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Negative impacts of felling in exotic spruce plantations on moth diversity mitigated by remnants of deciduous tree cover

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Abstract:

Moths are a vital ecosystem component and are currently undergoing extensive and severe declines across multiple species, partly attributed to habitat alteration. Although most remaining forest cover in Europe consists of intensively managed plantation woodlands, no studies have examined the influence of management practices on moth communities within plantations. Here, we aimed to determine: (1) how species richness, abundance, diversity of macro and micro moths in commercial conifer plantations respond to management at multiple spatial scales; (2) what the impacts of forest management practices on moth diversity are, and (3) how priority Biodiversity Action Plan (BAP) species respond to management. BAP species were selected as they represent formerly widespread and common species, which have undergone substantial declines in the UK and Europe. We assessed moth communities in three conifer plantations in Northern England and Scotland by light trapping, combining local (e.g. age of planting) and landscape level (e.g. proximity to felled areas) characteristics to evaluate the impacts of forest management on moths. We found no relationship between local factors and moth richness, abundance and diversity but the amount of clear felling in the surrounding landscape had a strongly negative correlation. In contrast, the amount and proximity of broadleaf cover in the surrounding landscape positively influenced macro moth richness and abundance. For six BAP species, abundances were lower close to felled areas but increased with the size of adjacent broadleaf patches. We conclude that clear felling negatively affects moths, probably through alteration of habitats, the loss of larval host plants, and by limiting dispersal. A shift to continuous cover and maintaining broadleaf tree cover within plantations will greatly enhance their value for moth communities.

Keywords: Moth; Lepidoptera; abundance; species richness; plantation management; landscape heterogeneity

1. Introduction

Maintaining and restoring biodiversity is a key tenet in sustainable ecosystem management, the paradigm currently guiding habitat management practices across Europe and North America (Ober & Hayes 2010). This is driven by concern about world-wide declines in species and populations across a range of taxa (Dirzo et al. 2014) and recognition that much of this is driven by habitat loss and fragmentation, caused by anthropogenic change (Thomas 2004). In many countries the timber industry has responded to recognition of the importance of biodiversity by shifting focus from purely timber production to one which encourages sustainable practices that promote both wildlife conservation and sustainable timber yields (Macdonald et al. 2009). In Europe this has been driven by policy change initiated as a result of the Convention of Biological Diversity, requiring explicit consideration of environmental, economic and social objectives and a multi-purpose approach to forestry (Watts et al. 2008). However, efforts to assess the impact of forest practices can be challenging as there is often inadequate knowledge of the current distribution and abundance of many taxa in managed forest systems (Ober & Hayes 2010).

Plantation forests are generally considered poor for biodiversity as they are primarily composed of non-native tree species, often in monocultures, which are under an intensive management regime (Brockerhoff et al. 2008). However, they usually constitute the largest patches of tree cover in many European countries and as such may be valuable for preserving biodiversity if managed sympathetically. One of the few studies carried out at a national scale demonstrated that plantations can support diverse invertebrate communities in the UK, and that invertebrate community composition and abundance is most affected by tree species planted and geographic location (Humphrey et al. 2003). The structure of the plantation was also important for some groups: ground dwelling Carabid diversity decreased with canopy cover whereas overall Coleopteran richness and abundance in the canopy increased (Humphrey et al. 2003). The effect of stand age on invertebrate communities can also vary between taxa. Higher abundance and diversity of Coleoptera has been associated with older *Larix kaempferi* (Larch) and *Picea sitchensis* (Sitka Spruce) plantations in Japan and Northern Ireland due to increased heterogeneity and regeneration of native trees (Ohsawa 2005; Oxbrough et al. 2010). However, the high canopy cover in mature plantations can negatively affect other groups associated with open habitats (e.g. Arachnid diversity; Oxbrough et al. 2010).

Despite being a speciose taxonomic group and an important component of the invertebrate community, the impacts of plantation forestry on night active Lepidoptera are yet to be explored. Substantial declines of many moth species have occurred in the last few decades; two thirds of common and widespread species in the UK have suffered rapid population decreases (Conrad et al.

2006) with similar patterns occurring in Finland (Mattila et al. 2006) and Sweden (Franzén & Johannesson 2007). Rapid economic development, urbanisation, changes to silvicultural management and agricultural expansion have all been implicated in causing these declines (Conrad et al. 2006; Fox et al. 2013). Taken together, these studies provide overwhelming evidence that moths are facing declines on a large geographic scale, across a range of habitats, which mirrors similar effects found in less species rich groups such as butterflies and bumblebees (Warren et al. 2001; Goulson et al. 2008). Such losses are likely to have substantial effects at both higher and lower trophic levels. Moths are a key component of terrestrial ecosystems, providing ecosystem services through modification of ecosystem functioning by saproxylic species (Merckx et al. 2012), impacting upon plant growth through larval feeding activity, acting as pollinators and providing food for a range of taxa such as birds, small mammals and bats (Fox et al. 2013).

Intensified silvicultural practices have been suggested as one major driver of the decline in moth diversity and abundance (Fox et al. 2013). However, most studies have only focussed on the negative effects that a reduction in traditional deciduous forest management practices has had on lepidopteran species, and have not considered the role that non-native plantations may play. Reductions in deciduous forest management techniques such as coppicing and opening up rides have resulted in lower moth diversity by increasing structural complexity and changing botanical communities (Fox et al. 2013; Merckx et al. 2012; Warren & Bourn 2011). In general, moths associated with deciduous trees have declined throughout Europe, with larval host plant specificity a key factor in extinction likelihood in parts of Scandinavia (Mattila et al. 2006; Franzén & Johannesson 2007), whilst species associated with conifer trees have increased (Fox et al. 2013). Our current knowledge of moths in non-native coniferous plantations comes largely from studies which have focused on the management of pest species, and to the best of our knowledge no research has explicitly explored moth community composition and the impacts of forest management in exotic plantations.

Whilst little is known about the impacts of timber harvesting on Lepidoptera in non-native plantations, studies in native hardwood forests have suggested that effects are largely negative. In Indiana and Ohio, Summerville and Crist (2002, 2014) demonstrated that clear felling in native hardwood forests disrupted moth communities beyond the stand being felled, limiting the diversity of species able to persist within the landscape. Impacts of timber harvest on Lepidoptera can persist for up to 60 years (Summerville et al. 2009), although Summerville (2013) suggests that less intensive practices such as shelterwood harvest (removal of 15% standing wood) may support a higher richness and abundance of moth communities. In native conifer forests in Oregon, moth dominance and diversity was associated with greater canopy cover whereas richness was only affected by elevation, with higher species richness at lower elevations (Ober & Hayes 2010). These studies from North America demonstrate that managed native forest systems can support diverse lepidopteran communities, but the extent to which this is true in managed non-native plantations has not yet been examined. Specifically, in this study we aim to assess the impact of the following on moth abundance, richness, diversity and dominance in conifer plantations:

1. Influential, local scale plantation characteristics (e.g. age of planting, ground cover);
2. Proximity and prevalence of clear felling in the surrounding landscape;
3. Proximity and prevalence and of broadleaf tree cover within the surrounding landscape.

Since declining moth species might respond differently to the wider moth community, we examined the impacts of the above characteristics for moth communities as a whole, and separately for priority biodiversity action plan (BAP) species. These are formerly widespread and common species which have undergone population declines of between 70 – 90% in the last few decades, and as such are of particular scientific interest (Fox et al. 2013).

2. Methods

The study was conducted in three plantation forests in Central and Southern Scotland and Northern England (Figure 1). Widespread deforestation had already occurred in this area by the Holocene;

prior to the planting of the plantations in 1920 – 1940, the sampling areas would have consisted of open, upland moorland predominantly used for sheep grazing, with small patches of remaining broadleaf. The three forests were chosen for their large size (ranging from 30,000 ha in Cowal and Trossachs to 60,000 ha in Kielder and 114,000 ha in Galloway), high productivity and the predominance of *Picea sitchensis*, the most commonly planted and intensively managed coniferous tree species in the UK, and a common plantation tree species in Europe (Boye & Dietz 2005). Within each forest, multiple sites, a minimum of 4 km from each other, were selected using a Forestry Commission sub-compartment database within a Geographic Information System (GIS) (ArcMap 10.1, ESRI) based on stand (a unit of plantation management) age and species composition (Figure 1).

In total, seven sites were surveyed in Cowal and Trossachs, 12 in Galloway Forest and 12 in Kielder Forest. Where possible a stand of trees at each management stage was selected in each site, which was a maximum of $2km^2$ in size. Not all sites had all stands of each management age resulting in an unbalanced design of between four and six stands per site and a total of 285 stands across 31 sites. See supplementary data (4) for a description of the different stand types.

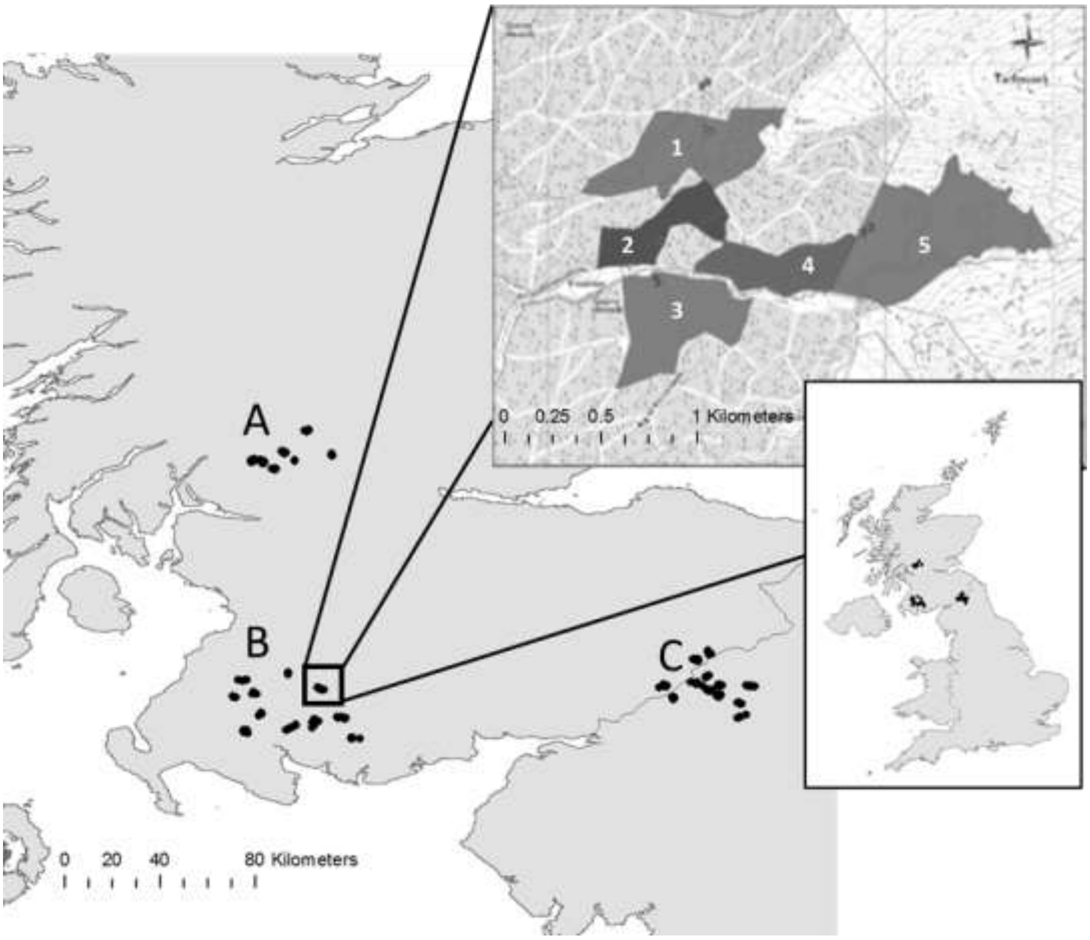

Figure 1. Location of field sites at three different study areas in (A) Cowal and Trossachs, South West Scotland, (B) Galloway, South West Scotland and (C) Kielder, Northern England. Stand types were as follows: Clearfell (1), Young (2), Thicket (3), Thin (4), Mature (5). See Supplementary data 4 for stand details.

