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Annex-1: Biodiversity

Beauchamp, K.[1], Alison, J.[2], Broome, A.[1], Burton, V.[1], Griffiths, R.[2], Keith, A.M.[2], Maskell, L.C.[2], Siriwardena, G.[3] & Smart, S.M.[2]

[1] Forest Research, [2] UK Centre for Ecology & Hydrology, [3] British Trust for Ornithology

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UKCEH contact details	Bronwen Williams UK Centre for Ecology & Hydrology (UKCEH) Environment Centre Wales, Deiniol Road, Bangor, Gwynedd, LL57 2UW 01248 374500 erammp@ceh.ac.uk
Corresponding author	Kate Beauchamp, Forest Research kate.beauchamp@forestresearch.gov.uk
Authors	Beauchamp, K.[1], Alison, J.[2], Broome, A.[1], Burton, V.[1], Griffiths, R.[2], Keith, A.M.[2], Maskell, L.[2], Siriwardena, G.[3] & Smart, S.[2] [1] Forest Research, [2] UKCEH, [3] BTO
Contributing authors & reviewers	Anthony Geddes[1], John Healey[2], Tom Jenkins[3], Keith Kirby[4], Jerry Langford et al. [5], Darren Moseley[3], David A. Robinson[6], Eleanor Warren-Thomas[6], Bridget Emmett[6] [1] Confor, [2] Bangor University, [3] Forest Research, [4] Oxford University, [5] Woodland Trust, [6] UKCEH
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Abbreviations Used in this Annex

AES	Agri-environmental schemes
ASNW	Ancient Semi Natural Woodland
AW	Ancient Woodland
BTO	British Trust for Ornithology
CCF	Continuous Cover Forestry
DEFRA	Department for Environment, Food and Rural Affairs
ERAMMP	Environment and Rural Affairs Monitoring and Modelling Programme
FMA	Forest Management Approach
GHG	Greenhouse Gas
GMEP	Glastir Monitoring and Evaluation Programme
OTU	Operational Taxonomic Unit
PAWS	Plantations on ancient woodland sites
PCR-DGGE	Polymerase Chain Reaction - Denaturing Gradient Gel Electrophoresis
PLFAs	Phospholipid-derived fatty acids
PoMS	UK Pollinator Monitoring Scheme
UKFS	UK Forestry Standard
UKCEH	UK Centre for Ecology & Hydrology
UKWAS	UK Woodland Assurance Standard
WIG	Woodland Improvement Grant

Abbreviations and some of the technical terms used in this report are expanded on in the programme glossaries: https://erammp.wales/en/glossary (English) and https://erammp.cymru/geirfa (Welsh)

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1. INTRODUCTION TO ANNEX-1

In this Annex we explore the benefits and disbenefits of woodland creation & management on forest biodiversity in Wales. This Annex first sets out the key factors influencing woodland biodiversity and further sections detailed evidence reviews for specific taxa.

2. LANDSCAPE ECOLOGY AND FOREST BIODIVERSITY

2.1 Biodiversity and Land Use Change

Biodiversity, also called biological diversity, is the variety of all life on earth. Forest biodiversity considers the richness and evenness of species located in and adjacent to forests and woodlands, and those which depend on these habitats for all or part of their lifecycle.

Forests harbour most of global terrestrial biodiversity (Aerts & Honnay 2011; Liang et al. 2016; Perera, Peterson, Pastur, & Iverson 2018), but human exploitation has led to substantial changes in forest extent, dynamics, structure, and species composition globally (Halme et al. 2013) and in a UK context (Quine et al. 2011). Restoring and managing biodiversity is a key policy aim (e.g. Welsh Government 2015, 2016, 2018).

When considering biodiversity and land-use change such as woodland creation, discussions are comparative between the current and potential future range of land-uses and management types and their biological and cultural value, as well as context in the wider landscape. Land-use change can also have impacts on biodiversity at a wider, landscape scale and therefore the proximity to other habitats also needs to be considered. The movement towards landscape-scale ecology and decision making is highly beneficial to biodiversity.

Woodland biodiversity research presents particular challenges, given the long timescales and often large spatial scales over which it takes place, as well as huge variation in study design (Spake & Doncaster 2017). There are many approaches to assessing habitat biodiversity including measures of genetic, species, and ecosystem-level variation (Bellamy et al. 2018). Comparing the biodiversity consequences of land-use change is more complex, as alternative approaches to assessing biodiversity may have been used for different habitats, at different scales. Assessments of biodiversity and decisions regarding land-use change may also be complicated by either scientific, cultural, or political weightings of the value of species and/or ecosystems.

Woodland creation can have both positive and negative impacts on specific and general aspects of biodiversity (Burton et al. 2018), depending on a number of variables, including current site condition, soil type, woodland size, type of woodland and tree species, type of management, adjacent land type and condition, including those down slope and downstream, and nearby water sources (Natural England 2009). The impacts vary at different stages of the forest rotation/regeneration cycle (e.g. establishment, mature, veteran) and within different taxa, including microbes, invertebrates, vascular plants, fungi, lichen, mosses, reptiles, amphibians, mammals, and birds, therefore there is often a lag between woodland creation and species colonisation, in particular woodland specialists; therefore there may be a site-specific reduction in biodiversity in the short term before benefits are seen. The scale of implementation also has an effect.

