep slopes). Grazing intensity levels were assessed at each quadrat based on the quantity of dung present in each quadrat and was ranked on a three-level ordinal factor scale of "None" < "Low" < "High". Specifically, "None/very low" = 0 droppings, "Low" = 1-10 droppings, High > 10 droppings. All explanatory variables were rescaled to units of standard deviation prior to model fitting. "Quadrat" nested within "Site" was fitted as a random factor to account for variation explained by replication within quadrats and of quadrats within sites. Models were initially fitted using a Gaussian response distribution. Where model residuals were not normally distributed (based on a Shapiro-Wilk test), models were refitted by Laplace estimation with a gamma distribution.

Functional plant groups

Partial Redundancy Analysis (pRDA) was used to examine the response of plant functional groups (i.e. shrubs, bracken, herbs, graminoids, sphagnum, other bryophytes and Cladonia lichens) in terms of percentage cover. Cover measurements from pre-wildfire NIEA condition assessments and all three survey years (2012, 2013 and 2014) were included. Vegetation matrices were transformed using a Hellinger transformation in order to meet the assumptions of pRDA. Explanatory factors and

variables were the same as those used in the "recovery" model of vegetation height. "Site" was fitted as a "conditional" variable in the pRDA, meaning that differences between sites were accounted for by partial ordination, prior to fitting all other explanatory factors and variables. The most parsimonious pRDA model was selected based on model AIC values using a forward step-wise selection procedure. Additional explanatory variables were added to the null model where they significantly improved the model fit based on permutation testing (inclusion criteria $p < 0.05$).

Plant species

Species level analysis of the vegetation community was conducted on three different subsets of the data, each corresponding to a different part of the vegetation community and functional role. These three groups were; vascular plants, bryophytes other than sphagnum mosses (hereafter, "non-sphagnum" bryophytes), and sphagnum mosses. For each of these plant groups differences in species richness were assessed using GLMM's and differences in community composition were assessed using pRDA. For species richness explanatory factors and variables, and GLMM model selection procedure was the same as that described above for "recovery" of shrub vegetation height. Models were initially fitted using a Gaussian response distribution. Where model residuals were not normally distributed (based on a Shapiro-Wilk test), models were refitted by Laplace estimation with "Poisson", "negative binomial" , "zero inflated Poisson" and "zero inflated negative binomial" response distributions and the optimal response distribution was chosen based on the lowest model AIC value. For species community analyses explanatory factors and variables and pRDA model selection procedure were the same as those explained above for functional plant groups, except that data from pre-burn condition assessments was not used, as this was not available for all species.

Seed bank

Seed bank data were analysed following the similar same methods as those used for higher plant community data, except that year was not included as a variable in the

analyses as seed bank samples were all collected in the same year (2012). For further details on methods employed see Kelly *et al.* 2016.

Invertebrate communities

Two key groups were analysed from the invertebrate fauna, ground beetles (Carabidae) and spiders (Araneae). 2012 field surveys of ground beetle and spider abundances were conducted at two sites and are described in Kelly et al. (2013). In this report we have analysed a the much larger dataset based on surveys conducted at all ASSI's in 2013 and 2014 (for details on survey methods see above). Here, we present results based on the analysis of results from 5 sites: Cuilcagh, Glennasheevar, Mullaghcarn Slieveanorra, and Slievebeagh. Unfortunately, it was

necessary to exclude samples taken at the Eastern Mournes from the final analysis due to repeated disturbance of the pitfall traps during surveys.

Differences in abundance and richness of ground beetle and spider species were assessed using a GLMM approach. Differences in community composition were assessed using pRDA. Explanatory variables in both GLMM's and pRDA were "Burning" (Burnt/Unburnt), "Year" (2013/2014) and "Month" (June/July/August/ September for ground beetles and June/September for spiders). Interactions between "Burning" and "Month" and between "Burning" and "Year" were also fitted to account for potentially differing impacts in different seasons or differences burnt areas between years (e.g. recovery or increasing divergence). In GLMMs "Transect" was fitted as a random factor nested within "Site" to account for similarly between samples within transects and sites. In pRDA analysis "Site" was fitted as a "conditional" factor meaning that differences between sites were accounted prior to fitting all other explanatory factors. Model selection procedures for both GLMM and pRDA models were the same as those explained above for plant species analysis.

Birds

Abundance and species richness of bird communities were assessed using the same GLMM approach explanatory factors and variables as those explained above for vegetation height. One additional dry heath quadrat was removed prior to the analysis of bird data due to the close proximity (70m) to another quadrat (meaning that the bird counts conducted in a 50m radius around each quadrat overlapped). Hence, the number of quadrats in the bird dataset was *n*=115. Partial Redundancy Analysis of community composition was not conducted as meadow pipits (*Anthus pratensis*) accounted for the overwhelming majority of records at all locations surveyed.

RESULTS

Soil chemistry

Calcium and phosphorus concentrations were significantly higher at burnt than unburnt quadrats ca. 1.5 years post wildfires (p = 0.016 and p = 0.021). Furthermore, reassessment of calcium (mg/l) and phosphorus levels (mg/l) in 2014 showed that these remained higher in burnt than unburnt areas ca. 3.5 years post wildfires (p = 0.009 and p = 0.006 respectively). Phosphorus concentrations remained higher in burnt areas in both blanket bog and wet heath habitats, but did not differ significantly from unburnt areas in dry heath habitats (p = 0.035; Fig. 2a). There was no significant interaction between burning and habitat in calcium concentration models indicating that differences in calcium concentration between burnt and unburnt areas were similar across the three habitat types.

The six other soil chemistry in variables measures 1.5 years post wildfires; namely, carbon (%), magnesium (mg/l), nitrogen (%), potassium (mg/l), pH and sulphur (mg/l) showed no significant differences between burnt and unburnt quadrats after accounting for differences between habitats and sites.
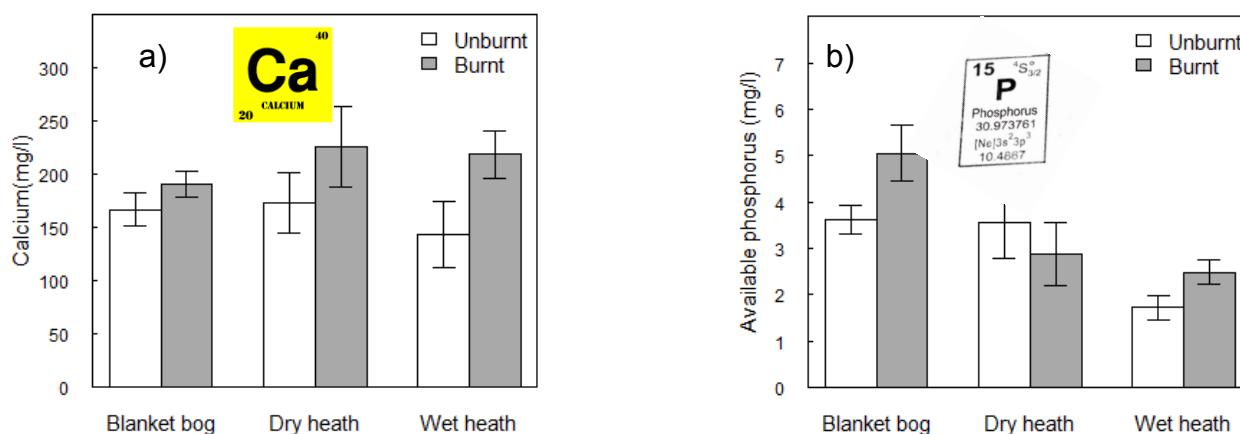

Fig. 2 Differences in soil chemistry between unburnt and burnt quadrats 3.5 years post wildfires, for **a)** calcium and **b)** phosphorus ± standard errors.

Shrub height

Two sets of models were conducted for shrub height, the first "the before-and after" model shows the initial impact of wildfires and compares pre-burn NIEA datasets with post-burn 2012 surveys, the second "the recovery model" examines the change in shrub height post-fire over the 3 year period from 2012-2014. The mean height of

heather prior to wildfires was 32.4 ± 1.9 cm in blanket bog sites, 35.7 ± 4.1 cm in dry heath sites and 31.0 ± 3.1 cm in wet heath sites. Before-and-after models showed that shrub height was significantly lower in areas where wildfires had occurred and the impact of wildfires on shrub height was greater in areas with steeper slopes ($p < 0.001$ in both cases, Fig 3i). Shrub height increased between over the 3 years of survey post-wildfires and height increased more quickly in areas where fires had occurred than in unburnt areas ($p = 0.026$ and $p < 0.001$ respectively, Fig 3ii and 4). In 2014, 3 to 3.5 years post wildfire, average heather height in burnt quadrats was 22.7 ± 1.1 cm in blanket bog sites, 15.0 ± 2.6 cm in dry heath sites and 16.5 ± 2.5 cm in wet heath sites. Heather height was also negatively associated with altitude and positively associated with slope in the pre-wildfire surveys (probably due to reduced grazing pressure in steeper areas) ($p < 0.001$ in both cases; Fig 3i). Grazing pressure was negatively associated with shrub height, with shrub height significantly lower in areas with high grazing pressure than in areas with no grazing ($p = 0.051$ in "before-and-after" models and $p < 0.001$ in "recovery models"; Fig 3 & 5). After 3.5 years average shrub height in areas with the "high" grazing pressure was 11.4 ± 1.1 cm, 16.4 ± 3.0 cm in areas with "low" grazing pressure and 23.1 in areas with very little or no grazing pressure. For comparison, prior to wildfires average vegetation height under the same grazing conditions was 28.3 ± 10.9 cm, 30.7 ± 3.9 cm and 33.2 ± 1.7 cm under "high", "low" of no grazing pressure respectively.

i) Initial impact (before-and-after wildfires)
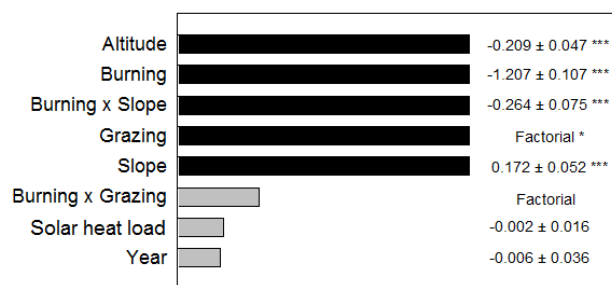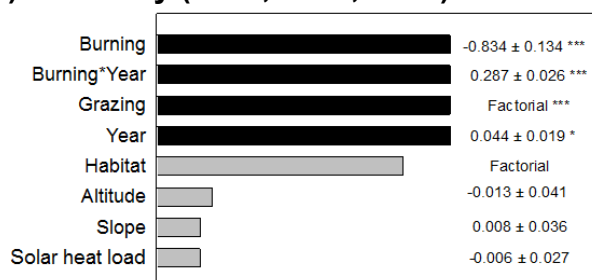

ii) Recovery (2012, 2013, 2014)