2.1 Invertebrate trapping

Each site was surveyed for one night. Moths were trapped using portable 6W heath light traps using E7586 9'' actinic tube lights, powered with 12V batteries which were activated 15 mins after sunset and switched off after 4 hours (approximating the duration of the shortest night in the study area). This ensured that species flying at dusk and during the night were surveyed regardless of night duration. Species flying at dawn would most likely be missed as traps were often turned off before dawn. Surveys were only conducted on nights that were above 8^{o}c in temperature and wind speed of less than Beaufort 4, and were randomised as far as possible during the survey season between the different geographical areas. We recognise that surveying each site only once provides a coarse estimate of local moth assemblages; however, we are primarily interested in comparisons between stand types to identify potentially influential characteristics, which requires a large sample size. This same approach has been used to identify the influence of woodland characteristics on species richness, diversity and abundance of moth populations in both agricultural and urban landscapes (Fuentes-Montemayor et al. 2012; Lintott et al. 2014). In addition, previous studies have suggested that patterns of moth community composition remain consistent despite seasonal turnover (Summerville and Crist 2003).

Within each stand a heath trap was placed 15 metres from the edge, at least 200m from the next nearest trap and the location recorded with a GPS. Traps were selectively positioned to ensure that similar light levels were emitted (e.g. avoiding vegetation obscuring the light). In most cases, the traps were not visible from each other, apart from in felled stands. This may introduce a bias in traps at felled sites as the lights were visible from further away, reducing spatial independence (Lacki et al. 2007) although the attraction radii of heath light traps is commonly only between 10 – 30m depending on moth family (Truxa & Fiedler 2012). Any moths attached to the outside of the trap at the end of the trapping session were gently removed and released. A cotton wool ball soaked in ethyl acetate was immediately added to the trap and left overnight to kill trapped invertebrates. Macro moths were removed and pinned to boards for later identification and micro moths were separated for identification by an expert at the National Museum of Scotland. Approval for this work was obtained from the Ethical Review Committee within the Department of Biological & Environmental Sciences at the University of Stirling. Species data were shared with local moth recorders and added to the National Moth Monitoring Scheme (Fox et al. 2010).

2.2 Local habitat characteristics

We carried out vegetation surveys in two 0.01 ha plots at each stand type; due to the homogenous nature of stands these plots were considered representative of the stand as a whole. At each plot we recorded the total number of trees with diameter at breast height greater than 7 cm (stand density) and recorded the dominant ground cover (vegetated / non vegetated). Since dead wood is important for saproxylic moths we assessed the amount of dead wood on the forest floor using the following scale: 0 – no coarse woody debris, 1 – small twigs, 2 – large twigs and branches over 7cm in diameter, 3 – both large and small branches. Understory vegetation height was measured at 10 evenly spaced points across the radius of the circle and canopy cover was recorded at each point using a sighting tube with an internal crosshair; if the crosshair intersected with any canopy vegetation presence of canopy cover was recorded and converted to a percentage cover score (Lintott et al. 2015).

2.3 Landscape analysis

The GUIDOS toolbox (Soille & Vogt 2009) was used to determine percentage cover of core (more than 20m from the edge), and edge (patches within 20m of the edge) broadleaf tree cover woodland and felled patches within 4km of each moth trap by combining data from the OS Mastermap (EDINA, 2014) and a high resolution Forestry Commission database specific to the study areas. Distance to both broadleaf patches and felled areas as well as the size of the nearest broadleaf / felled patch were also recorded. It should be noted that broadleaf cover could be remnants of deciduous woodland cover from before the plantation was planted. Finally, the complexity of the broadleaf patch (a score of the total area of broadleaf / felled divided by the total edge area of broadleaf / felled) was calculated which approximates fragmentation (a highly fragmented area will have a high complexity score, see Appendix 1 for details on landscape variables included in analysis).

2.4 Statistical analysis

All analysis was carried out using R (version 3.4, R core development team) using the following packages: MuMIn, lme4, vegan, ggplot2. We used Margalef diversity to assess species diversity as it is straightforward to interpret and because it can deal with occasions where the number of individuals in a trap is equal to the number of species (Magurran 1988).

Many of the local and landscape variables were collinear so we used principle components analysis (PCA) to remove collinearity and reduce the number of predictors. Three separate PCAs were conducted for local characteristics and the felling and broadleaf tree cover metrics (See Supplementary data 1 for an explanation of the variables included in the PCA). For each PCA we retained those axes which explained more variation than random using the "broken stick" approach (Jackson 1993). For the local characteristics (Local PC), the first two axes explained 77% of the variation between stands; Local PC1 mainly described the stands with low canopy cover and high understorey vegetation height (which loaded low on PC1) and stands with low vegetation cover and high canopy cover (which loaded high on PC1), loosely catagorising different stand types (Supplementary data 2, Figure A). Local PC2 was driven largely by differences in altitude, describing the difference between the three different forests, with Galloway sites primarily at low altitudes, Kielder stands predominantly at high altitudes and Cowal and Trossachs falling in between. For felling characteristics (Felling PC), only the first axis explained more variation (63%) than chance; stands with low values of Felling PC1 were closer to patches of clearfell and surrounded by greater areas of felling in a 1km radius and those loading high on Felling PC1 were further from felled areas with less overall felling in a 1km radius (Supplementary data 2, Figure C). For characteristics relating to broadleaf woodland in the landscape (Broadleaf PC), only the first axis explained more variation (67%) than by chance; stands loading high on Broadleaf PC1 tended to be further from smaller patches of broadleaf woodland, with less broadleaf tree cover in the surrounding landscape whereas sites loading low on Broadleaf PC1 were closer to larger broadleaf patches, with more overall broadleaf tree cover in the surrounding habitat (Supplementary data, Figure B).

Using an information theoretic approach, we assessed the influence of stand and landscape variables on the **abundance** and **species richness** of macro and micro moths separately, using each metric per stand as the unit of replication. We used generalised linear models with a negative binomial error structure to account for overdispersion, and included an interaction between latitude and longitude as a fixed effect in all models to account for spatial autocorrelation. Models were validated by visual assessment of the residuals (Crawley 2007). Continuous variables were standardised and centred around a mean of zero and a standard deviation of 1 to allow direct comparisons of estimates, and model fit was assessed by comparing the change in AIC, retaining the best model (change in AIC greater than 2). McFaddens pseudo R^2 (McFadden 1974) was used to assess the amount of variation explained by each model. Local PC2 was not used, as this mainly described the difference in altitude between the stands and was collinear with date; in all cases simply using date was a better predictor. Models were fitted using either the stand type or the Local PC1, depending on model fit. We assessed the impact of felling and surrounding broadleaf tree cover on each response measure including either Felling PC1 or Broadleaf PC1 separately, then together and as an interaction. The same process was followed for **Margalef diversity** and **dominance** using a Gaussian error distribution. For each response measure, if there was no clear "top" model we averaged the coefficients across the top models in the set which accounted for a change in AIC of less than 2, using full averaged models to reduce the bias from explanatory factors which do not appear in every model (Burnham and Anderson 2002). Explanatory variables were considered to have a "significant" effect on the responses if the standard error of the estimate did not cross zero (Burnham & Anderson 2002). Micro and macro moths were analysed separately. Although the distinction between macro moths and micro moths is not taxonomically supported, micro moths typically have lower dispersal distances apart from some migratory species (Nieminen et al. 1999)

In addition to moth community measures outlined above, we modelled the influence of local and landscape characteristics on the occurrence of six of 13 **BAP priority species** recorded in the plantations. The following six species (*Eugnorisma glareosa* (Autumnal Rustic), *Arctia caja* (Garden Tiger), *Celaena haworthii* (Haworths Rustic), *Xestia castanea* (Neglected Rustic), *Ecliptopera silaceata* (Small Phoenix) and *Spilosoma lubricipeda* (White Ermine)) were present at the most sites and represented species which have declined between 70 – 90% over the last ten years (Conrad et al. 2006). We had insufficient data to model abundance at stand-level, so presence of these species was modelled using a binomial mixed effects model with species ID as a random intercept and Local PC1 as a random slope in order to assess species specific responses to stand level changes. We used the same approach as the previous analyses but here visual inspection of the data and subsequent model checking indicated that species occurrence was strongly and similarly associated with distance to felled areas and the size of broadleaf patches, so these were used in preference to the Felled and Broadleaf PC axis.

We graphically present the results for the single best model for each analysis including standardised parameters and standard errors for all explanatory variables. Inferences were made by comparing each parameter's standardised estimate with other predictor variables to assess its relative importance, the upper and lower 95% quantiles of each parameter obtained from N = 2000 simulated draws from the estimated distribution (Lintott et al. 2014) and a comparison of selected models using AIC.

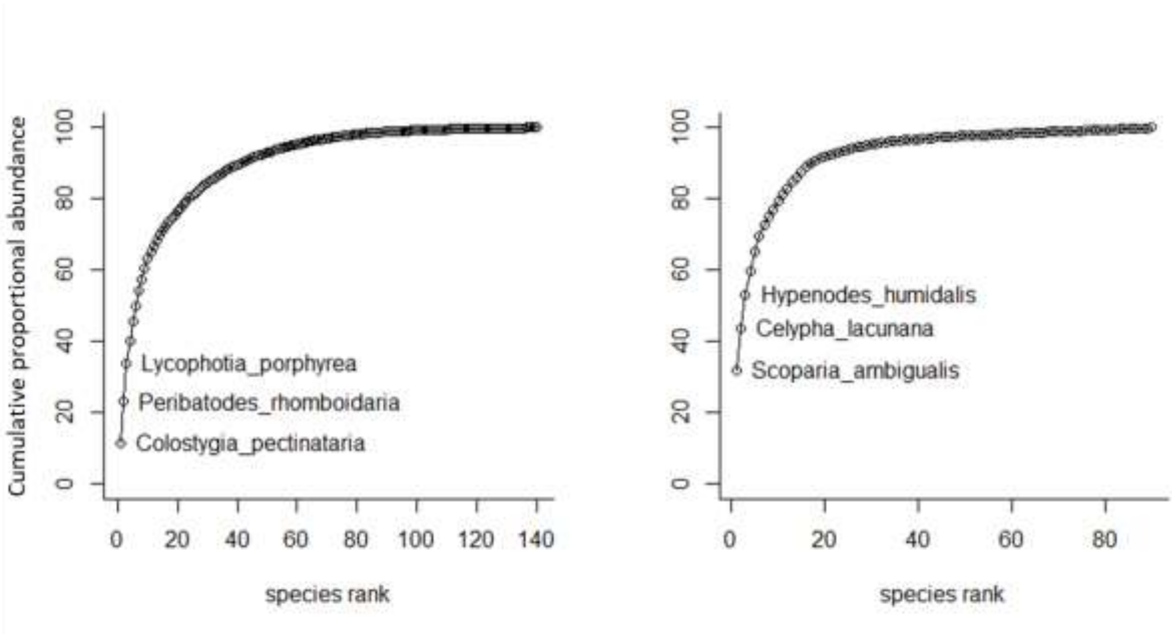

Figure 2. Species rank abundance curves for macro and micro moths considered separately. The three most abundant species are named. Rank abundances are given as cumulative proportions of total abundance.

3. Results

Composition of moth populations in commercial coniferous plantations

We collected a total of 8074 moths comprising 6464 macro moths belonging to 140 species and 10 families, and 1762 micro moths, belonging to 90 species and 19 families (Supplementary data 1) over 170 trap nights. Of these, 60% were generalist species while only 14% were woodland specialists and 26% were associated with open habitats (open specialists). We recorded an average of 38 (± 4.2) macro moth species and 10 (± 1.5) micro moth species per stand. Community composition was dominated by a few, highly abundant species such as the micro moth *Scoparia ambigualis* (Crambidae) and the macro moth *Colostygia pectinataria* (Geometridae), with less than 20% of micro moth species accounting for over 80% of all micro moths collected and 34% of macro moth species accounting for over 80% of all macro moths (Figure 2). We recorded 13 BAP priority species, with an average of 3.2 ±0.6 per stand.

3.1 Influence of local characteristics on moth communities

After accounting for date and temperature, there was relatively little correlation between local characteristics and moth communities (Table 1), with correlations between Local PC1 and macro moth abundance only. Abundance was highest in stands with a low Local PC1 score (low canopy cover and high understorey vegetation height), falling 53% in older stands with a closed canopy and lower understory vegetation height. Fewer moths of both groups were collected later in the season, with a similar pattern for species richness and diversity, but not dominance. Finally, the interaction between latitude and longitude influenced richness, abundance and diversity for micro moths but not macro moths (Table 1) revealing regional differences in species richness and abundance, with the highest abundance in Galloway plantation (27.0 ± 3) and lower in Kielder (13.0 ± 1.6) and Cowal and Trossachs (8.5 ± 1.3).