The impacts on biodiversity from bringing un- and under-managed woodland into active management are considered in Annex-2/ERAMMP Report-34: *Managing Undermanaged Woodland*. In this section we review key evidence on the benefits and disbenefits of woodland creation on biodiversity in Wales, and how disbenefits might be mitigated.

2.2 Forest Biodiversity Principles

This section outlines key factors which may affect woodland biodiversity, giving context for the more detailed evidence sections for specific taxa which follow.

Woodland Type

Woodland priority habitats in Wales include all types of native woodland, especially ancient woodland (AW) and ancient semi-natural woodland (ASNW), upland oakwood, upland ashwood, wet woodland, lowland mixed deciduous woodland and lowland beech woodland as well as wood pasture, parkland, and traditional orchards (Welsh Government 2011). A high proportion of priority species are associated with native woodland and restoring plantations on ancient woodland sites (PAWS) and expanding areas of ASNW have been identified as some of the best routes to increasing biodiversity in these habitats (Lindenmayer et al. 2008). Due consideration is needed however to consider the habitat lost due to woodland expansion.

Tree species differ in many respects including lifecycle and number of associated species (Mitchell et al. 2014; Mitchell et al. 2019). Higher diversity in tree and shrub species can support greater numbers of species and increase resilience to environmental stresses such as drought and pests (Bellamy et al. 2018). Environmental heterogeneity in general encourages biodiversity, with protection of older veteran trees within stands, and maintenance of deadwood and wetland habitats within woodlands positively influencing biodiversity (Bellamy et al. 2018). Fostering and maintaining diversity in new woodland species mix, structure, and stand age over time is expected to be beneficial to a range of taxa (Burton et al. 2018, Filyushkina et al. 2018).

Woodland Size & Shape

The size of an individual woodland has an influence on the biodiversity it contains and its potential resilience. There is a well-established species-area relationship (Connor and McCoy 1979). Larger woodlands incorporate greater environmental heterogeneity, provide more ecological niches and support larger populations. Small woodlands support woodland edge species but may not provide sufficient conditions for woodland interior specialists, due to light levels, humidity, and foraging area. This relationship is more important for smaller woodlands, with increases in area having a greater effect than increasing the area of larger woodlands. There is a logarithmic relationship which means size is more important for smaller woodlands than for large.

The definition of a ‘small’ woodland also depends on the focal species being considered. In particular, woodland birds differ in their area thresholds for breeding (Dolman et al. 2007). The shape of the woodland is also important, with longer thinner forests supporting lower levels of biodiversity than more compact or circular (Bellamy et al. 2018, Usher et al. 1992, Usher et al. 1993, Usher & Keiler 1998). This is dependent of on the relative abundance of edge specialist versus woodland interior specialist species in the regional species pool. See Section 7.2 in this Annex for a summary of empirical studies of this question for birds.

Quality

The quality of woodland habitat may be even more important than size for biodiversity. Improving existing woodland quality is recommended before other measures such as increasing size, number of woodland patches, stepping-stones, and corridors (Crick et al. 2020). This can be done by encouraging natural processes, creating more niches, varying physical structure, restoring missing biodiversity, maintaining rare species, and reducing edge effects by buffering sites.

Time Lags

Species respond to conservation action in different ways, and temporal lags in species response could mask the ability to observe progress towards conservation success (Watts et al. 2020). Generalist species, which have less specific habitat requirements, are expected to respond first. Specialist long-lived species respond slower to changes, and display a much longer time lag, perhaps continuing to show decline after woodland improvement or creation (Watts et al. 2020). However, over longer time periods, they are expected to stabilise and show positive response to interventions. To take account of this, a shift away from measuring ‘total species’ as a measure of success, towards checking for more detailed milestones (e.g. arrival of generalists, successful breeding of generalists, arrival of specialists, self-sustaining populations of specialists) is recommended (Watts et al. 2020).

Wider Landscape

Given the above factors, it is accepted that woodland creation will enhance some components of biodiversity, particularly through following ‘the right tree in the right place’ approach; however, this varies by taxa and by landscape context. Deciding where to locate new woodland will depend on several factors, not only where woodland may establish most successfully, but on the value of the underlying habitat (and its associated biodiversity) to be converted. There will need to be a consideration of trade-offs to facilitate ‘net biodiversity gain’. Quantitative evidence of trade-offs may not be readily available and requires long-term, landscape scale monitoring or experiments. There is a significant lack of long-term monitoring studies for woodland creation in the UK, and a shift in focus of biodiversity metrics may be more informative (Burton et al. 2018). Most evidence to date is based on plot-based samples for single taxa, defined as ‘alpha’ diversity. It has been suggested that measures of ‘beta’ (spatial) or ‘gamma’ (total) diversity may be better suited to assessing land use change at the landscape level (von Wehrden et al. 2014). Focusing more on ‘functional diversity’, or the role that species play in ecosystems (Aerts & Honnay 2011) could also be beneficial for supporting decisions between habitats and species.