Fig. 3 Relative importance of explanatory variables in explaining variation in shrub height; i) initial impact based on before-and-after data and ii) recovery based on data collected on post-fire surveys in 2012, 2013 and 2014. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.
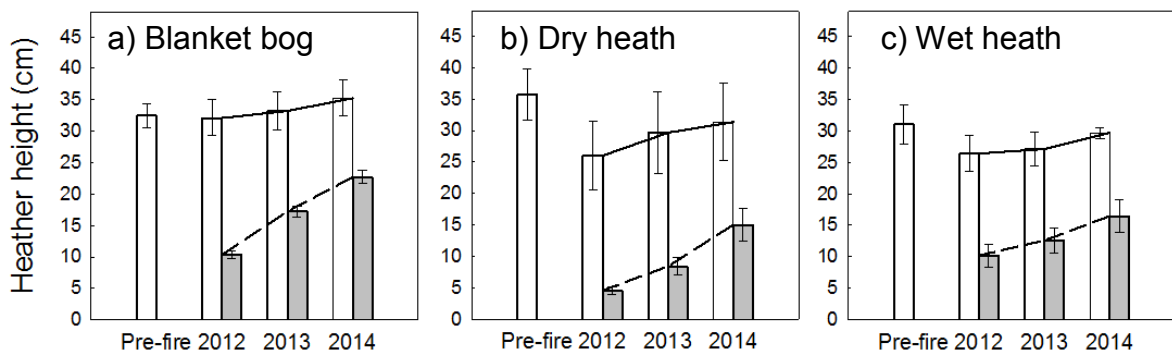
Fig. 4 Mean heather heights in burnt and unburnt quadrats in each survey year. White bars indicate unburnt quadrats, grey bars indicate burnt quadrats, error bars show standard error.
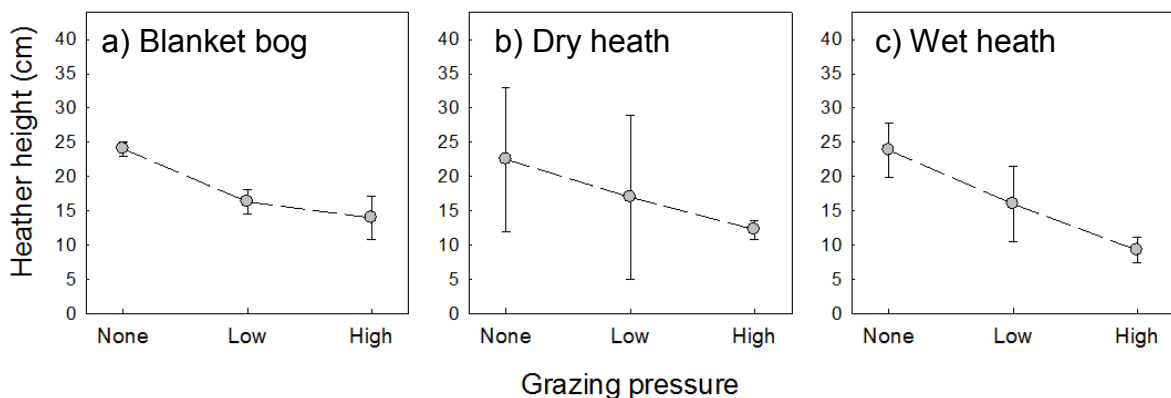

Fig. 5 Mean heather heights in burnt quadrats in the last survey year (2014), showing impact of grazing on vegetation height in burnt areas. Grey circles show the mean heather height at each level of grazing, error bars show standard error.

Functional plant groups

The most important variables explaining variation in functional plant groups were burning (p = 0.001), grazing (p = 0.001), habitat (p = 0.001), slope (p = 0.006) and the time period (i.e. pre-burn/2012/2013/2014) (p = 0.001). There was a significant interaction between time and burning (p = 0.001) and between burning and grazing (p = 0.001). These interactions indicate that the rate of change in functional plant groups differed between burnt and unburnt quadrats (burnt areas changed over time, whilst unburnt areas remained similar), and that grazing pressure affected burnt and unburnt areas differently. There was also an interaction between habitat and time, indicating that the rate of change in functional plant groups differed between habitat types (p = 0.004). The full model explained 25.2% of the variation in functional plant groups, after accounting for differences between sites. Burning and its interactions

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with grazing and time period, explained a total of 12.5% of the variation in functional plant groups.

Pre-wildfire and unburnt quadrats had a higher cover of shrub species, bryophytes (including sphagnum) and Cladonia lichens burnt quadrats. Burning was associated with higher cover of bare peat and a slight increase in graminoid cover. Over the time period 2012-2014 initial recovery of plant functional groups took place in burnt areas. This was characterised by increasing cover of shrubs and bryophytes including sphagnum. Grazing was primarily associated with a reduction in graminoid and sphagnum cover and an increase in bare peat, but also resulted in a higher cover of non-sphagnum bryophytes. When the impacts of grazing and burning occurred together, graminoid cover was lower than following burning alone (see Fig 6).
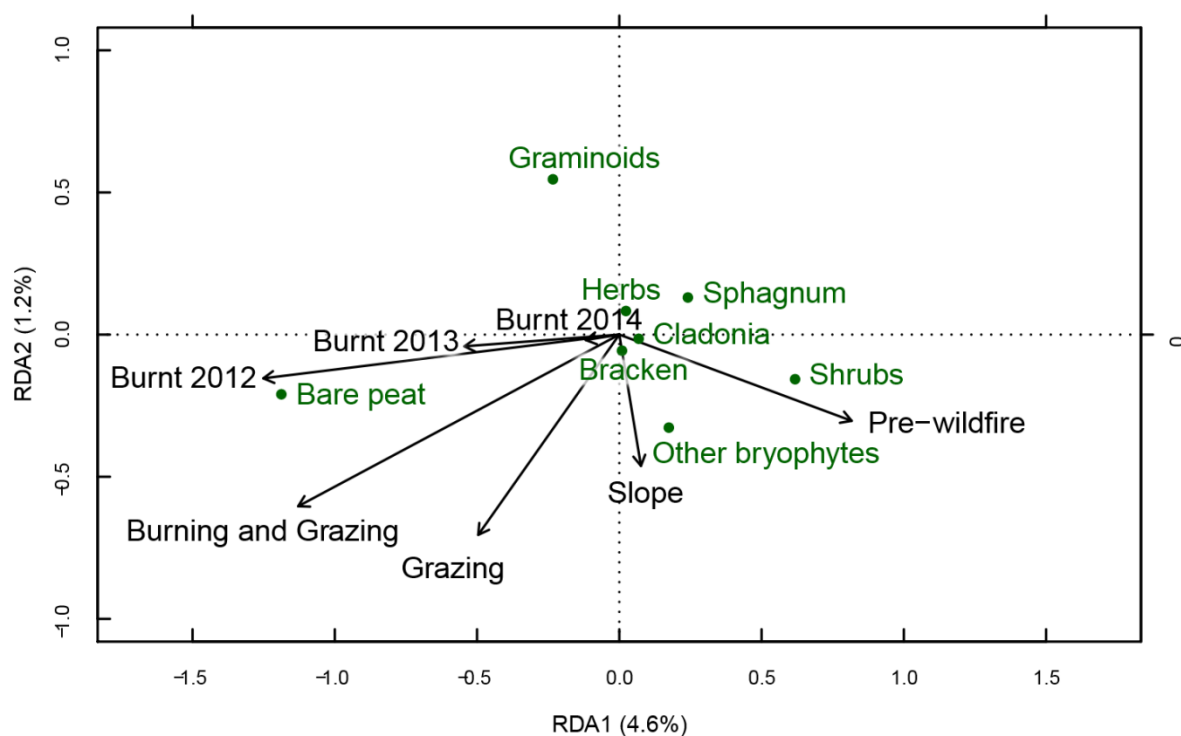

Fig. 6 Partial Redundancy Analysis (pRDA) biplot showing relationship between burning and grazing and the cover of functional plant groups, when site and quadrat were accounted for by pRDA.

Recovery rates also differed between ASSI's. For example, the area of bare peat had returned to approximately baseline levels at Mullaghcarn, Slieve Beagh and Slieveanorra after 3.5 years but remained higher at Cuilcagh, Eastern Mournes and

Glennasheevar. Large fluctuations were also evident in the recovery of sphagnum cover with the largest difference between pre-burn sphagnum cover and 2014 measurements evident at Cuilcagh and Glennasheevar (see Fig 7).
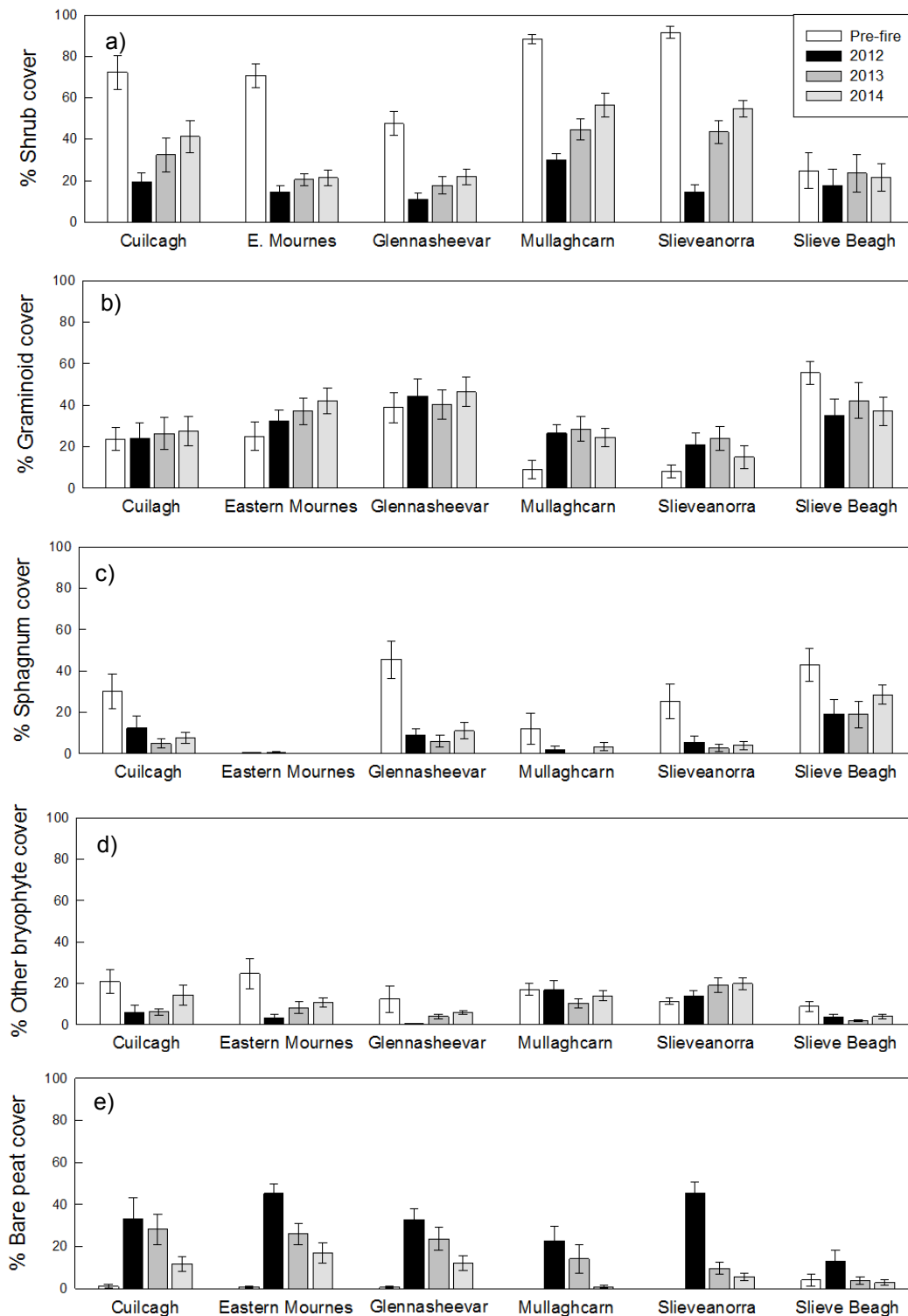