Table 1. Best approximating GLM's assessing influence of local, felling and broadleaf parameters on moth richness, abundance, diversity and dominance, conducted using an information theoretic approach with model averaging to assess importance of parameters. NA's indicate parameters not included in the top model sets. Dominant ground cover, coarse woody debris and the interaction between Felling PC1 and Broadleaf PC1 was never included in any top models and are not presented here. Parameters in bold are those which have a significant effect on response values, determined by whether the standard error of the estimate crosses zero (Burnham & Anderson 2002). Akaikes weight is the total weight explained by all models. Averaged estimates are presented ± the standard error.

		No. models averaged across	Intercept	Local PC1	Felling PC1	Broadleaf PC1	Date	Temp	Lat:Long	Akaike's weight
Macro moths	Sp. Richness	7	**264.9 ± 177.6**	**-0.70 ± 0.20**	**0.27 ± 0.08**	**-0.10 ± 0.00**	**-0.35 ± 0.11**	0.01 ± 0.05	0.27 ± 0.34	0.62
	Abundance	5	**3.21 ± 0.17**	**-0.26 ± 0.11**	**0.49 ± 0.12**	**-0.38 ± 0.17**	**-0.61 ± 0.19**	0.16 ± 0.16	0.21 ± 0.39	0.63
	Marg. Diversity	6	**2.01 ± 0.18**	-0.11 ± 0.12	**0.34 ± 0.11**	**-0.33 ± 0.14**	**-0.39 ± 0.15**	0.02 ± 0.07	0.30 ± 0.42	0.64
	Simp. diversity	9	**1.22 ± 0.19**	-0.04 ± 0.12	-0.03 ± 0.10	NA	0.08 ± 0.21	0.04 ± 0.11	NA	0.27
Micro moths	Sp. Richness	4	**1.23 ± 0.15**	-0.02 ± 0.06	**0.24 ± 0.09**	**-0.28 ± 0.11**	**-0.10 ± 0.13**	**0.33 ± 0.10**	**1.54 ± 0.40**	0.77
	Abundance	2	**2.28 ± 0.20**	NA	**0.47 ± 0.12**	**-0.45 ± 0.17**	**-0.32 ± 0.25**	**0.42 ± 0.15**	**1.93 ± 0.50**	0.73
	Marg. Diversity	4	**1.07 ± 0.10**	-0.01 ± 0.03	**0.13 ± 0.06**	**-0.14 ± 0.10**	**-0.13 ± 0.10**	**0.30 ± 0.07**	**0.86 ± 0.24**	0.65
	Simp. diversity	13	**0.75 ± 0.17**	-0.04 ± 0.11	NA	0.01 ± 0.05	-0.03 ± 0.11	0.10 ± 0.16	NA	0.43

Table 2: Best approximating GLM's assessing influence of local, felling and broadleaf parameters on BAP moth species probability of being detected. These were conducted using an information theoretic approach with model averaging to assess importance of parameters. NA's indicate parameters which were not included in the model. Dominant ground cover, coarse woody debris and the interaction between Felling PC1 and Broadleaf PC1 was never included in any top models and is not presented here. Parameters in bold are those which have a significant effect on response values, determined by whether the standard error of the estimate crosses zero (Burnham and Anderson 2002). Akaikes weight is the total weight explained by all models. Estimates for the full averaged model are presented ± the standard error. Estimates provided for the top 7 models, with a change in AIC of less than 2. The same variables as for the overall moth communities were originally used but inspection of the broadleaf and felling PC output showed that the main relationships were with specific components of the principle components.

	Intercept	Size of nearest broadleaved patch	Altitude	Distance to felled stand	Lat:Long	Local_PC1	AICc	Akaikes weight
Averaged Model	**-2.88 ± 0.25**	**0.22 ± 0.09**	-0.04 ± 0.10	**0.16 ± 0.14**	0.02 ± 0.09	-0.44 ± 0.35		0.50
1	-2.95	0.22	NA	0.22	NA	-0.59	479.90	0.13
2	-2.92	0.22	NA	NA	NA	-0.53	480.92	0.08
3	-2.97	0.23	-0.13	0.24	NA	-0.59	480.96	0.07
4	-2.67	0.21	NA	0.21	NA	NA	481.17	0.07
5	-2.96	0.23	NA	0.20	0.09	-0.59	481.53	0.06
6	-2.98	0.27	-0.22	0.21	0.18	-0.59	481.56	0.05
7	-2.67	0.21	NA	NA	NA	NA	481.82	0.05

3.2 Influence of felling on moth communities

There appeared to be a large, negative impact of clear felling on species richness, abundance and diversity for both macro- and micro moths (Figure 3, Table 1). Macro moth species richness declined from 13.4 (9.3 – 19.4) in sites furthest from clear felled areas and with less felling within 1km to 4.0 (2.5 – 6.6) in sites nearest to felled areas or surrounded by more felling in 1km. Similarly, micro moth species richness fell from 4.2 (2.9 – 6.2) to 1.5 (0.9 – 2.5) in sites close to felling or with a greater proportion of felling in the surrounding landscape (Figure 3 A, D).

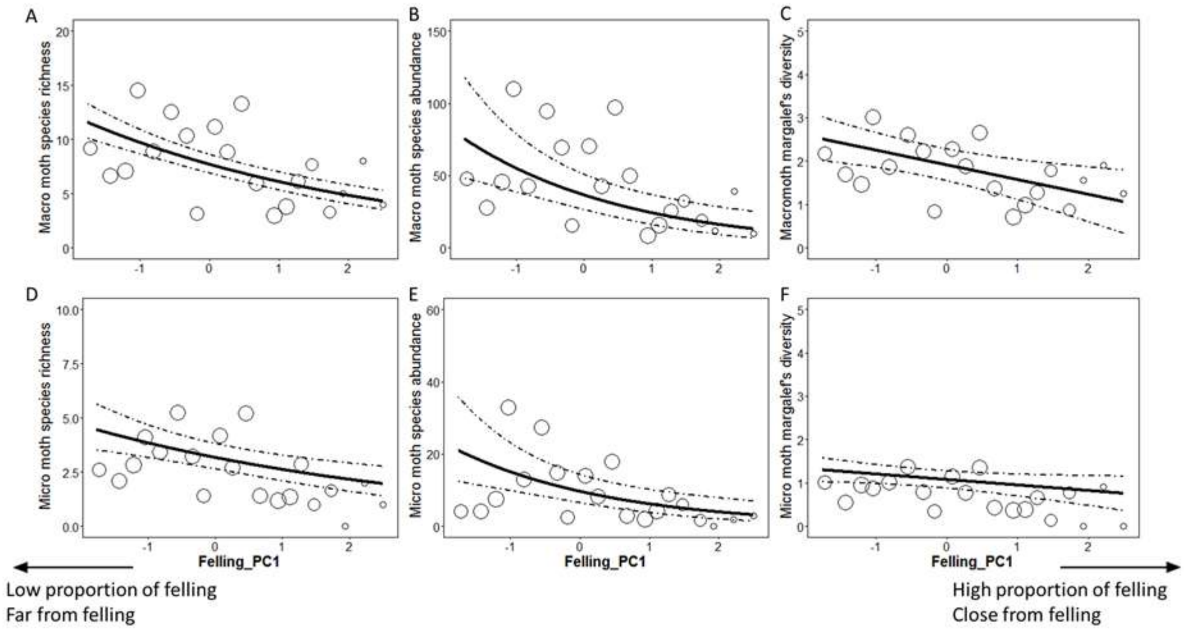

Figure 3. Impacts of felling on (A – C) Macro moth species richness, abundance and diversity and (D – F) Micro moth species richness, abundance and diversity per site. Different scales are used for abundance and richness due to higher richness and abundance in macro moths compared to micro moths. Original data on richness, abundance and diversity are superimposed as grey circles with diameter proportional to the number of sampling points where mean values occurred. Dashed lines represent 95% confidence intervals around the predictions (solid line).

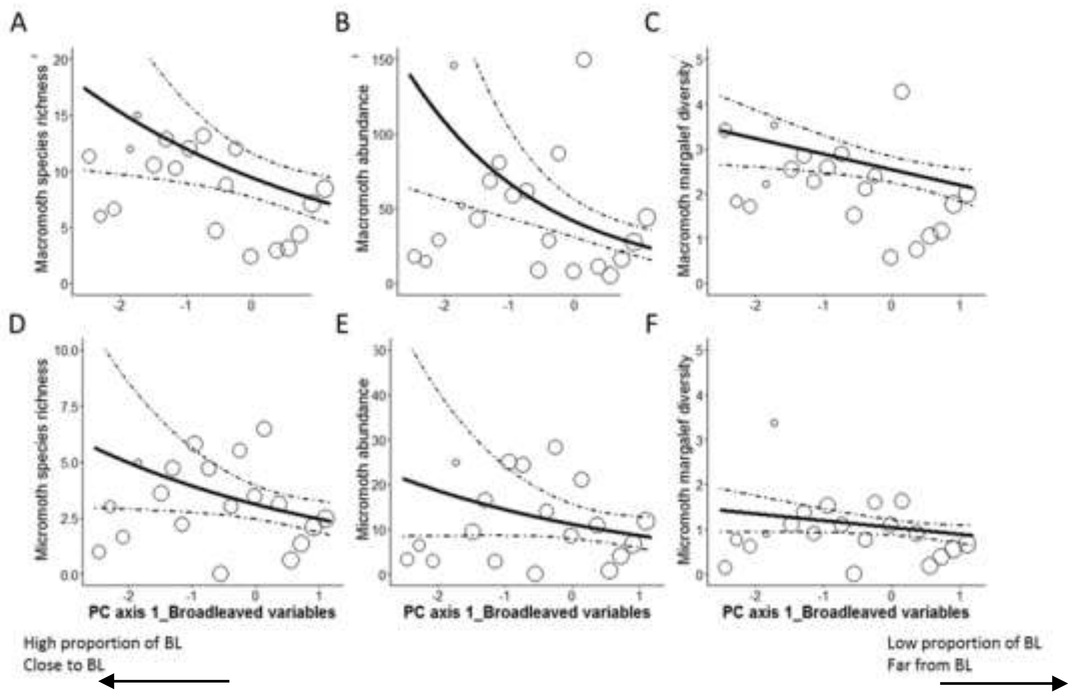

Fig. 4. Impacts of the amount and proximity of broadleaf woodland (BL) on (A – C) Macro moth species richness, abundance and diversity, and (D – F) Micro moth species richness, abundance and diversity per stand. Different scales are used for abundance and richness due to higher richness and abundance in macro moths compared to micro moths. Original data on richness, abundance and diversity are superimposed as grey circles with diameter proportional to the number of stands where mean values occurred. Dashed lines represent 95% confidence intervals around the predictions.