Fig. 7 Recovery of vegetation cover types at each ASSI; a) shrub cover, b), graminoid cover, c) *Sphagnum* cover, d) bryophyte cover and e) bare soil. White bar shows mean pre-survey cover in areas which were subsequently burnt, black bars show mean of burnt quadrats in 2012, mid-grey bars show mean of burnt quadrats in 2013, light grey bars show mean of burnt quadrats in 2014. Error bars show standard error.

Seed bank

In total, 6,369 seeds geminated from seed-bank samples. All 26 plant species that emerged from seed-bank soil samples, germinated prior to vernalisation by freezing. The most commonly occurring species in the seed-bank were Ling heather (*C. vulgaris*; present in 98% of quadrats), Cross-leaved heath (*Erica tetralix*; 62%) and Bell Heather (*Erica cinerea;* 34%). Only two species in the germinable seed-bank were not present in the vegetation, Toad Rush (*Juncus bufonius*) and Procumbent Pearlwort (*Sagina procumbens*). Thus, 24 species were common to both the germinable seed-bank and vegetation.
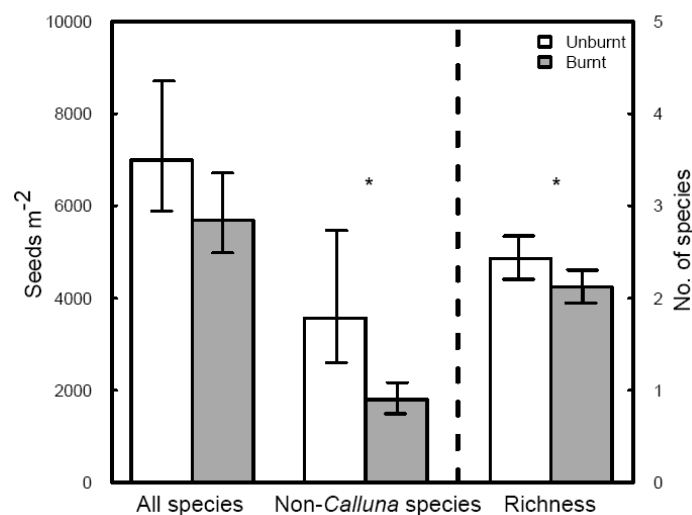

Fig 8. Mean seed abundance and mean species richness in burnt and unburnt areas. Error bars show bootstrapped 95% confidence intervals on the mean. * Indicates a significant difference between burnt and unburnt areas based on generalised linear mixed model outputs.

The total number of seedlings germinating was not significantly affected by whether soils had been subject to wildfires (unburnt mean = 6,994 m-2, 95% confidence = 5,889 – 8,710 m^{-2}, burnt mean = 5,662 m^{-2}, 95% confidence = 4,981 – 6,714 m^{-2}; Fig 8). The most important variable associated with total seedling abundance was slope, with which it was significantly negatively associated (Fig 9a). However, seedling abundance of species other than Ling heather (*C. vulgaris*) was negatively associated with wildfires (unburnt mean = 3,567 m^{-2}, 95% confidence = 2,598 - 5466 m^{-2}; burnt mean = 1,802 m^{-2}, 95% confidence = 1,490 – 2,168 m^{-2}; Fig 8). Non-*C. vulgaris* seedling abundance was significantly negatively associated with altitude (Fig. 9b). Burning was negatively associated with mean species richness (unburnt mean = 2.4, 95% confidence = 2.2 – 2.6; burnt mean = 2.1, 95% confidence = 1.9 –

2.3; Fig. 8). Seedling species richness was significantly positively associated with slope (Fig. 9c).
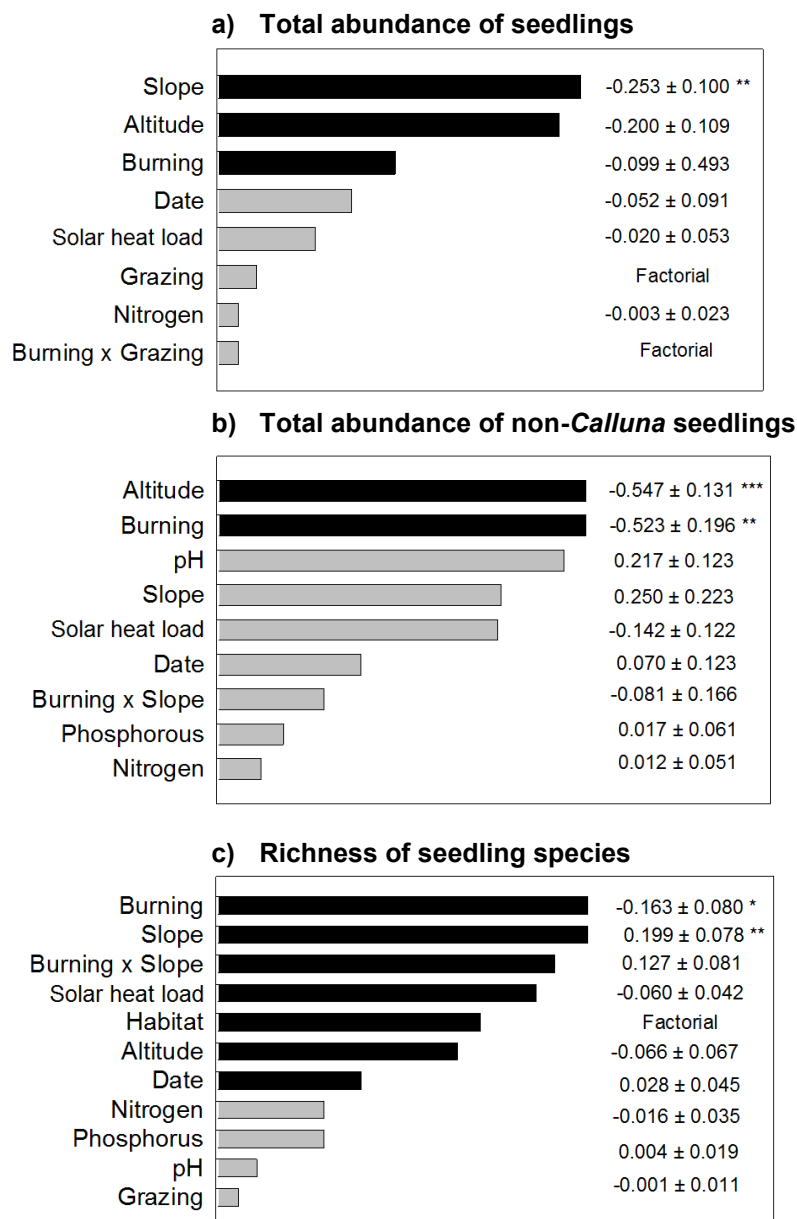
a) Total abundance of seedlings

Variable	Value
Slope	-0.253 ± 0.100 **
Altitude	-0.200 ± 0.109
Burning	-0.099 ± 0.493
Date	-0.052 ± 0.091
Solar heat load	-0.020 ± 0.053
Grazing	Factorial
Nitrogen	-0.003 ± 0.023
Burning x Grazing	Factorial

b) Total abundance of non-*Calluna* seedlings

Variable	Value
Altitude	-0.547 ± 0.131 ***
Burning	-0.523 ± 0.196 **
pH	0.217 ± 0.123
Slope	0.250 ± 0.223
Solar heat load	-0.142 ± 0.122
Date	0.070 ± 0.123
Burning x Slope	-0.081 ± 0.166
Phosphorous	0.017 ± 0.061
Nitrogen	0.012 ± 0.051

c) Richness of seedling species

Variable	Value
Burning	-0.163 ± 0.080 *
Slope	0.199 ± 0.078 **
Burning x Slope	0.127 ± 0.081
Solar heat load	-0.060 ± 0.042
Habitat	Factorial
Altitude	-0.066 ± 0.067
Date	0.028 ± 0.045
Nitrogen	-0.016 ± 0.035
Phosphorus	0.004 ± 0.019
pH	-0.001 ± 0.011
Grazing	

Fig. 9 Relative importance of variables explaining variation in **a)** total abundance of seedlings, **b)** total abundance of non-*Calluna* seedlings, and **c)** richness of seedling species based on model averaging. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Community composition of the seed bank differed between burnt and unburnt quadrats ($p=0.010$) after differences between sites were accounted for by pRDA, However, this only explained 1.4% of the variance in the seed-bank community. Variance in the seed-bank community was also explained by soil pH, soil

phosphorus and habitat type (*p* = 0.005, 0.010 and 0.060 respectively, and variance explained = 5.1%, 1.8% and 2.1% respectively). In the germinable seed-bank only two species Ling Heather (*C. vulgaris*) and Cross-leaved Heath (*E. tetralix*) showed increased germination in soil samples from burnt areas, whilst the majority of other species present in the germinable seed-bank including sedge, rush and grass species showed a negative association with burnt areas (Fig. 10). For further details of impacts of seed-banks initial vegetation regeneration at these sites see "The role of seed bank in the recovery of temperate heath and blanket bog following wildfires" (Kelly *et al.* 2016).