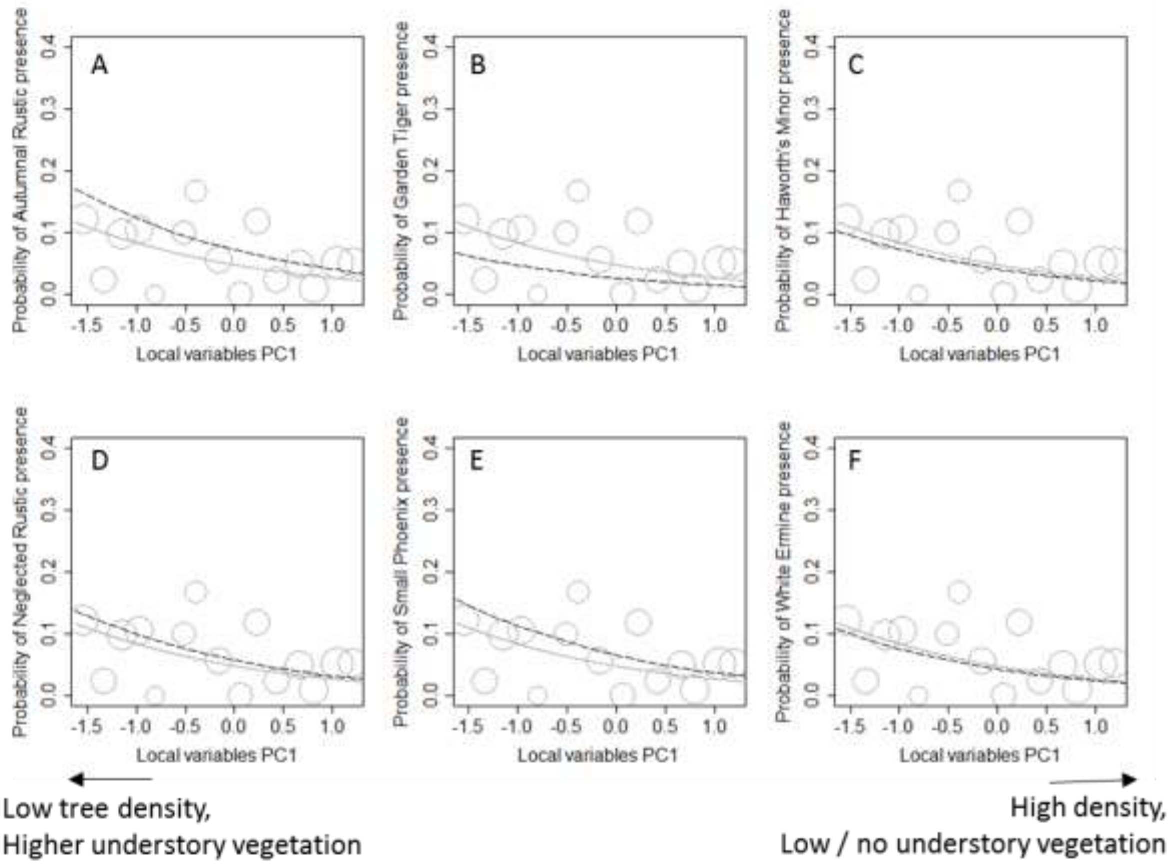

Figure 5. Probability of recording priority BAP species by Local PC1 scores (associated with a shift from stands with low canopy cover and taller vegetation height to stands with high canopy cover and low vegetation height). Sites low on PC1 are predominantly clear fell and young, moving to thinned and mature aged stands loading high on PC1. Dotted lines are species specific, whilst the grey line shows the trend across all six BAP species. Original data on richness, abundance and diversity are superimposed as grey circles with diameter proportional to the number of stands where mean values occurred.

Both micro and macro moth abundance responded strongly to Felling PC1; macro moth abundance decreased from 68.0 (40.0 – 114.0) moths in sites far from felling or with a low proportion of felling in the surrounding landscape to 10.0 (5. 4 – 18.5) in sites closer to felling or with more felling in the surrounding landscape, and micro moth abundance decreased from 25.0 (14.0 – 42.0) individuals to

2.5 (1.3 – 4.9) individuals (Figure 3 B, E). There was little response of diversity of either group to the proximity or prevalence of felling (Figure 3 C, F).

3.3 Effects of the presence of broadleaf tree cover on moth communities

In general, the proximity and amount of broadleaf tree cover within 4km of sampling sites appeared to be positively associated with species richness, abundance and diversity for both macro and micro moths, although the effect was smaller than the impact of felling (Table 1, Figure 4). The effect is clearest for species richness, with richness of macro moths in stands nearest to the largest patches of broadleaf tree cover double that of stands furthest from smaller patches of broadleaf, increasing from 7.0 (5.0 – 9.0) species to 15.0 (8.0 – 29.0) species per stand (Figure 4 A). Similarly, micro moth richness increased from 2.5 (1.3 – 3.1) species in stands far from broadleaf tree cover and with a low proportion of broadleaf in the surrounding area to 5.0 (3.0 – 10.0) species richness in stands closest to broadleaf patches or with a high proportion of broadleaf tree cover in the surrounding landscape (Figure D). Whilst the influence of broadleaf woodland on abundance of both groups is similar, the relationship appears to be weaker than for species richness (Figures 4B, E), and for macro moths appears to be driven by high abundance at one site (Figure 4 B). Neither local variables, felling nor broadleaf characteristics had any correlation with macro or micro moth dominance.

3.4 Influence of local characteristics, felling and broadleaved woodland on BAP priority species

The likelihood of catching a BAP species increased further from felled areas, and as the size of the nearest broadleaf patch increased, with all six species having very similar response to both variables. However, the correlation of Local PC1 with micro moth presence differed between the BAP priority species. *Eugnorisma glareosa* (Autumnal Rustic; Figure 5 A) and *Ecliptopera silaceata* (Small Phoenix; Figure 5 E) responded relatively strongly to Local PC1, and were more likely to be recorded in open stands with taller vegetation whereas there was relatively little change in the probability of capture for *Arctia caja* (Garden Tiger; Figure 5 B).

4. Discussion:

Here, we demonstrate that plantations can support large communities of moths, including several BAP priority species. Lepidoptera are one of the most abundant and diverse insect orders, but are currently undergoing widespread declines across Europe (Fox et al. 2013). Loss of habitat and changes to silvicultural practices in native woodlands have been cited as drivers of these losses, but to date the value of coniferous plantations for moths has been ignored due to their perception as being a poor habitat for biodiversity.

Moth abundance was dominated by generalist species which are preferentially found in heath or bog habitats, or by a small number of conifer specialist species. Macro moth abundance was highest in relatively low density stands with vegetation cover, which are more likely to support appropriate larval host plants, compared to dense stands with predominantly bare or moss as dominant ground cover. In addition, sites loading low on Local PC1 were often recently felled and young stands with large amounts of dead wood remaining which would benefit saproxylic species (Thorn et al. 2015). However, we saw no effect of stand characteristics on species richness or diversity in macro moths or for any micro moth response metric, possibly because we captured a high proportion of generalist moth species which have less strict habitat associations. We have no data on the species composition of moth communities prior to afforestation but it is likely to include species which specialise on low nitrogen, open habitats.

Felling was strongly and negatively correlated with both macro- and micro moth species richness, abundance and diversity. Macro and micro moth species richness was three times higher in sites furthest from felling, and with fewer felled patches in the immediate landscape, whereas abundance for macro and micro moths was between 7 and 10 times higher in sites further from felling and with less felling in the surrounding landscape. This reflects patterns reported from managed native broadleaf forests in Indiana, which found that clear felling significantly reduced moth species richness compared to either no management or selective felling (Summerville & Crist

2002). Clear felling causes substantial changes in the floristic composition of the forest habitat and through substantial changes in microclimate, to herbaceous ground cover and host plant availability (Summerville 2011). Summerville (2011, 2013) similarly found that species richness of moths was 40% lower after timber removal, with the impacts of felling persisting up to 200m from the cleared site itself.

The nature of the landscape matrix stands are embedded in may impede or facilitate dispersal between habitat types (Tscharntke et al. 2012); if there is too much felling in the surrounding landscape it may impede moth movements. Felled stands themselves may still be attractive to particular moth species due to intermediate levels of disturbance allowing pioneer and to some degree specialist species to coexist (Hamer et al. 2003). Indeed, in simplified landscapes, characterised by high disturbance, dynamics in habitat patches are likely to be determined by the availability of landscape wide remnant communities, particularly for species able to disperse over wide distances (Tscharntke et al. 2012).

Disturbed habitats are often characterised by a high abundance of a few generalist species, with the same subset of taxa dominating local stands and at the regional level. The majority of the moths we trapped were generalist species (Supplementary data 1), this may reflect the fact that moths using the plantations are those which can persist in a disturbed environment, as generalist species are more resilient to disturbance (Franzén & Johannesson 2007). For example, although 14% of all the moths we recorded are deciduous specialist feeders, the tree species they specialise on are often planted as deciduous tree cover in plantations (Tallamy & Shropshire 2009). It is not possible to tell from our study whether moth populations in plantations differ significantly from those in native broadleaf woodlands. However, due to the levels of disturbance caused by felling and the potential lack of host plants, as well as the predominance of generalist species we found in our plantation sites, we would expect plantation woodlands to support a less diverse moth population than broadleaf woodlands do. Macro moth species richness in the plantations was similar to that found in broadleaf woodlands within an agricultural matrix, although abundance was lower, while micro moth richness was 25% higher than in agricultural woodlands (Fuentes-Montemayor et al. 2012). Micro moth richness was similar to that reported from urban woodlands, but macro moth richness was 40% higher in plantations (Lintott et al. 2014). It is surprising that similar or lower species richness and abundance was found in urban (Lintott et al. 2014) and agricultural woodlands (Fuentes-Montemayor et al. 2012). It would be interesting to determine whether this is due to geographical differences (sites surveyed by Fuentes-Montemayor et al. 2012, and Lintott et al. 2014 were in Scotland but further north than the majority of sites surveyed for this study) or whether woodlands surrounded by agricultural and urban land are similarly disturbed habitats due to a more hostile matrix (Tscharntke et al. 2012), although the drivers of disturbance may differ.

Continuous cover forestry, which involves the continuous and uninterrupted maintenance of forest cover and avoids clear felling (Pommerening & Murphy 2004), has been advocated as an alternative forest management system. The UK forest standard requires managers to identify areas “which can be managed under a continuous cover forestry system and build them into forest design” (Mason et al. 1999). Despite not being appropriate for widespread use in all plantation forests due to the potential risk of wind damage to stands, there is evidence to suggest that multi aged systems may be more resilient to impacts of wind (O’Hara & Ramage 2013) and the potential forest health and yield benefits are increasingly recognised, with over 10% of Forestry Commission woodlands now under continuous cover management (Macdonald et al. 2009; O’Hara & Ramage 2013). Switching to continuous cover forestry may benefit moth communities; in Indiana (USA) Summerville et al (2009) found that shelterwood harvesting (removal of 15% biomass and similar in concept to continuous cover forestry) did not reduce functional and compositional resilience of lepidopteran communities compared to group selection harvesting (80% of tree biomass removed) and clear felling which had a significant negative impact. Additionally, moth communities showed signs of recovery within three years compared to other studies showing impoverished moth communities up to 60 years after clear felling (Summerville 2013; Summerville et al. 2009)

We found that the amount and proximity of broadleaf tree cover positively influenced moth species richness, and to some extent abundance. Many native tree species such as *Betula*, *Quercus* and *Salix* have large numbers of moth species associated with them (Tallamy & Shropshire 2009) and are commonly planted in conifer plantations as broadleaf regeneration trees. Fuentes-Montemayor et al (2012) found that species richness was highest in woodland with no conifers, so increasing landscape heterogeneity by planting patches of broadleaf tree cover within the plantation landscape may be invaluable islands allowing moth species to persist within the plantation matrix despite felling disturbance.

We recorded 13 BAP priority species using plantation woodlands. BAP priority species are so designated due to their rapidly declining populations across the United Kingdom and the need for further scientific study in order to assess and understand their population declines (Conrad et al. 2006). Of these, seven were present in fewer than 10 sites and were removed from further modelling. Of the six remaining species, all are habitat generalists or conifer and moorland habitat specialists. These species responded to stand type characteristics (separated by local PC1) differently. The Autumnal Rustic (*Eugnorisma glareosa*) and the Small Phoenix (*Ecliptopera silaceata*) were most likely to be detected in open stands with low canopy cover and stand density; the Autumnal Rustic is a generalist species often associated with moorland habitats which constitute a large proportion of the surrounding landscape and the Small Phoenix is a conifer specialist, and therefore likely to thrive in conifer plantations. All BAP species were significantly less likely to be recorded in stands closer to felled areas regardless of the size of the felled area or the proportion of felling in the surrounding area which, considering the two species' preference for open stands is somewhat surprising. All BAP species also responded equally positively to the size of the nearest patch of broadleaf tree cover. Broadleaf patches within plantations are not part of active harvesting programs, and are maintained or increased to meet biodiversity and restructuring guidelines (Watts et al. 2008), so may provide a potential source from which moth species can disperse.

4. 1 Management recommendations:

Worldwide, forest managers increasingly recognise the importance of sustainable forest management to improve biodiversity, but exotic pine plantations have received relatively little attention for their potential contribution to moth communities above and beyond the impacts of pest moth species. However, we found similar or higher levels of abundance and diversity compared to fragmented urban and agricultural woodlands in nearby regions (Lintott et al. 2014; Fuentes-Montemayor et al. 2012), and more BAP priority species in conifer plantations than urban woodlands (Lintott et al. 2014). We found that moth richness, abundance and diversity were influenced by plantation management and consider that the following should be taken into account when considering how plantation management may affect moth communities:

1. **Switching to continuous cover forestry:**

Similar to other studies in native woodlands under felling pressure (Summerville 2014; Summerville 2011; Summerville & Crist 2002; Summerville 2013; Summerville et al. 2009), felling significantly affected moth populations in our study sites, reducing species richness and abundance. Since clear felling was the only timber extraction technique used at our sites we were not able to compare with other lower-intensity methods. Switching to continuous cover forestry where appropriate will benefit moth communities and in turn the small mammal, bird and bat species which rely on them as a prey source while not negatively impacting forest productivity (Macdonald et al. 2009).

2. **Maintaining broadleaf woodland:**

Moth abundance and richness was far higher close to broadleaf tree cover; continued replanting of broadleaf trees and reduced intensity of management where possible near broadleaf stands should benefit both micro and macro moth richness and abundance. Many moth species can only disperse over relatively short distances (Merckx et al. 2012), therefore increasing the amount and connectivity of broadleaf woodland may allow moth species to persist within and disperse throughout plantations. All BAP priority species responded strongly to the size of the nearest patch of broadleaf tree cover, so reducing forestry operations near large patches of broadleaf trees is likely to benefit moth communities in general and BAP species in particular.

3. **Monitoring BAP priority species in plantations:**

   Of all the BAP priority species, the Garden Tiger (*Arctia caja*) moth was of particular interest as it is a conspicuous species that has declined widely across the UK, possibly due to climatic changes such as warmer wetter winters (Conrad 2002). More northerly habitats may be essential for the persistence of this species, and low density plantation stands may be an important refuge for this species in the face of future climate change. In addition, the Autumnal Rustic (*Eugnorisma glareosa*) which was abundant in plantation sites, has undergone substantial declines throughout the UK, thought to be related to pesticide use. Plantation sites should be included in long term monitoring programs to understand further how BAP priority species are using plantation woodlands.

Moth populations in Sitka spruce plantations appear to be predominantly generalist species, which may imply a disturbed community (Summerville et al. 2009). However, the presence of some BAP species demonstrates the importance of surveying sites that may historically be perceived as poor for biodiversity. With sympathetic management, plantation forests may have a role to play in preserving and supporting moth populations, particularly as climate change may result in changing species distributions.

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References:

Boye, P. & Dietz, M., 2005. Development of good practice guidelines for woodland management for bats. *English Nature Research Reports*, 661, pp.1–89.

Brockerhoff, E.G. et al., 2008. Plantation forests and biodiversity: Oxymoron or opportunity? *Biodiversity and Conservation*, 17(5), pp.925–951.

Burnham, K. & Anderson, D., 2002. *Model selection and multimodel inference: a practical information theoretic approach*, Springer Verlag, New York.

Conrad, K.F. et al., 2006. Rapid declines of common, widespread British moths provide evidence of an insect biodiversity crisis. *Biological Conservation*, 132(3), pp.279–291.

Crawley, M., 2007. *The R Book*, West Sussex: John Wiley & Sons.

Dirzo, R. et al., 2014. Defaunation in the Anthropocene. *Science*, 345(6195), pp.401–406. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1251817%5Cnhttp://www.sciencemag.org/content/345/6195/401.short.

Fox, R. et al., 2010. Moths count : recording moths for conservation in the UK. *Journal of Insect Conservation*, 15, pp.55–68.

Fox, R., Parsons, M. & Chapman, J., 2013. The State of Britain's Larger Moths 2013. *Butterfly Conservation and Rothamsted Research, Wareham, Dorset, UK*, pp.1–32. Available at: http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:The+State+of+Britain+'+s+Larger+Moths+2013#1%5Cnhttp://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:The+State+of+Britain's+Larger+Moths+2013#0.

Franzén, M. & Johannesson, M., 2007. Predicting extinction risk of butterflies and moths (Macrolepidoptera) from distribution patterns and species characteristics. *Journal of Insect Conservation*, 11(4), pp.367–390.

Fuentes-Montemayor, E. et al., 2012. Factors influencing moth assemblages in woodland fragments on farmland: Implications for woodland management and creation schemes. *Biological Conservation*, 153, pp.265–275. Available at: http://dx.doi.org/10.1016/j.biocon.2012.04.019.

Goulson, D., Lye, G.C. & Darvill, B., 2008. Decline and conservation of bumble bees. *Annual review of entomology*, 53, pp.191–208. Available at: http://www.ncbi.nlm.nih.gov/pubmed/17803456.

Hamer, K.C. et al., 2003. Ecology of butterflies in natural and selectively logged forests of northern Borneo: The importance of habitat heterogeneity. *Journal of Applied Ecology*, 40(1), pp.150–162.

Humphrey, J.W., Ferris, R. & Quine, C., 2003. Biodiversity in Britain's planted forests: Results from the Forestry Commission's Biodiversity Assessment Project. *Forestry Commission, Edinburgh*, pp.51–62.

Jackson, D.A., 1993. Stopping Rules in Principal Components Analysis: A Comparison of Heuristical and Statistical Approaches. *Ecology*, 74(8), pp.2204–2214.

Lacki, M.J. et al., 2007. Prey consumption of insectivorous bats in coniferous forests of north-central Idaho. *Northwest Science*, 81(3), pp.199–205.

Lintott, P.R. et al., 2015. Differential responses to woodland character and landscape context by cryptic bats in urban environments. *Plos One*, 10(5), p.e0126850. Available at: http://dx.plos.org/10.1371/journal.pone.0126850.

Lintott, P.R. et al., 2014. Moth species richness, abundance and diversity in fragmented urban woodlands: implications for conservation and management strategies. *Biodiversity and Conservation*, pp.2875–2901.

Macdonald, E., Gardiner, B. & Mason, W., 2009. The effects of transformation of even-aged stands to continuous cover forestry on conifer log quality and wood properties in the UK. *Forestry*, 83(1), pp.1–16. Available at: http://forestry.oxfordjournals.org/cgi/content/long/83/1/1.

Magurran, A.E., 1988. *Ecological diversity and its measurement.*, Princeton: Princeton University Press.

Mason, W., Kerr, G. & Simpson, J., 1999. *What is continuous cover forestry?*, Information Note.

Mattila, N. et al., 2006. Ecological determinants of distribution decline and risk of extinction in moths. *Conservation Biology*, 20(4), pp.1161–1168.

McFadden, D., 1974. Conditional logit analysis of qualitative choice behaviour. In P. Zarembka, ed. *Frontiers in Econometrics*. New York: Academic Press, pp. 105–142.

Merckx, T. et al., 2012. Conserving threatened Lepidoptera: Towards an effective woodland management policy in landscapes under intense human land-use. *Biological Conservation*, 149(1), pp.32–39. Available at: http://dx.doi.org/10.1016/j.biocon.2012.02.005.

Nieminen, M., Rita, H. & Uuvana, P., 1999. Body size and migration in moths. *Ecography*, 22(6), pp.697–707.

O'Hara, K.L. & Ramage, B.S., 2013. Silviculture in an uncertain world: Utilizing multi-aged management systems to integrate disturbance. *Forestry*, 86(4), pp.401–410.

Ober, H.K. & Hayes, J.P., 2010. Determinants of nocturnal Lepidopteran diversity and community structure in a conifer-dominated forest. *Biodiversity and Conservation*, 19(3), pp.761–774.

Ohsawa, M., 2005. Species richness and composition of Curculionidae (Coleoptera) in a conifer plantation, secondary forest, and old-growth forest in the central mountainous region of Japan. *Ecological Research*, 20(6), pp.632–645.

Oxbrough, A. et al., 2010. Ground-dwelling invertebrates in reforested conifer plantations. *Forest Ecology and Management*, 259(10), pp.2111–2121.

Pommerening, A. & Murphy, S.T., 2004. A review of the history , definitions and methods of continuous cover forestry with special attention to afforestation and restocking. , 77(1).

Soille, P. & Vogt, P., 2009. Morphological segmentation of binary patterns. *Pattern Recognition Letters*, 30, pp.456–459.

Summerville, K.S., 2014. Do seasonal temperatures, species traits and nearby timber harvest predict variation in moth species richness and abundance in unlogged deciduous forests? *Agricultural and Forest Entomology*, 16(1), pp.80–86.

Summerville, K.S., 2013. Forest lepidopteran communities are more resilient to shelterwood harvests compared to more intensive logging regimes. *Ecological Applications*, 23(5), pp.1101–1112.

Summerville, K.S., 2011. Managing the forest for more than the trees: Effects of experimental timber harvest on forest Lepidoptera. *Ecological Applications*, 21(3), pp.806–816.

Summerville, K.S., Courard-Hauri, D. & Dupont, M.M., 2009. The legacy of timber harvest: Do patterns of species dominance suggest recovery of lepidopteran communities in managed hardwood stands? *Forest Ecology and Management*, 259(1), pp.8–13.

Summerville, K.S. & Crist, T.O., 2002. Effects of timber harvest on forest Lepidoptera: community, guild, and species responses. *Ecological Applications*, 12(3), pp.820–835.

Tallamy, D.W. & Shropshire, K.J., 2009. Ranking lepidopteran use of native versus introduced plants. *Conservation Biology*, 23(4), pp.941–947.

Thomas, J.A., 2004. Comparative Losses of British Butterflies , Birds , and Plants and. , 1879(2004).

Thorn, S. et al., 2015. Forest Ecology and Management Guild-specific responses of forest Lepidoptera highlight conservation-oriented forest management – Implications from conifer-dominated forests. *Forest Ecology and Management*, 337, pp.41–47. Available at: http://dx.doi.org/10.1016/j.foreco.2014.10.031.

Truxa, C. & Fiedler, K., 2012. Attraction to light-from how far do moths (Lepidoptera) return to weak artificial sources of light? *European Journal of Entomology*, 109(1), pp.77–84.

Tscharntke, T. et al., 2012. Landscape moderation of biodiversity patterns and processes - eight hypotheses. *Biological Reviews*, 87(3), pp.661–685.

Warren, M.S. et al., 2001. Rapid responses of British butterflies to opposing forces of climate and habitat change. , pp.65–70.

Warren, M.S. & Bourn, N. a D., 2011. Ten challenges for 2010 and beyond to conserve Lepidoptera in Europe. *Journal of Insect Conservation*, 15(1), pp.321–326.

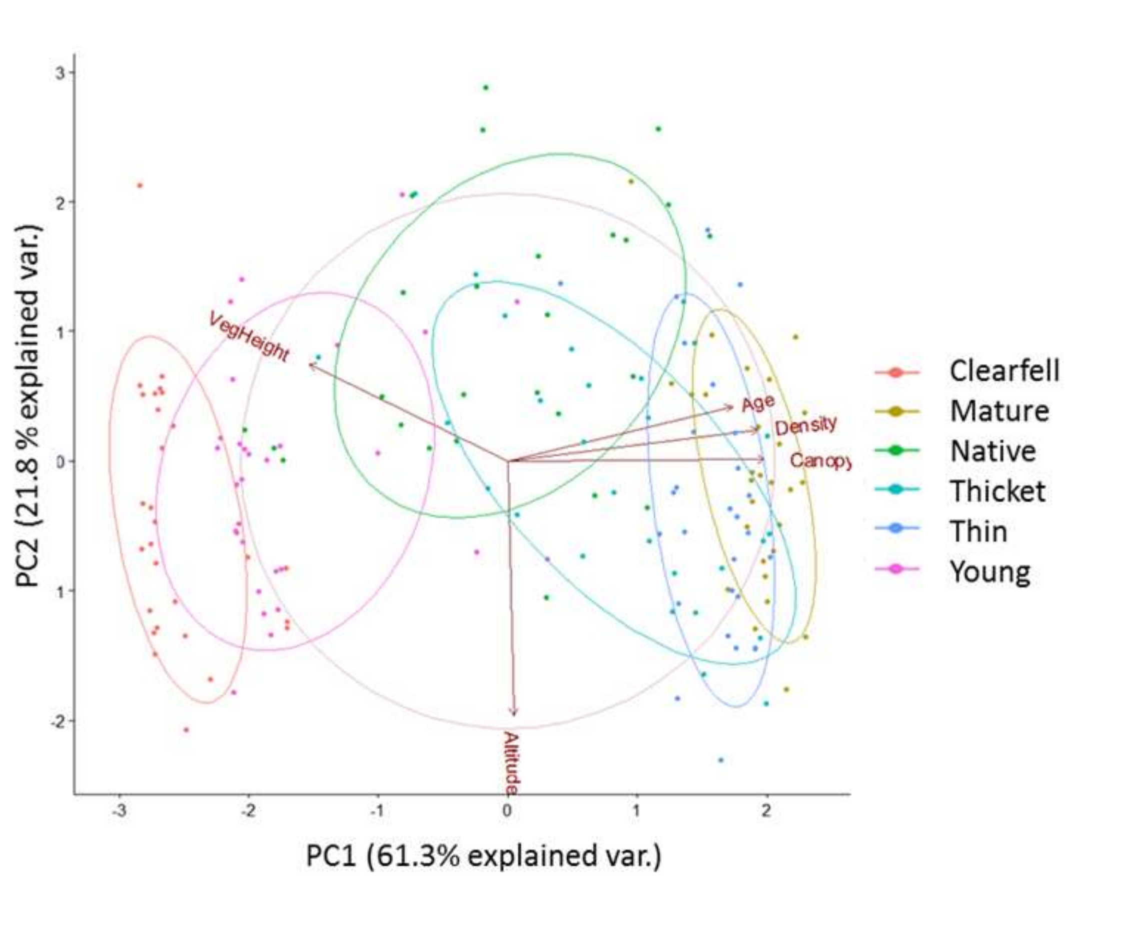
Watts, K. et al., 2008. Conserving forest biodiversity: recent approaches in UK forest planning and management. *Patterns and Processes in Forest Landscapes*, pp.373–398.