Fig. 10 The association between environmental variables and species composition of the soil seed-bank. Species are plotted where they occur in more than 1 quadrat and more than 5% percent of the variation in their abundance is explained by the pRDA model after differences between sites are accounted for.
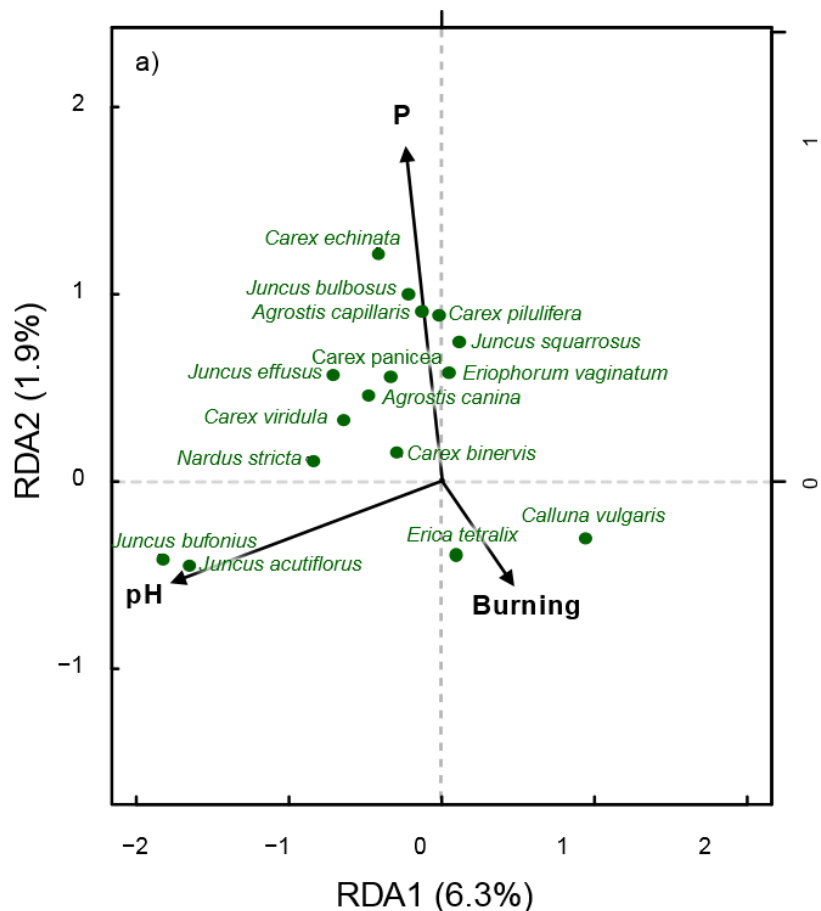
Vascular plants

A total of 59 vascular plant species were recorded, including 8 shrub, 4 tree, 11 grass, 11 sedge, 7 rush, 1 horsetail, 14 herb and 3 fern species. The median number of higher plant species per quadrat was 6 (min = 3, max =17). None of the species recorded were included in the Northern Ireland priority species list. Higher plant species richness did not differ significantly between burnt and unburnt areas (*p* = 0.262). Species richness differed between habitat types, being higher in dry and wet heath than in blanket bog (*p* < 0.001 in both cases). Dry and wet heath did not differ

significantly from one another in terms of their species richness ($p = 0.143$). Species richness was also negatively associated with "solar heat load" ($p < 0.001$) and positively associated year ($p = 0.013$) (Fig. 11).

Variable		Estimate
Habitat	████████████	Factorial ***
Solar heat load	███████████	-0.068 ± 0.022 ***
Year	████████████	0.051 ± 0.021 *
Burnt	█████████	0.053 ± 0.047
Slope	███████	-0.030 ± 0.442
Burnt x Slope	██	-0.002 ± 0.017
Altitude	██	0.001 ± 0.009

Fig. 11 Relative importance of explanatory variables in explaining variation in vascular plant species richness. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

The species composition of the higher plant community differed significantly between burnt and unburnt quadrats, and these differences were influenced by the interaction between burning and grazing ($p = 0.001$), burning and habitat ($p = 0.002$), burning and slope ($p = 0.015$) and burning and year ($p = 0.001$). Altitude and solar heat load also had a significant effect on plant community ($p = 0.001$ in both cases). Differences between sites explained 22.0% of the variation in the vascular plant community, and whilst all other environmental variables explained a further 18.6%.

Burnt areas had a lower abundance of ling heather (*Calluna vulgaris*), round-leaved sundew (*Drosera rotundifolia*), crowberry (*Empretrum nigrum*), bog asphodel (*Narthecium ossifragum*) and deer grass (*Trichophorum germanicum*) which are all positive indicator species for blanket bog and wet heath in the UK (JNCC, 2006). However, burnt areas were associated with an increase in the abundance of common bilberry (*Vaccinium myrtillus*) which is a positive indicator species for blanket bog and wet heath and hare's-tail cottongrass (*Eriophorum vaginatum*) which are both positive indicator species for blanket bog (JNCC, 2006). The abundance of the common grass species 'wavy hair grass' (*Deschampsia flexuosa*) was also associated with burnt areas. Grazing was negatively associated with deer grass and cross-leaved heath heather (*Erica tetralix*) a positive indicator species of blanket bog, wet heath and dry heath, and with hare's tail cottongrass (*Eriophorum*

vaginatum) which is a positive indicator species for blanket bog (JNCC, 2006). Grazed areas had an increased abundance of graminoids including velvet bent grass (*Agrostis canina*) and matgrass (*Nardus stricta*), and sedges such as pill sedge (*Carex pilulifera*) and green ribbed sedge (*Carex binervis*). Areas which were both burnt and grazed had common graminoid species associated with both factors (e.g. wavy hair grass, velvet bent grass, matgrass and pill sedge), but a lower abundance of the indicator species hare's–tail cottongrass, than burnt areas which had lower levels of grazing pressure. Recovery of the vascular plant community was evident over the three year period of this study, with at the plant communities moving in the direction of the baseline community composition (Fig. 12).
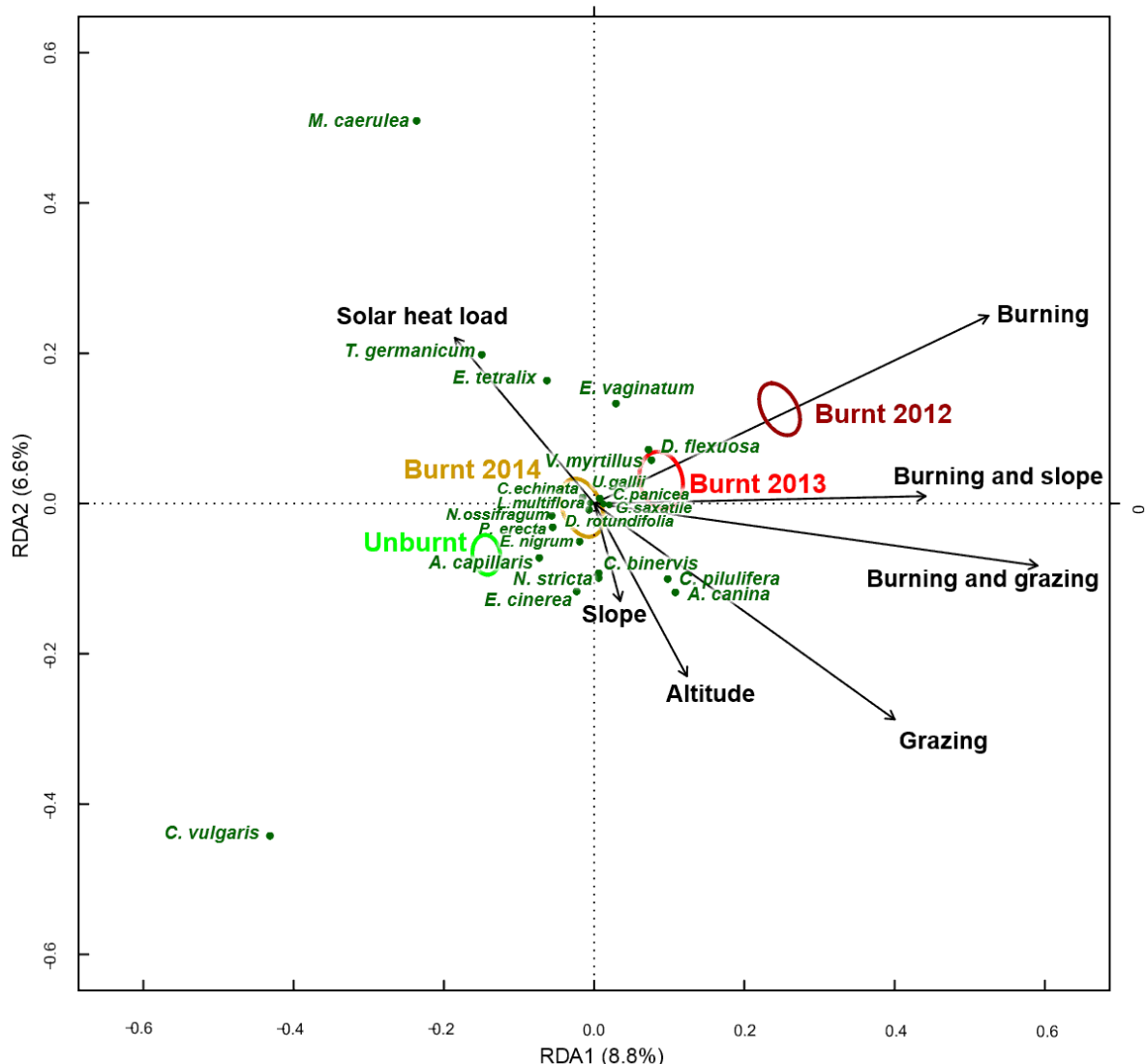

Fig. 12 The relationship between environmental variables and the vascular plant community, after accounting for differences between sites by pRDA. Plant species are plotted where they were present in more than 10 quadrats and more than 5% of their variation was explained by the RDA model are plotted. Coloured ellipses show standard error on the mean location of burnt quadrats in 2014 (dark red), 2013 (red) and 2012 (orange) and unburnt quadrats (green); illustrating the return of higher plant communities towards the baseline plant community in the 3 years after wildfires.