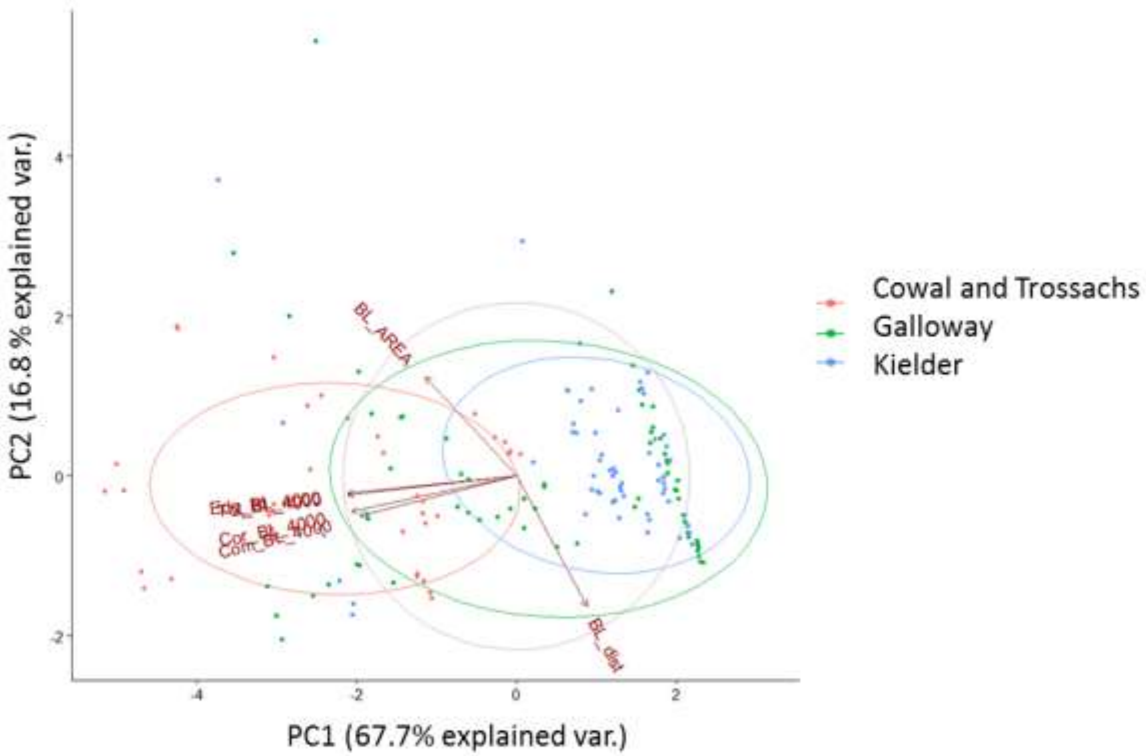
Supplementary data 1:

Table 1: Variables included in Principle Components Analysis.

PC axis	Measure	Unit	Minimum	Maximum	Median	Description
Local PC1	Altitude	m	83.8	466	230.7	Height above sea level
Local PC1	Density	trees per ha	0	3000	600	Number of trees per hectare
Local PC1	Veg height	mm	0	1744.1	156.6	Height of vegetation measured at 10 points across plot
Local PC1	Canopy cover	%	0	1	0.67	Total canopy cover as a percent
Local PC1	Stand Age	years	0	133	14	Stand age calculated from year of planting
Broadleaf PC1	BL_distance	m	0	3934	682	Distance in metres to nearest patch of mature broadleaf
Broadleaf PC1	BL_area	m2	0.1	163.2	1.3	Size of nearest mature broadleaf patch
Broadleaf PC1	Tot_BL_4000	%	0	11.3	0.8	Total broadleaf cover as a % of a 4km2 circle
Broadleaf PC1	Edge_BL_4000	%	0	2.9	0.2	Edge broadleaf cover as % of a 4km2 circle
Broadleaf PC1	Core_BL_4000	%	0	4.9	0.05	Core broadleaf (at least 10m from an edge) as a % of a 4km2 circle
Broadleaf PC1	Com_BL_4000	%	0	2.1	0.3	Total area / Edge area - complexity of cover within the landscape
Felled PC1	FE_distance	m	0	2670	527	Distance in metres to nearest felled stand
Felled PC1	FE_area	m2	0.04	92	13.9	Size of nearest felled stand
Felled PC1	Tot_FE_4000	%	0	35	5.1	Felled cover as a % of a 4km2 circle
Felled PC1	Edge_FE_4000	%	0	8	1.9	Edge felled cover as % of a 4km2 circle
Felled PC1	Core_FE_4000	%	0	26.5	2.4	Core felled (at least 10m from an edge) as a % of a 4km2 circle
Felled PC1	Com_FE_4000	%	0.8	2.1	1.5	Total area / Edge area - complexity of cover within the landscape

Supplementary data 2. Output from principle components analysis:




PC2 (16.8 % explained var.)
PC1 (67.7% explained var.)
BL_AREA
BL_dist
Cowal and Trossachs
Galloway
Kielder

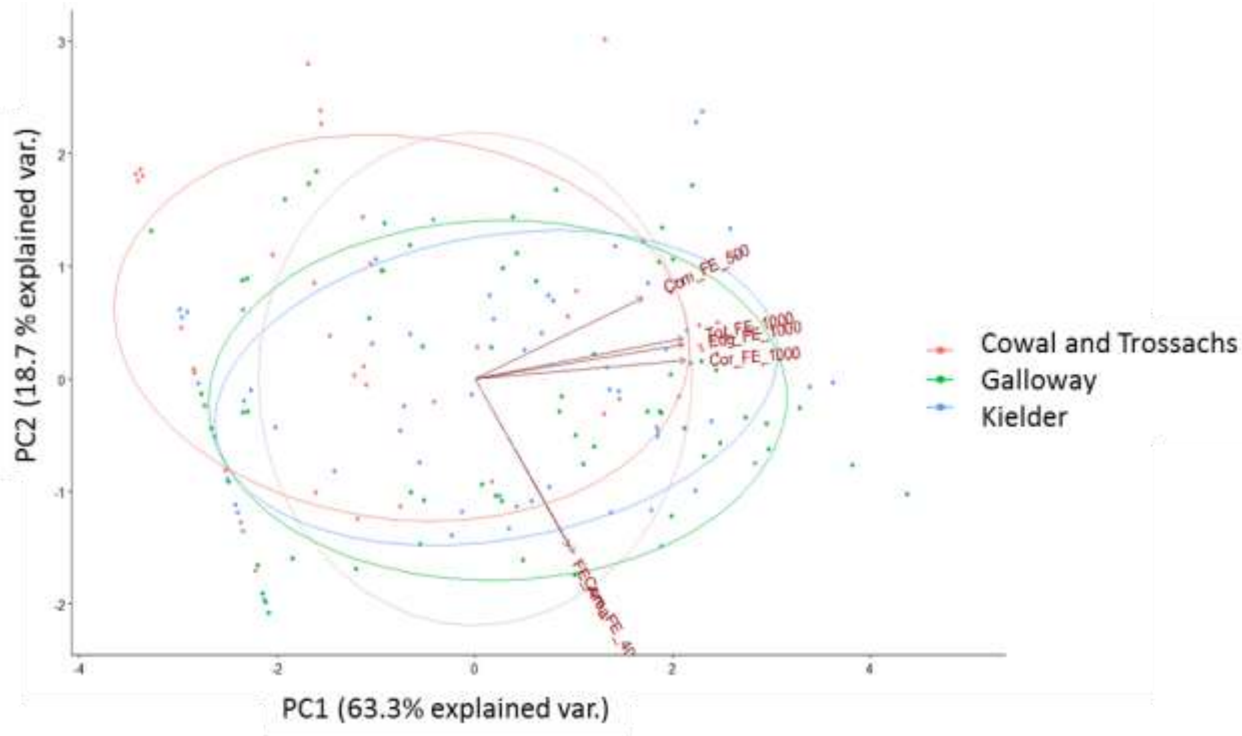


Figure 1. Principle components loadings for A) Local variables, B) Broadleaf variables and C) Felling variables. (see Supplementary data 1 for a description of the variables included in each PCA). Sites are coloured by stand type (Local PC) and by plantation (Broadleaf PC and Felling PC). Coloured ellipses delineate sites within each plantation that are similar to each other based on a normal probability distribution of 0.68. Dark red ellipsoid encompasses sites across all three plantations within a normal probability distribution of 0.68. Arrows indicate direction and magnitude of relationship, variables that are close together or directly opposite are highly correlated.

Supplementary data 3: Full list of moth species recorded as part of study

Table 1: Macro moth species:

Common name (Family)	Latin Name	Abundance per trap (± SE)	Habitat preference
Antler Moth (Noctuidae)	*Cerapteryx graminis*	0.24 ± 0.08	Grassland
Autumnal Rustic (Noctuidae)[a]	*Eugnorisma glareosa*	0.47 ± 0.17	Generalist
Barred Chestnut (Noctuidae)	*Diarsia dahlii*	0.01 ± 0.01	Deciduous
Barred Red (Geometridae)	*Hylaea fasciaria*	1.21 ± 0.33	Conifer
Barred Straw (Geometridae)	*Gandaritis pyraliata*	0.34 ± 0.13	Generalist
Barred Umber (Geometridae)	*Plagodis pulveraria*	0.01 ± 0.01	Deciduous
Beautiful Carpet Moth (Geometridae)	*Mesoleuca albicillata*	0.04 ± 0.02	Deciduous
Beautiful Golden Y (Noctuidae)	*Autographa pulchrina*	0.39 ± 0.10	Generalist
Bordered Beauty (Geometridae)	*Epione repandaria*	0.01 ± 0.01	Deciduous
Bordered Gothic (Noctuidae)	*Sideridis reticulata*	0.01 ± 0.01	Open ground
Bordered Pug (Geometridae)	*Eupithecia succenturiata*	0.01 ± 0.01	Generalist
Bordered Sallow (Noctuidae)	*Pyrrhia umbra*	0.02 ± 0.01	Grassland
Bordered White (Noctuidae)	*Bupalus piniaria*	0.07 ± 0.03	Conifer
Bright Line Brown Eye (Geometridae)	*Lacanobia oleracea*	0.02 ± 0.01	Generalist
Brimstone Moth (Noctuidae)	*Opisthograptis luteolata*	0.04 ± 0.03	Generalist
Broom Moth (Crambidae)	*Ceramica pisi*	0.12 ± 0.05	Moorland
Brown Rustic (Arctiidae)	*Elophila nymphaeata*	0.17 ± 0.11	Deciduous
Buff Ermine (Erebidae)	*Spilosoma lutea*	0.08 ± 0.03	Generalist
Buff Footman (Notodontidae)	*Eilema depressa*	0.19 ± 0.13	Wood generalist
Buff Tip (Noctuidae)	*Phalera bucephala*	0.01 ± 0.01	Deciduous
Burnished Brass (Geometridae)	*Diachrysia chrysitis*	0.12 ± 0.04	Open ground
Chevron (Arctiidae)	*Eulithis testata*	0.01 ± 0.01	Open ground
Clouded Border (Noctuidae)	*Tyria jacobaeae*	0.11 ± 0.05	Deciduous
Clouded Bordered Brindle (Erebidae)	*Apamea crenata*	0.06 ± 0.03	Grassland
Clouded Buff (Geometridae)	*Diacrisia sannio*	0.02 ± 0.01	Moorland
Clouded Magpie (Geometridae)	*Abraxas sylvata*	0.04 ± 0.03	Grassland
Common Carpet (Erebidae)	*Epirrhoe alternata*	0.23 ± 0.06	Generalist
Common Footman (Drepanidae)	*Eilema lurideola*	0.01 ± 0.01	Generalist
Common Lute String (Geometridae)	*Ochropacha duplaris*	0.01 ± 0.01	Deciduous
Common Marbled Carpet (Noctuidae)	*Dysstroma truncata*	0.01 ± 0.01	Wood generalist
Common Rustic (Hepialidae)	*Mesapamea secalis*	0.11 ± 0.07	Generalist
Common Wainscot (Geometridae)	*Korscheltellus lupulina*	0.42 ± 0.22	Grassland
Common Wave (Geometridae)	*Cabera exanthemata*	0.8 ± 0.22	Deciduous
Coxcomb Prominent (Noctuidae)	*Cabera pusaria*	0.07 ± 0.03	Deciduous
Dark Arches (Geometridae)	*Apamea monoglypha*	0.27 ± 0.09	Generalist
Dark Brocade (Geometridae)[a]	*Xanthorhoe ferrugata*	0.17 ± 0.10	Generalist
Dark Marbled Carpet (Noctuidae)	*Dysstroma citrata*	0.54 ± 0.14	Generalist
Dark Tussock (Noctuidae)	*Abrostola triplasia*	0.01 ± 0.01	Open ground
Dotted Carpet (Noctuidae)	*Aporophyla lutulenta*	0.01 ± 0.01	Wood generalist

Dotted Clay (Noctuidae)	*Xestia baja*	0.22 ± 0.10	Generalist
Double Dart (Noctuidae)[a]	*Graphiphora augur*	0.02 ± 0.01	Wood generalist
Double Square Spot (Geometridae)	*Xestia triangulum*	0.27 ± 0.13	Deciduous
Double Striped Pug (Lasiocampidae)	*Gymnoscelis rufifasciata*	0.04 ± 0.02	Generalist
Drinker Moth (Noctuidae)	*Euthrix potatoria*	0.36 ± 0.09	Generalist
Dusky Brocade (Noctuidae)[a]	*Apamea remissa*	0.02 ± 0.02	Generalist
Dwarf Pug (Geometridae)	*Eupithecia tantillaria*	0.02 ± 0.01	Conifer
Ear Moth (Geometridae)[a]	*Amphipoea oculea*	0.08 ± 0.04	Generalist
Flame Carpet (Noctuidae)	*Selenia dentaria*	0.53 ± 0.14	Generalist
Flame Shoulder (Noctuidae)	*Ochropleura plecta*	0.58 ± 0.14	Generalist
Four Dotted Footman (Geometridae)	*Luperina testacea*	0.1 ± 0.05	Generalist
Foxglove Pug (Geometridae)	*Eupithecia pulchellata*	0.01 ± 0.01	Generalist
Frosted Orange (Noctuidae)	*Gortyna flavago*	0.01 ± 0.01	Generalist
Garden Carpet (Geometridae)	*Xanthorhoe fluctata*	0.13 ± 0.10	Generalist
Garden Tiger (Erebidae)[a]	*Arctia caja*	0.33 ± 0.12	Generalist
Gold Spangle (Noctuidae)	*Autographa bractea*	0.01 ± 0.01	Generalist
Gold Swift (Hepialidae)	*Phymatopus hecta*	0.02 ± 0.02	Generalist
Golden Rod Pug (Geometridae)	*Eupitheca virgaureata*	0.02 ± 0.01	Generalist
Golden Y (Noctuidae)	*Autographa jota*	0.09 ± 0.04	Generalist
Gothic (Noctuidae)	*Naenia typica*	0.01 ± 0.01	Deciduous
Green Arches (Noctuidae)	*Anaplectoides prasina*	0.01 ± 0.01	Generalist
Green Carpet (Geometridae)	*Colostygia pectinataria*	4.44 ± 0.86	Deciduous
Green Pug (Geometridae)	*Pasiphila rectangulata*	0.01 ± 0.01	Deciduous
Grey Arches (Noctuidae)	*Polia nebulosa*	0.01 ± 0.01	Generalist
Grey Dagger (Noctuidae)[a]	*Acronicta psi*	0.01 ± 0.01	Deciduous
Grey Mountain Carpet (Geometridae)[a]	*Entephria caesiata*	0.13 ± 0.05	Generalist
Grey Pine (Geometridae)	*Thera obeliscata*	0.03 ± 0.03	Moorland
Haworths Minor (Noctuidae)[a]	*Celaena haworthii*	0.18 ± 0.08	Conifer
Heath Rustic (Noctuidae)[a]	*Xestia agathina*	0.15 ± 0.13	Moorland
Ingrailed Clay (Noctuidae)	*Diarsia mendica*	2.36 ± 0.50	Open ground
July Highflyer (Geometridae)	*Hydriomena furcata*	1.54 ± 0.44	Generalist
Knotgrass (Noctuidae)[a]	*Acronicta rumicis*	0.03 ± 0.01	Wood generalist
Larch Pug (Geometridae)	*Eupithecia lariciata*	0.05 ± 0.03	Conifer
Large Emerald (Geometridae)	*Geometra papilionaria*	0.09 ± 0.04	Generalist
Large Yellow Underwing (Noctuidae)	*Noctua pronuba*	1.66 ± 1.01	Generalist
Latticed Heath (Geometridae)[a]	*Chiasmia clathrata*	0.01 ± 0.01	Generalist
Lempkes Gold Spot (Noctuidae)	*Plusia putnami*	0.14 ± 0.05	Generalist
Lesser Swallow Prominent (Notodontidae)	*Pheosia gnoma*	0.01 ± 0.01	Open ground
Lesser Yellow Underwing (Noctuidae)	*Noctua comes*	0.36 ± 0.15	Generalist
Light Emerald (Geometridae)	*Campaea margaritaria*	0.15 ± 0.07	Generalist
Map Winged Swift (Hepialidae)	*Korscheltellus fusconebulosa*	2.09 ± 0.39	Wood generalist
Marbled Minor (Noctuidae)	*Oligia strigilis*	0.11 ± 0.06	Generalist
Middle Barred Minor (Noctuidae)	*Oligia fasciuncula*	0.39 ± 0.14	Generalist
Mouse Moth (Noctuidae)[a]	*Amphipyra tragopoginis*	0.01 ± 0.01	Generalist

Muslin Footman (Arctiidae)	*Nudaria mundana*	0.09 ± 0.03	generalist
Narrow Winged Pug (Geometridae)	*Eupithecia nanata*	0.21 ± 0.09	Generalist
Neglected Rustic (Noctuidae)[a]	*Xestia castanea*	0.04 ± 0.02	Open ground
Northern Arches (Noctuidae)	*Apamea exulis*	0.91 ± 0.31	Open ground
Northern Spinach (Geometridae)	*Eulithis populata*	0.01 ± 0.01	Open ground
Pale Eggar (Lasiocampidae)[a]	*Trichiura crataegi*	0.02 ± 0.01	Generalist
Peach Blossom (Drepanidae)	*Thyatira batis*	0.04 ± 0.02	Generalist
Pebble Prominent (Notodontidae)	*Notodonta ziczac*	0.05 ± 0.02	Deciduous
Pine Carpet (Geometridae)	*Pennithera firmata*	0.01 ± 0.01	Deciduous
Pink Barred Sallow (Noctuidae)	*Xanthia togata*	0.04 ± 0.02	Conifer
Poplar Grey (Noctuidae)	*Subacronicta megacephala*	0.02 ± 0.01	Generalist
Poplar Hawk Moth (Sphingidae)	*Laothoe populi*	0.01 ± 0.01	Deciduous
Pretty Pinion (Geometridae)	*Perizoma blandiata*	0.29 ± 0.07	Generalist
Purple Bar (Geometridae)	*Cosmorhoe ocellata*	0.56 ± 0.15	Moorland
Purple Clay (Noctuidae)	*Diarsia brunnea*	0.09 ± 0.03	Open ground
Red Carpet (Geometridae)[a]	*Xanthorhoe decoloraria*	0.01 ± 0.01	Generalist
Red Twin Spot Carpet (Geometridae)	*Xanthorhoe spadicearia*	0.01 ± 0.01	Generalist
Riband Wave (Geometridae)	*Idaea aversata*	0.01 ± 0.01	Moorland
Rosy Minor (Noctuidae)	*Litoligia literosa*	0.01 ± 0.01	Generalist
Rustic (Noctuidae)[a]	*Hoplodrina blanda*	0.01 ± 0.01	Grassland
Sallow (Noctuidae)[a]	*Cirrhia icteritia*	0.01 ± 0.01	Generalist
Satyr Pug (Geometridae)	*Eupithecia satyrata*	0.01 ± 0.01	Moorland
Saxon (Noctuidae)	*Hyppa rectilinea*	0.28 ± 0.14	Generalist
Scalloped Hazel (Geometridae)	*Odontopera bidentata*	0.01 ± 0.01	Generalist
Scalloped Hooktip (Drepanidae)	*Falcaria lacertinaria*	0.04 ± 0.02	Wood generalist
Scalloped Oak (Geometridae)	*Crocallis elinguaria*	0.01 ± 0.01	Generalist
Scalloped Shell (Geometridae)	*Hydria undulata*	0.08 ± 0.04	Wood generalist
Scarce Silver Y (Noctuidae)	*Syngrapha interrogationis*	1.25 ± 0.43	Deciduous
Shoulder Striped Wainscot (Noctuidae)[a]	*Leucania comma*	0.01 ± 0.01	Moorland
Silver Ground Carpet (Geometridae)	*Xanthorhoe montanata*	0.01 ± 0.01	Generalist
Sixstriped Rustic (Noctuidae)	*Xestia sexstrigata*	0.17 ± 0.04	Generalist
Small Angleshades (Noctuidae)	*Euplexia lucipara*	0.01 ± 0.01	Generalist
Small Dotted Buff (Noctuidae)	*Photedes minima*	0.07 ± 0.03	Generalist
Small Fanfoot (Erebidae)	*Herminia grisealis*	0.09 ± 0.04	Generalist
Small Phoenix (Geometridae)[a]	*Ecliptopera silaceata*	0.07 ± 0.02	Deciduous
Small Rivulet (Geometridae)	*Perizoma alchemillata*	0.01 ± 0.01	Generalist
Small Square Spot (Noctuidae)[a]	*Diarsia rubi*	0.07 ± 0.04	Generalist
Small Wainscot (Noctuidae)	*Denticucullus pygmina*	0.21 ± 0.14	Generalist
Smokey Wainscot (Noctuidae)	*Mythimna impura*	0.01 ± 0.01	Generalist
Snout (Erebidae)	*Hypena proboscidalis*	0.08 ± 0.04	Generalist
Spruce Carpet (Geometridae)	*Thera britannica*	0.01 ± 0.01	Generalist
Square Spot Rustic (Noctuidae)	*Xestia xanthographa*	0.05 ± 0.03	Conifer
Square Spotted Clay (Noctuidae)	*Xestia stigmatica*	0.22 ± 0.10	Generalist
Straw Dot (Noctuidae)	*Rivula sericealis*	0.22 ± 0.11	Deciduous
Striped Twin Spot Carpet (Geometridae)	*Coenotephria salicata*	0.01 ± 0.01	Open ground

Swallow Prominent (Notodontidae)	*Pheosia tremula*	0.34 ± 0.10	Generalist
Tawny Barred Angle (Geometridae)	*Macaria liturata*	0.01 ± 0.01	Deciduous
The Clay (Noctuidae)	*Mythimna ferrago*	0.02 ± 0.02	Conifer
Treble Bar (Geometridae)	*Aplocera plagiata*	0.01 ± 0.01	Open ground
Triple Spotted Clay (Noctuidae)	*Xestia ditrapezium*	4.09 ± 0.82	Generalist
True Lovers Knot (Noctuidae)	*Lycophotia porphyrea*	0.07 ± 0.07	Deciduous
Twin Spot Carpet (Geometridae)	*Mesotype didymata*	0.01 ± 0.01	Moorland
Water Carpet (Geometridae)	*Lampropteryx suffumata*	0.04 ± 0.02	Open ground
Welsh Wave (Geometridae)	*Venusia cambrica*	0.05 ± 0.02	Generalist
White Ermine (Erebidae)[a]	*Spilosoma lubricipeda*	0.02 ± 0.01	Generalist
White Wave (Geometridae)	*Cabera pusaria*	4.07 ± 1.03	Generalist
Willow Beauty (Geometridae)	*Peribatodes rhomboidaria*	0.05 ± 0.03	Wood generalist
Wormwood Pug (Geometridae)	*Eupithecia absinthiata*	0.01 ± 0.01	Generalist

Table 2: List of micro moth species recorded:

Common name (Family)	Latin Name	Habitat preference	Abundance per trap (± SE)
Water Veneer (Crambidae)	*Acentria ephemerella*	Water	0.05 ± 0.03
Caledonian Button (Tortricidae)	*Acleris caledoniana*	Moorland	0.01 ± 0.01
Notched winged Tortricid (Tortricidae)	*Acleris emargana*	Deciduous	0.04 ± 0.02
Dark-triangle Buttion (Tortricidae)	*Acleris laterana*	Open ground	0.01 ± 0.01
Rhomboid Tortrix (Tortricidae)	*Acleris rhombana*	Generalist	0.01 ± 0.01
Thistle Conch (Tortricidae)	*Aethes cnicana*	Grassland	0.04 ± 0.03
Burdock Conch (Tortricidae)	*Aethes rubigana*	Open ground	0.01 ± 0.01
Hook-marked Straw Moth (Tortricidae)	*Agapeta hamana*	Open ground	0.01 ± 0.01
Hemlock Moth (Depressariidae)	*Agonopterix alstromeriana*	Open ground	0.01 ± 0.01
Angelica Flat-body (Depressariidae)	*Agonopterix angelicella*	Generalist	0.01 ± 0.01
Brindled Flat-body (Depressariidae)	*Agonopterix arenella*	Generalist	0.01 ± 0.01
Gorse Tip Moth (Depressariidae)	*Agonopterix nervosa*	Generalist	0.02 ± 0.01
Coastal Flat-body (Depressariidae)	*Agonopterix yeatiana*	Generalist	0.01 ± 0.01
Barred Grass Veneer (Crambidae)	*Agriphila inquinatella*	Grassland	0.02 ± 0.01
Pearl Veneer (Crambidae)	*Agriphila straminella*	Grassland	0.68 ± 0.18
Common Grass Veneer (Crambidae)	*Agriphila tristella*	Grassland	0.01 ± 0.01
Broken Barred Roller (Tortricidae)	*Ancylis unguicella*	Moorland	0.01 ± 0.01
Birch Marble (Tortricidae)	*Apotomis betuletana*	Deciduous	0.05 ± 0.03
Rush Marble (Tortricidae)	*Bactra lancealana*	Open ground	0.22 ± 0.07
(Blastobasidae)	*Blastobasis decolorella*	Wood generalist	0.01 ± 0.01
Dark Groundling (Gelechiidae)	*Bryotropha affinis*	Generalist	0.01 ± 0.01
(Gelechiidae)	*Bryotropha boreella*	Generalist	0.01 ± 0.01
Cinereous Groundling (Gelechiidae)	*Bryotropha terrella*	Grassland	0.01 ± 0.01
Pearl-band Grass Veneer (Crambidae)	*Catoptria margaritella*	Moorland	0.21 ± 0.10
Pearl Grass Veneer (Crambidae)	*Catoptria pinella*	Moorland	0.01 ± 0.01
Dark Strawberry Tortrix (Tortricidae)	*Celypha lacunana*	Generalist	1.23 ± 0.30
Garden Grass Veneer (Crambidae)	*Chrystoteuchia culmella*	Grassland	0.14 ± 0.06
Flax Tortrix (Tortricidae)	*Cnephasia asseclana*	Generalist	0.01 ± 0.01
Hedge Case-bearer (Coleophoridae)	*Coleophora striatipennella*	Deciduous	0.01 ± 0.01
Hook-streaked Grass Veneer (Crambidae)	*Crambus lathoniellus*	Grassland	0.01 ± 0.01
Grass Veneer (Crambidae)	*Crambus pascuella*	Grassland	0.31 ± 0.09
Grey Gorse Piercer (Tortricidae)	*Cydia ulicetana*	Open ground	0.23 ± 0.07
Northern Tubic (Oecophoridae)	*Denisia similella*	Deciduous	0.01 ± 0.01
Little Grey (Crambidae)	*Dipleurina lacustrata*	Deciduous	0.01 ± 0.01
(Crambidae)	*Donacaula micronellus*	Moorland	0.04 ± 0.02
Dotted Shade (Tortricidae)	*Eana osseana*	Open ground	0.01 ± 0.01
Brown China Mark (Crambidae)	*Elophila nymphaeata*	Water	0.01 ± 0.01
Knapweed Bell (Tortricidae)	*Epiblema cirsiana*	Deciduous	0.01 ± 0.01
Thistle Bell (Tortricidae)	*Epiblema scutulana*	Open ground	0.01 ± 0.01

Bramble Shoot Moth (Tortricidae)	*Epiblema uddmanniana*	Deciduous	0.01 ± 0.01
Square Barred Bell (Tortricidae)	*Epinotia fraternella*	Conifer	0.01 ± 0.01
Common Birch Bell (Tortricidae)	*Epinotia immundana*	Deciduous	0.16 ± 0.09
Grey Poplar Bell (Tortricidae)	*Epinotia nisella*	Deciduous	0.03 ± 0.03
Small Birch Bell (Tortricidae)	*Epinotia ramella*	Deciduous	0.01 ± 0.01
Variable Bell (Tortricidae)	*Epinotia solandriana*	Deciduous	0.01 ± 0.01
Common Spruce Bell (Tortricidae)	*Epinotia tedella*	Conifer	0.01 ± 0.01
White Blotch Bell (Tortricidae)	*Epinotia trigonella*	Wood generalist	0.11 ± 0.04
Bright Bell (Tortricidae)	*Eucosma hohenwartiana*	Grassland	0.02 ± 0.02
Two-coloured Bell (Tortricidae)	*Eucosma obumbratana*	Open ground	0.04 ± 0.02
Pied Grey (Crambidae)	*Eudonia delunella*	Deciduous	0.01 ± 0.01
Small Grey (Crambidae)	*Eudonia mercurella*	Deciduous	0.01 ± 0.01
Brassy Tortrix (Tortricidae)	*Eulia ministrana*	Deciduous	0.25 ± 0.16
Lilac Leafminer (Gracillariidae)	*Gracillaria syringella*	Generalist	0.01 ± 0.01
Small Fanfoot (Erebidae)	*Herminia grisealis*	Deciduous	0.02 ± 0.01
Marsh Oblique-barred (Erebidae)	*Hypenodes humidalis*	Open ground	0.99 ± 0.44
Red Piercer (Tortricidae)	*Lathronympha strigana*	Wood generalist	0.01 ± 0.01
Rust-blotch Cosmet (Momphidae)	*Mompha lacteella*	Grassland	0.01 ± 0.01
Little Cosmet (Momphidae)	*Mompha raschkiella*	Grassland	0.01 ± 0.01
Carrion Moth (tineidae)	*Monopis weaverella*	Generalist	0.01 ± 0.01
Heather Groundling (Gelechiidae)	*Neofaculta ericetella*	Moorland	0.01 ± 0.01
Beautiful China Mark (Crambidae)	*Nymphula stagnata*	Water	0.08 ± 0.03
Sorrel Bent-wing (Opostegidae)	*Opostega salaciella*	Grassland	0.01 ± 0.01
Woodland Marble (Tortricidae)	*Orthotaenia undulana*	Generalist	0.01 ± 0.01
Barred Fruit Tree Tortrix (Tortricidae)	*Pandemis cerasana*	Deciduous	0.04 ± 0.03
White-faced Tortrix (Tortricidae)	*Pandemis cinnamomeana*	Deciduous	0.01 ± 0.01
Dark Fruit Tree Tortrix (Tortricidae)	*Pandemis hepararia*	Deciduous	0.03 ± 0.01
Large Marble (Tortricidae)	*Phiaris schulziana*	Moorland	0.01 ± 0.01
Small Clouded Knot-horn (Pyralidae)	*Phycitodes saxicola*	Open ground	0.01 ± 0.01
Light Streak (Oecophoridae)	*Pleurota bicostella*	Moorland	0.01 ± 0.01
Diamondback Moth (Plutellidae)	*Plutella xylostella*	Generalist	0.01 ± 0.01
Ash Bud Moth (Praydicae)	*Prays fraxinella*	Deciduous	0.01 ± 0.01
White Plume Moth (Pterophoridae)	*Pterophorus pentadactyla*	Generalist	0.01 ± 0.01
Common Purple and Gold (Crambidae)	*Pyrausta purpuralis*	Grassland	0.01 ± 0.01
Holly Tortrix Moth (Tortricidae)	*Rhopobota naevana*	Generalist	0.04 ± 0.02
Pinion Streaked Snout (Hypenodinae)	*Schrankia costaestrigalis*	Open ground	0.57 ± 0.15
Common Grey (Crambidae)	*Scoparia ambigualis*	Deciduous	3.26 ± 0.62
Meadow Grey (Crambidae)	*Scoparia pyralella*	Open ground	0.05 ± 0.02
Brown Plume (Pterophoridae)	*Stenoptilia pterodactyla*	Generalist	0.01 ± 0.01
Fulvous Clothes Moth (tineidae)	*Tinea semifulvella*	Generalist	0.01 ± 0.01
Birds-nest Moth (tineidae)	*Tinea trinotella*	Generalist	0.01 ± 0.01
Pale Straw Pearl (Crambidae)	*Udea lutealis*	Generalist	0.17 ± 0.07
Olive Pearl (Crambidae)	*Udea olivalis*	Generalist	0.02 ± 0.01
Dusky Pearl (Crambidae)	*Udea prunalis*	Generalist	0.01 ± 0.01
Spindle Ermine (Yponomeutidae)	*Yponomeuta cagnagella*	Deciduous	0.01 ± 0.01
Bird-cherry Ermine (Yponomeutidae)	*Yponomeuta evonymella*	Deciduous	0.44 ± 0.43

White-shouldered Smudge (Ypsolophidae)	*Ypsolopha parenthesella*	Deciduous	0.01 ± 0.01
Larch Tortrix (Tortricidae)	*Zeiraphera griseana*	Conifer	0.15 ± 0.12
Spruce Bud Moth (Tortricidae)	*Zeiraphera ratzeburgiana*	Conifer	0.17 ± 0.06

Supplementary data 4:

Stand characteristics for each management stage and stand features associated with management.

*Diameter at Breast Height – estimate of tree maturity

Stand Age	Management Stage	Key stand features
40 – 60 years	Mature	Occasionally thinned, stand density between 500 and 2200 stems ha^{-1}, average stand density: 1267 stems ha^{-1}, canopy closure between 80 and 100%, average closure 99%
20 – 40 years	Thin	Trees more densely packed, losing midstem branches and some trees dying off (self thinned). Occasionally thinned through management. Stand density between 600 – 2800 stems ha^{-1}, average stand density: 1624 stems ha^{-1}. Canopy closure between 50 and 100%, average closure: 95%
10 – 20 years	Thicket	Very dense, retain midstem branches, no undergrowth. Stand density between 300 – 3000 stems ha^{-1}, average stand density: 1850 stems ha^{-1}. Canopy closure between 16 and 100%, average closure: 69%
5 – 10 years	Young	Small, nearly all trees < 7cm DBH*, no canopy closure, lots of vegetation and ground cover
Clearfell	Felled < 5 years ago	Lots of dead wood and brash, standing water and undergrowth
Native	Unmanaged	Broadleaf stand, planted as part of plantation restructuring