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Non-sphagnum bryophytes

48 species of bryophytes other than sphagnum were recorded, including 31 species of moss and 17 species of liverwort. The median number of bryophyte genera per quadrat was 3 (min = 0, max = 10). Richness of non-sphagnum bryophyte species was not significantly associated with burning, but was negatively associated with solar heat load (i.e. richness was higher in cooler areas) and increased between years (Fig. 13). However, burning significantly affected the community composition of non-sphagnum species (p = 0.001), indicating that although the overall number of species did not differ the type of species differed between burnt and unburnt sites (Fig 14).
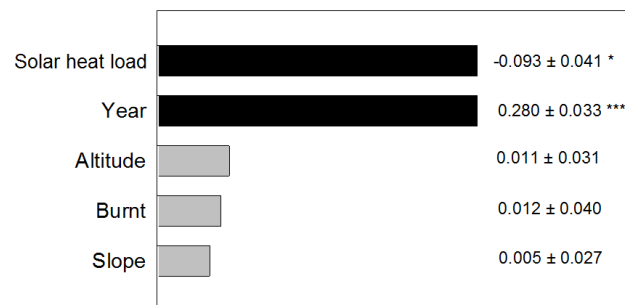

Fig. 13 The relative importance of variables in explaining variation in non–sphagnum bryophyte species richness based on model averaging. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

In addition to differences in community composition between burnt and unburnt areas and there was a significant interaction between burning and year (p=0.001), burning and slope (p = 0.019), burning and habitat (p = 0.001), burning and grazing (p=0.027); indicating that the response of species communities to burning depended on these other environmental variables. Of particular note, is the interaction between "burning" and "year" which was characterised mainly by an increasing abundance of the alien pioneer acrocarp species *Campylopus introflexus* in burnt areas (Fig. 14). Conversely, unburnt areas were characterised by pleurocarpous mosses including *Racomitrium lanuginosum, Rhytidiadelphus loreus* and *Hylocomium splendens* which are considered to be indicators of "good condition" in blanket bog and wet heath sites (JNCC, 2006). Burning was also associated with a higher cover of the cosmopolitan pioneer acrocarp *Ceratodon purpureus*, and a marginal increase in

Polytrichiatrum formosum and *Polytrichum commune*. Burning explained considerably less variation in the liverwort community (Fig. 14). The 3 most common liverworts in both burnt and unburnt quadrats were *Calypogeia muelleriana, Odontoschisma sphagni* and *Diplophyllum albicans.* Two liverwort species *Calypogeia muelleriana* and *Odontoschisma sphagni* were marginally more abundant in burnt areas. However, both were also found in unburnt quadrats are relatively common liverwort species in the upland flora. Altitude and solar heat load also had a significant effect on community composition (p = 0.001 in both cases). Differences between sites explained 11.3% of the differences in the bryophyte community, and the above environmental variables explained a further 17.0%.
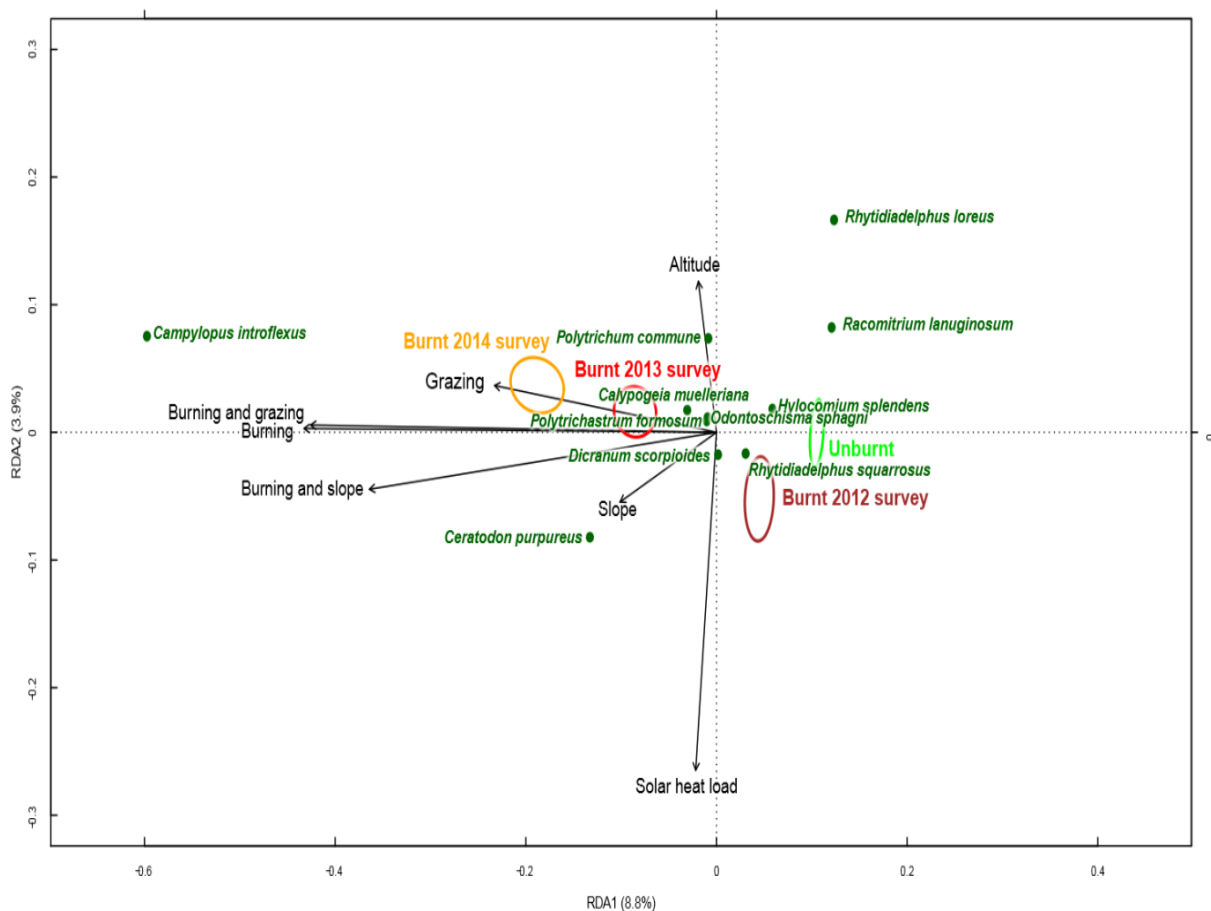

Fig. 14 Partial Redundancy Analysis (pRDA) plot showing relationship between environmental variables and non-sphagnum bryophyte cover after accounting for variation between sites by pRDA. Bryophyte species were plotted where they occurred in 10 or more quadrats in the study, and environmental variables explained more than 5% of the variation in their abundance. Coloured ellipses show standard error on the mean location in the biplot space of burnt quadrats in 2014 (dark red), 2013 (red) and 2012 (orange) and unburnt quadrats (green); illustrating that burnt quadrats did not return towards the unburnt bryophyte community composition over the 3 years of the study.

Sphagnum

Ten *Sphagnum* spp. were recorded. The median number of *Sphagnum* spp. per quadrat was 1 (min = 0, max = 5). The response of richness and diversity of *Sphagnum* spp. following burning was complex, with many factors contributing to the top set of models (Fig. 15). Overall, *Sphagnum* spp. richness did not differ significantly between burnt and unburnt areas (p = 0.271). The most important factor associated with *Sphagnum* species richness was habitat. As expected, *Sphagnum* species richness was significantly lower in dry heath habitats than in blanket bog (p < 0.001) or wet heath (p = 0.004), but did not differ significantly between wet heath and blanket bog (p = 0.418). In both wet heath and blanket bog the median species richness was 1 and the most commonly occurring species was *Sphagnum capillifolium.* Species richness also differed between years (p < 0.001). "Grazing", "slope and burning" and "slope" alone were also included in the top model, but were not significant when averaged across the top model set (p = 0.468, p = 0.391 and p = 0.837).
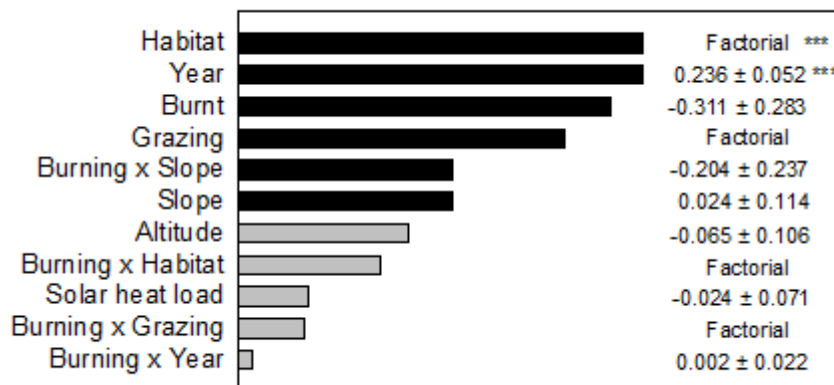

Fig 15. Relative importance of variables explaining variation in *Sphagnum* species richness. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

Sphagnum communities differed significantly between burnt and unburnt quadrats, and these differences depended on the habitat type (i.e. Dry heath, wet heath, blanket bog) (p = 0.001). The greatest differences were between unburnt and burnt quadrats on blanket bog habitats, where a higher diversity and abundance of *Sphagnum* spp. characterise the undamaged state of the habitat. Conversely, the smallest difference was seen between unburnt and burnt areas in dry heath, where *Sphagnum* spp. are always rare (Fig 13). There was no significant effect of "year", or

interaction "burning" and "year" in the final model, indicating that there was no evidence of a change (e.g. recovery) of sphagnum communities in burnt areas over the 3 years of survey. Differences between sites explained 26.4% of the variance in sphagnum communities and a further 7.1% was explained by the interaction of burning and habitat. Burning was negatively associated with the abundance of all frequently observed *Sphagnum* spp. (i.e. species found in more than ten quadrats; Fig. 16). This negative associations with burning was strongest for *S. capillifolium* and *S. papillosum* which are key peat building species. Four other species were found during field surveys: *S. palustre, S. denticulatum, S. fuscum* and *S. magellanicum* (present in nine, four, one and one quadrats respectively). *S. palustre* and *S. denticulatum* are relatively common species in the upland flora and showed no clear affinity for either burnt or unburnt areas. *S. fuscum* and *S. magellanicum* are more uncommon species of blanket bog habitats, *S. fuscum* was present in only one unburnt quadrat at Glennasheevar, whilst *S. magellanicum* was present in one burnt quadrat also at Glennasheevar. *S. magellanicum* was also noted outside of the formal survey quadrats in unburnt areas at Slieveanorra.
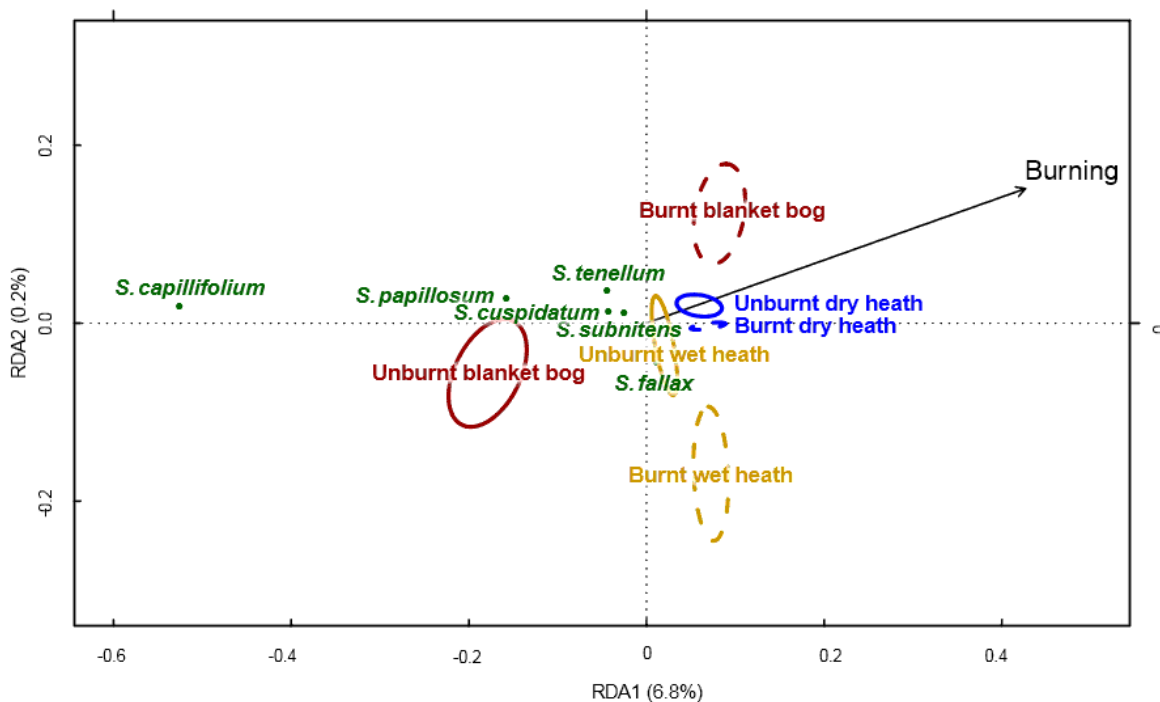

Fig. 16 Partial Redundancy Analysis (pRDA) plot showing relationship between Sphagnum species composition and burning, after differences between sites are accounted for by pRDA. Species which were present in more than 10 quadrats are shown. Coloured ellipses show standard error on the mean location in the biplot space of burnt and unburnt quadrats of each habitat type: blanket bog (dark red), dry heath (blue) and wet heath (yellow). The effect of "year" is not shown as there was no significant difference in communities in either burnt or unburnt quadrats over the 3 years of survey.
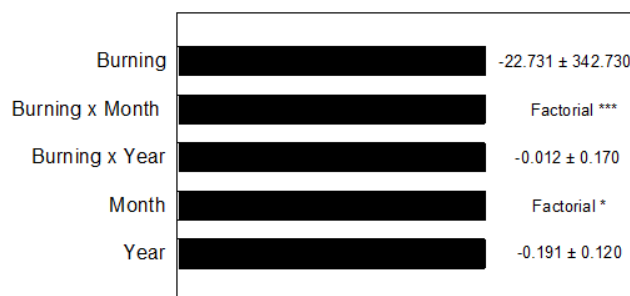
Invertebrate communities

The pilot study in 2012 conducted at 2 sites (Slieveanorra and the Eastern Mournes) suggested in initial increase in ground beetle abundance and significant differences in community composition of the ground beetle and spider faunal communities 1-year post-wildfires (Kelly *et al*. 2013). In 2013 and 2014, ground beetle communities were assessed at all six ASSI's in the study and the results of this work have further elucidated these changes. Here, we present results based on the analysis of results from 5 sites: Cuilcagh, Glennasheevar, Mullaghcarn Slieveanorra, and Slievebeagh. Unfortunately, it was necessary to exclude samples taken at the Eastern Mournes from the final analysis due to repeated disturbance of the pitfall traps during surveys.

Ground beetles (Carabidae)

37 species of ground beetles were recorded including the Northern Ireland Priority Species *Carabus clatratus* (recorded at Cuilcagh) other species deemed "nationally scarce" in Great Britain *Carabus nitens* (Cuilcagh and Slievebeagh) and *Cymindis vaporariorum* (Glennasheevar). To our knowledge *Carabus nitens* had not been previously recorded at Slievebeagh ASSI nor *Cymindis vaporariorum* at Glennasheevar ASSI.

Differences in the abundance of ground beetle communities between unburnt and burnt areas were strongly influenced by month ($p < 0.001$). In the summer months (June, July and August) ground beetle abundance was higher in unburnt areas than in burnt areas (mean abundance per trap in unburnt / burnt traps = 10.0 / 7.5, 10.8 / 8.4, 11.9 / 7.6 in June, July, and August respectively). However, at the end of the sampling season (September) ground beetle abundances were higher in burnt than unburnt areas, (mean abundance per trap in unburnt / burnt traps = 6.4 / 8.9). These differences most likely relate to changes in the relative utilisation of burnt areas by ground beetle species later in the season, rather than an overall change in ground beetle abundance. Abundance, also differed significantly between months irrespective of burning, such that beetle abundances were highest July and lowest in September (p = 0.033). Beetle species richness did not differ significantly between burnt and unburnt areas, or between years or months (Fig. 17).

a) Beetle abundance

Burning	-22.731 ± 342.730
Burning x Month	Factorial ***
Burning x Year	-0.012 ± 0.170
Month	Factorial *
Year	-0.191 ± 0.120

b) Beetle species richness

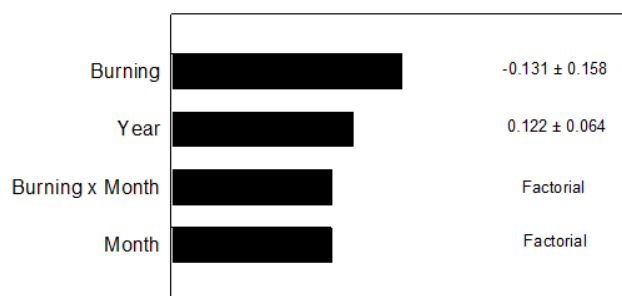|	
Burning	-0.131 ± 0.158
Year	0.122 ± 0.064
Burning x Month	Factorial
Month	Factorial

Fig. 17 The relative importance of variables in explaining variation in **a)** carabid beetle abundance **b)** carabid beetle species richness based on model averaging. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Ground beetle community composition differed significantly between burnt and unburnt areas, and there was a significant interaction between "burning and month" (p = 0.009) and "burning and year" (p = 0.020). This interaction between "burning" and "year" marginally increased the difference between species communities at burnt and unburnt sites in 2014 relative to 2013. Differences between sites accounted for 14.1% of the variance in ground beetle communities and a further 11.0% was accounted for by the combination of wildfires, month and year. Ground beetle species which were associated with unburnt areas included the widespread common upland species *Abax parallelepipedus*, *Cychrus carabidoides* and *Agonum fuliginosum* and the common but more localised peat species *Carabus glabratus*. Burnt areas were associated with higher abundances of the widespread and common upland species *Carabus problematicus*, *Nebria salina*, *Nebria brevicollis* and *Pterostichus diligens*. Burnt areas were also associated with the much less common species *Carabus nitens* (concidered "Nationally scarce" in Great Britain)

(Fig.18), this species was recently reported to show a strongly increasing trend across the UK potentially as a result of changes in climate or upland management (Brooks *et al.* 2012). No conclusions could be reached about the effect of burning on the priority species *Carabus clatratus* due to small sample sizes. However, the species was recorded in both unburnt and burnt patches within the single site where it occurred. The nationally scarce species *Cymindis vaporariorum* cannot be analysed for the same reasons, as this species was recorded in only one trap in an unburnt area of a single site. This species was only previously recorded in two locations in Northern Ireland (both in North Antrim), and the new record in Co. Fermanagh suggests that this species may be more widespread than previously thought due to low population sizes and under-recording of invertebrate taxa. It may be present but undetected in other peatland sites.
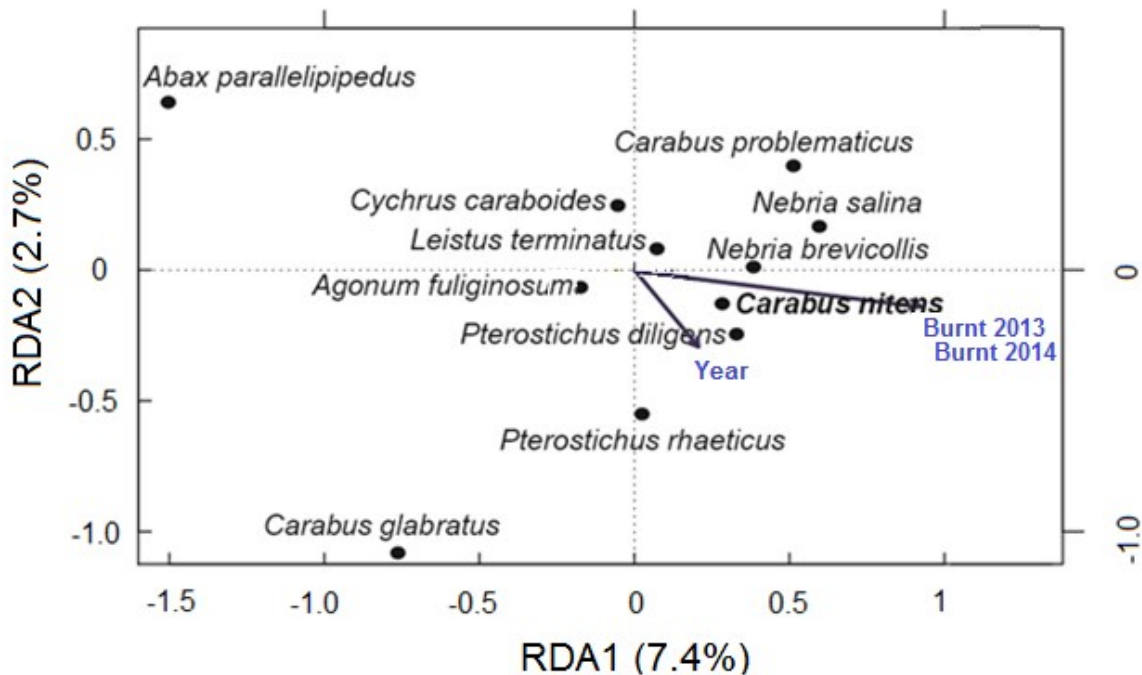

Fig 18 Partial Redundancy Analysis (pRDA) plot showing relationship between ground beetle species composition and burning, after accounting for differences between sites by pRDA. Species are plotted if they were present in more than 2 pitfall traps and more than 5% of the variance in their abundance was explained by the fitted RDA model, after accounting for differences between sites.

Spiders (Araneae)

A total of 75 species of spiders were recorded across the 5 ASSI's included in the invertebrate analyses. The median abundance of spiders per trap was 4 (min = 0, max = 47) and the median species richness was 2 (min = 0, max = 9).

There was no significant difference between spider abundance between burnt and unburnt areas or between years ($p = 0.277$ and $p = 0.510$ respectively). Spider abundance differed significantly between months being lower in September than June ($p < 0.001$). Species richness of spiders was significantly higher in unburnt areas ($p = 0.003$), but this difference was affected by year with spider richness in burnt areas having higher abundances in 2014 than in 2013, potentially indicating some recovery of species richness 2013 and 2014 ($p = 0.003$). Species richness of spider communities also differed between the 2 months surveyed and was significantly lower in September than in June ($p < 0.001$).

a) **Spider abundance**
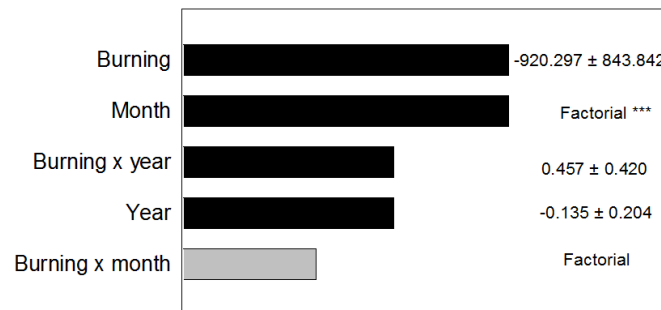
b) **Spider species richness**

Fig. 19 The relative importance of variables in explaining variation in **a)** spider abundance **b)** spider species richness based on model averaging. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

There was a significant association between burnt areas and spider community composition and this association differed between months ($p = 0.017$). Spider communities also differed between survey years ($p = 0.008$; Fig. 18), but there was no interaction of an interaction between "burning" and "year" (e.g. no evidence of recovery or deterioration of community composition in burnt areas between the 2013 and 2014). Differences between sites explained 8.9% of the variation in spider

communities and a further 7.0% was explained by the combination of the burning, month and year (Fig. 20).

Unburnt communities were characterised by a higher abundance of some wolf spider (*Lycosidae*) species, namely *Pirata uliginosus* and *Trochosa spinipalis* (in June) and *Trochosa terricola* (in September). *Pirata uliginosus* and *Trochosa spinipalis* have been previously noted to be a good peat bog indicator species in western Britain whilst *Trochosa terricola* which is common in peat bogs but also tolerates drier habitats (Scott et al. 2006). In September, unburnt sites were also characterised by a higher abundance of the *Gonatium rubens* (family Linyphiidae) and *Agroeca proxima* (family Liocranidae) which are common in the UK and found in a wide range of habitat types. Conversely, burnt areas were characterised by higher abundances of *Robertus lividus* (family Theridiidae) and *Centromerita concinna* (family Linyphiidae) which are considered appropriate for peatbogs in western Britain (Scott et al. 2006), but are also widespread and common in other habitat types. In September, burnt areas were characterised by a higher abundance of the *Antistea elegans* (family Hahnidae) which is also considered to be a good indicator for peat bogs in western Britain by Scott et al 2006, and by the common species *Xysticus cristatus* and is found in a wide range of habitat types.

Overall, there was no strong association between spider families and burning, with the two most abundance families wolf spiders (Lycosidae) and money spiders (Linyphiidae) occurring in both burnt and unburnt areas. Rather, associations with burnt or unburnt habitat types were found to be species-specific, for example, within the genus *Pardosa (*family Lycosidae)*, P. uliginosus* was found to be strongly associated with unburnt areas, whilst the abundant species *Pardosa pullata* was found equally frequently in both burnt and unburnt areas.
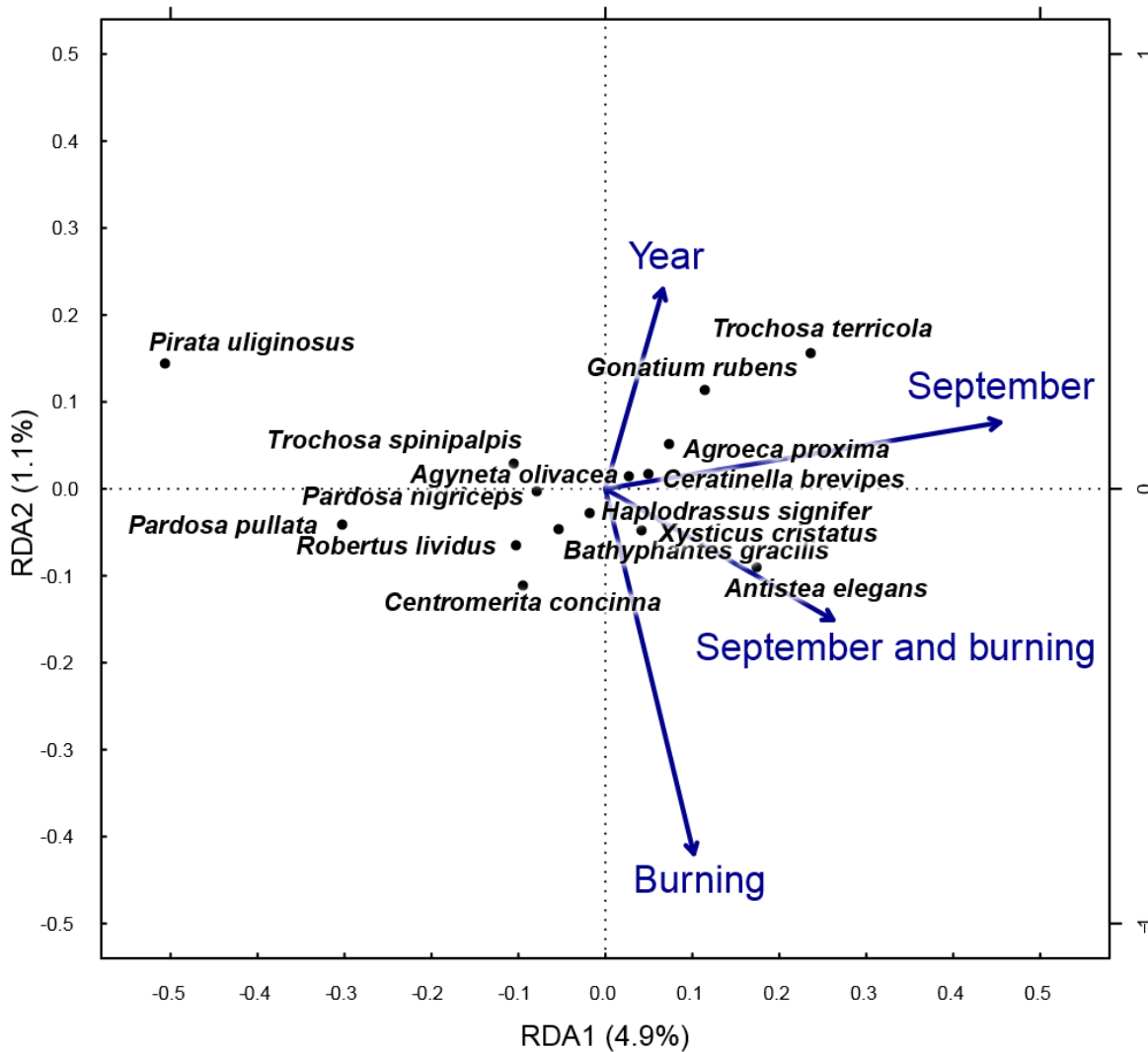
Fig. 20 Partial Redundancy Analysis (pRDA) plot showing relationship between spider species composition and burning, after accounting for differences between sites by pRDA. Species are plotted if they were present in more than 2 pitfall traps and more than 5% of the variance in their abundance was explained by the fitted RDA model, after accounting for differences between sites.

Birds

A total of 17 bird species were noted during bird counts (Table 2). The median species richness per quadrat was 1 (min = 0, max = 4). The most commonly observed species were meadow pipits (*Anthus pratensis*) and swallows (*Hirundo rustica*) which accounted for 77% and 12% of all observations respectively. There was a large amount of variation in the number of birds observed at each quadrat: the average number of birds observed at unburnt quadrats was 2.6 (min = 0, max = 15) and at burnt quadrats the average was 2.1 (min = 0, max = 21). Although burning was included in the top model for bird abundance it was not significant ($p = 0.295$).

However, the number of bird species was significantly lower at burnt quadrats than at unburnt quadrats (p =0.038), and there was no significant interaction between burning and habitat or burning and year, indicating that this effect on species richness was consistent across habitats and years. At unburnt areas in mean species richness was 0.88 (min = 0, max = 4) and at burnt quadrats the mean was 0.68 (min = 0, max = 3). Both abundance and species richness increased across survey years in both burnt and unburnt areas ($p < 0.001$, $p = 0.010$), presumably due to external factors affecting bird population numbers across sites (Fig. 21).

a) Total bird abundance

Variable	Value
Year	0.278 ± 0.079 ***
Burning	-0.176 ± 0.168
Grazing	Factorial
Altitude	-0.022 ± 0.061
Burning x Year	0.010 ± 0.054
Slope	-0.004 ± 0.033

b) Bird species richness

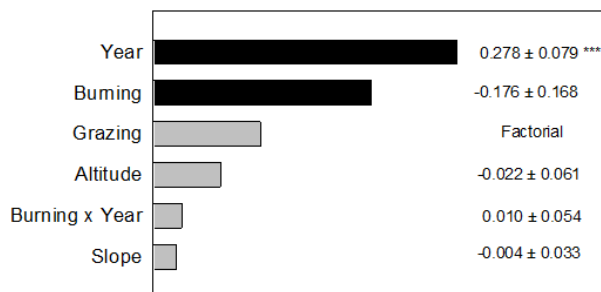| Variable	Value
Burning	-0.264 ± 0.127 *
Year	0.180 ± 0.071 **
Altitude	-0.101 ± 0.084
Burning x Year	0.008 ± 0.055
Solar heat load	0.004 ± 0.031

Fig. 21 The relative importance of variables in explaining variation in **a)** total bird abundance and **b)** bird species richness based on model averaging. Variables are ranked in order of the sum of their Akaike weights within the top set of models (i.e. models with ΔAIC <2). Black bars indicate variables included in the top model. Significance of individual model terms is indicated by * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

Table 3 Mean number of birds observed in 10 minute counts at unburnt and burnt bird quadrats in each year of survey.

Species	Mean abundance 2012		Mean abundance 2013		Mean abundance 2014	
	Unburnt	Burnt	Unburnt	Burnt	Unburnt	Burnt
Buzzard (*Buteo buteo*)	0.02	0.00	0.00	0.00	0.00	0.00
Hen harrier (*Circus cyaneus*)	0.00	0.00	0.00	0.00	0.02	0.00
Hooded crow (*Corvus cornix*)	0.00	0.00	0.00	0.00	0.08	0.00
Kestrel (*Falco tinnunculus*)	0.02	0.00	0.00	0.00	0.00	0.00
Meadow pipit (*Anthus pratensis*)	1.15	1.00	2.13	1.28	2.50	2.64
Raven (*Corvus corax*)	0.06	0.03	0.00	0.00	0.00	0.01
Redshank (*Tringa totanus*)	0.26	0.00	0.10	0.00	0.00	0.00
Rook (*Corvus frugilegus*)	0.00	0.00	0.00	0.00	0.25	0.10
Skylark (*Alauda arvensis*)	0.00	0.00	0.02	0.03	0.08	0.03
Stonechat (*Saxicola torquata*)	0.04	0.00	0.04	0.00	0.02	0.00
Swallow (*Hirundo rustica*)	0.34	0.21	0.06	0.25	0.35	0.43
Swift (*Apus apus*)	0.00	0.00	0.00	0.00	0.08	0.01
Wheatear (*Oenanthe oenanthe*)	0.00	0.00	0.02	0.01	0.00	0.00
Whinchat (*Saxicola rubetra*)	0.00	0.00	0.02	0.01	0.00	0.00
Willow warbler (*Phylloscopus trochilus*)	0.02	0.02	0.04	0.00	0.02	0.00
Wood pigeon (*Columba palumbus*)	0.00	0.00	0.00	0.00	0.00	0.03
Wren (*Troglodytes troglodytes*)	0.00	0.00	0.00	0.00	0.08	0.04
Total Species Richness	**8**	**4**	**8**	**5**	**10**	**8**
Total abundance	**1.91**	**1.26**	**2.44**	**1.60**	**3.50**	**3.31**

DISCUSSION

Burning during spring and summer 2011 had significant effects on the soil chemistry, seed bank, higher and lower plants, invertebrates and birds at six Areas of Special Scientific Interest (ASSIs) across Northern Ireland. Several of these differences could be considered as negative in light of conservation priorities. Ecosystem services of carbon sequestration, water purification and soil and water retention are likely to be negatively affected by the increased bare peat surface and the decreased cover of key peat forming sphagnum species and other vegetation. Furthermore, decreased abundance of key indicator species was observed, in addition to decreased species richness of some taxa and changes in overall vegetation and community structure (Table 4).

Table 4 Observed differences in burnt areas and their recovery within the 3.5 year time period of this study

Key differences in burnt areas of conservation concern	Evidence of start of recovery within survey period (2012-2014)
↑ Soil phosphorus (mg/l) and Calcium (mg/l)	NO
↑ Bare peat surface	YES
Changes in proportion of broad plant groups (e.g. shrubs, herbs, graminoids, bryophytes)	YES

↓ Key peatland indicator species
  1. Vascular plants
    • Ling heather, bog asphodel, round leaved sundew, crowberry & deer grass
  2. Non-sphagnum bryophytes
    • *Racomitrium lanuginosum, Rhytidiadelphus loreus & Hylocomium splendens*
  3. Sphagnum
  4. Spiders (Scott *et al.* 2006)
    • *Pirata uliginosus & Trochosa spinipalpis*
	1. YES
	
	2. NO
	
	3. NO
	4. NO

↓ Key peat forming *Sphagnum* species	NO
↑ Alien moss species *Campylopus introflexus*	NO
↓ Spider species richness	YES
↓ Bird species richness	NO

Soil chemistry

Burning changes the physicochemical composition of soils. Nutrient availability including phosphorus and micronutrients such as calcium increase immediately after fires but typically return to baseline levels within ca. 2 years (Certini, 2005). The majority of soil parameters including pH, nitrogen, carbon, potassium, magnesium and sulphur were either unaffected by the wildfires of 2011 (depending on their intensity) or any changes that had occurred have returned to their baseline state within 1.5 years. However, calcium and phosphorus concentrations remained higher in burnt quadrats than unburnt quadrats 3.5 years post wildfires. Calcium concentrations remained higher in burnt sites across all three habitat types, whilst phosphorus levels remained higher in blanket bog and wet heath but returned to the same level as unburnt plots in dry heath. These more long-term term effects on phosphorus concentrations in blanket bog and wet heath may result from differences in hydrological properties with greater nutrient leaching occurring in dry heath sites. These changes in phosphorus concentrations may have to have knock-on effects on key blanket bog and wet heath flora which are adapted to low nutrient conditions.

Vegetation structure

Vegetation height was strongly impacted by wildfires at all sites. Prior to wildfires average vegetation height in subsequently burnt areas was 32cm, in 2012 (ca. 1.5 years post-fire) shrub species including ling heather, bell heather, cross-leaved heath had recolonised burnt areas from seed and mean heather height was 9 cm. The recovery of vegetation height was strongly impacted by grazing such that in 2014 average shrub height was 11 cm in areas with the "high" grazing pressure, 16 cm in areas with "low" grazing pressure and 23 cm in areas with very little or no grazing pressure. The recovery of vegetation structure in terms of functional plant groups was also influenced by grazing pressure with a reduction in graminoid species in the post-wildfire succession associated with grazing pressure. These changes in vegetation structure, particularly height, are likely to affect the suitability of the habitat for

invertebrates and ground-nesting birds. We observed a reduced bird species richness in burnt areas in all survey years, with no detectable recovery in the number of bird species over the 3 year survey period.

Species communities

In this study we have taken a broad approach to the impacts of wildfires on upland sites, examining a broad-range of taxa. Our findings suggest that across all taxa differences in burnt areas were mainly characterised by changes in species-level community composition. Such changes were evident across all species groups examined; vascular plants, bryophytes, ground beetles and spiders. Furthermore, species richness did not differ significantly between burnt and unburnt areas for vascular plants, bryophytes or ground beetles. With the exception of *Sphagnum* spp., all taxonomic groups had species with higher abundances in burnt areas indicating that some species may take advantage of novel conditions post wildfires to expand their population sizes. Of particular note is the ground beetle species *Carabus nitens* which is classified as "nationally scarce" in Great Britain and was strongly associated with burnt areas in this study. However, this species was recently reported to show a strongly increasing trend across the UK potentially as a result of changes in climate or upland management (Brooks *et al.* 2012), for this reason it is currently unclear whether *Carabus nitens* will be a conservation concern in the UK in the future. The spider *Antistea elegans* was also found in higher abundances in burnt areas despite being considered characteristic of good condition peat bogs (Scott *et al*. 2006).

Overall plant communities shifted away from those characteristic of protected upland habitats, towards generalists and rapidly colonising species, allowing the establishment of certain invasive species (Table 4). Indicator species of blanket bog, wet heath or dry heath such as ling heather (*Calluna vulgaris*), crowberry (*Empretrum nigrum*), deergrass (*Trichophorum germanicum*), bog asphodel (*Narthecium ossifragum*) and bryophytes including *Sphagnum*, *Rhytidiadelphus* and *Racomitrium* species declined in burnt areas. However, two commonly used indicator species

common bilberry (*Vaccinium myrtillus*) and hare's-tail cotton-grass (*Eriophorum vaginatum*) did not show this pattern and were positively associated with burnt areas. Concurrently, generalist species such as purple moor grass (*Deschampsia flexuosa*) increased in burnt areas as did pioneer acrocarp mosses including *Ceratodon purpureus* and *Campylopus introflexus.* Grazing also increased the frequency of *Campylopus introflexus.* The main change in the bryophyte community composition of burnt areas over the period of this study was an increase in *Campylopus introflexus* and further divergence from community composition in unburnt areas with was primarily characterised by pleurocarpous mosses. *Campylopus introflexus* is an alien species to Europe from the southern hemisphere which was first introduced to the UK in 1941. Previous authors have suggested that it can out-compete other moss species and lichens following disturbance. However, as much of the previous research has been conducted in dune and alkaline grassland systems (Klink, 2010), further research is required on the implications of this species for upland habitats. One exception to this is research conducted on the competitive interactions of *Campylopus introflexus* with ling heather (*Calluna vulgaris*) the results of which were equivocal, suggesting that *Campylopus introflexus* reduces germination of *Calluna vulgaris* from the seed bank but increases growth rates once seedlings are established (Equihua *et al.,* 1993). After the 2014 field survey season, we intend to conduct a more detailed analysis to specifically examine the potential interactions of *Campylopus introflexus* with other upland flora.

Burning and grazing were negatively associated with the abundance of all frequently observed (i.e. species found in more than ten quadrats; Fig. 16). This negative association with burning was strongest for *S. capillifolium* and *S. papillosum* which were common in unburnt areas and are key peat building species. Results from studies of prescribed burning have suggested that hummock-forming species such as *Sphagnum capillifolium* may survive fires better due to their increased water retention (Peatscapes project, 2008). The reduction of *Sphagnum* spp*.* found here probably related to the higher heat intensity of uncontrolled wildfires.

Previous authors have noted that the prescribed burning of patches of upland may increase the overall species diversity of the site, by creating more open areas in heather dominated landscapes (e.g. Davies *et al.* 2010, Harris *et al.* 2011). However,

whilst the overall diversity of species within the site may increase the species composition shifts towards common pioneer species and away from characteristic upland species. Hence, it would be expected that the overall beta diversity of the landscape may decline post-fire due to the loss of species which are characteristic of protected upland habitats.

Previous work on the impacts of prescribed burning on areas of upland heath suggests that the recovery of a *Calluna*-dominated community occurs in upland heath habitats over a period of 20-25 years. However, many of these studies have been conducted following prescribed burning on previously degraded or intensively managed sites. It is unclear to what extent this can be generalised to large uncontrolled wildfires (Worrall *et al.* 2010). Here we have found some similarities with previous studies, such as the relatively rapid recovery of heather species (*C. vulgaris* and *E. tetralix*) and the increase in graminoid species post-wildfire (e.g. wavy-hair grass (*Deschampsia flexuosa*)). However, we have also found some differences from previous studies, in particular we the highlight potential negative impacts on key indicator species such as bog asphodel (*N. ossifragum*) and round-leaved sundew (*D. rotundifolia*) which are characteristic of wetter sites, and which, to our knowledge, were not present in previous studies of moorland burning in the UK. The absence of a focus on these species in previous studies may relate to shifting baselines, where species composition post-disturbance is being compared to an already reduced or altered species community.

In this study we have examined differences between burnt and unburnt areas over a short timeframe (1-3.5 years post-wildfire). Therefore, it was not possible to determine the long-term impacts of wildfires on species communities at these sites. However, of particular concern are changes to bryophyte, ground beetle and spider communities where no recovery is evident over the time period of this study. Further research, over longer time-frames is required in order to determine whether these species communities will recover in the longer term. However, given the detrimental impacts on ecosystem services and species communities evident immediately post wildfires a precautionary approach is advisable to prevent wildfires and protect these key upland habitats.

ACKNOWLEDGEMENTS

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