Surrounding habitat type and quality

Land use types adjacent to woodland can have negative impacts or ‘edge effects’ on biodiversity, especially if they are intensive or urban. Environmental heterogeneity within and between woodlands, and in the surrounding landscape, promotes the development of distinct communities (Bellamy et al. 2018). As described in priority habitats below, introducing woodland can also have negative impacts on surrounding habitats, and ecological surveys and assessment are needed.

Connectivity

Habitat fragmentation has had a negative impact on ancient woodland condition and diversity. The benefits of forest habitat networks are well established (Watts et al*.* 2005). An ecological network can be understood as a number of core, well connecting, high quality areas of well-functioning ecosystems (Crick et al. 2020). In principle, a well-connected, heavily woodland landscape will support higher levels of woodland biodiversity as species populations are more able to successfully colonise woodland patches (Bellamy et al. 2018).

Woodland specialists are most likely to benefit where native woodland creation is adjacent to existing ancient woodland. Establishing new woodlands near to existing forests can allow species to migrate, colonise or recolonise, and offer refuge during disturbance, and provide a source of seed, and wildlife to colonise/recolonise. Using natural regeneration of trees and shrubs native to the site or planting local genetic material can conserve valuable genetic diversity.

Hedgerows and trees outside woodlands can also make an important contribution to woodland connectivity by providing corridors and stepping-stones between woodlands that would otherwise be isolated. Connectivity between woodlands can facilitate exchange of species and genes between populations, increasing species and genetic diversity and therefore resilience to pests and pathogens (Bellamy et al. 2018) . However, improving quality and size of existing habitats is expected to be of greater benefit than stepping-stones or corridors (Natural England 2020). Currently 78% of hedgerows are estimated to be poor condition (Natural Resources Wales 2016) and this will affect the ability of woodland species to utilise linear features such as these to move through landscapes.

Connectivity can also increase some risks, facilitating the successful establishment and spread of a pest, pathogen or invasive species (Bellamy et al. 2018; Condeso and Montemeyer 2007; Maguire et al. 2015). In particular, with regards to red squirrels, new planting should avoid connecting red squirrel habitat to areas where grey squirrels dominate. The advantages of enhancing woodland connectivity for woodland species should therefore be weighed against their possible disadvantages, especially where the potential negative impacts may occur.

Woodland Management type

As noted in consideration of woodland type, diversity in species and age structure and management which supports these approaches is expected to be beneficial. Exploration of the effect of different Forest Management Approaches (FMAs) on UK forests highlights that management intensity influences species richness and abundance, with species dependent on continuity of forest cover, deadwood and large trees negatively affected by more intensive management (intensive even-aged and short rotation forestry; Sing et al. 2017). Forest management that mimics natural disturbances (close-to-nature and combined objective forestry) delivers greater biodiversity benefits through diversifying species and age classes of even-aged stands (Sing et al. 2017). Continuous Cover Forestry (CCF), where suitable, reduces many of the negatives associated with clear fell management, although increased management frequency can also have negative impacts to recreation and wildlife.

Non-native plantation forests can also deliver some biodiversity benefits for woodland specialists, by buffering native forest remnants and enhancing landscape connectivity where native woodland is scarce (Brockerhoff, Jactel, Parrotta, Quine, & Sayer 2008; Humphrey et al. 2000; Sing et al. 2017). Species mixtures also enhance biodiversity in plantation forests (Sing et al. 2017) as will retention of other habitat features. Planting density is also a key factor. Timber plantations planted at high density rapidly shade out understory vegetation and are associated with lower biodiversity as they age.

Forest management/regeneration cycle

Components of biodiversity fluctuate depending on the stage in the management or regeneration cycle of the forest, whether at establishment, early, pole stage, mature or veteran. Ongoing management including site preparation, fencing, planting, establishment, thinning, road formation, felling, and extraction all have different effects.

2.3 Preventing Disbenefits

2.3.1 Priority habitats

The Environment (Wales) Act 2016 Section 7 includes a list of priority species and priority habitats for the purpose of maintaining and enhancing biodiversity in Wales (Welsh Government 2016). Priority habitats have the potential to provide the richest and most varied components of biological diversity (Forestry Commission 2017). A species may be valued more highly if it is a habitat specialist, rare or endemic to a certain region, in a declining habitat, at risk of extinction or vulnerable to change. Likewise, a habitat may be valued by its high diversity, scarcity, uniqueness, sensitivity or its ability to host valuable species. Many habitats that are important for biodiversity in the UK have experienced a reduction in size and experienced fragmentation and need require restoration and expansion (Watts et al. 2005).

Woodland creation may negatively affect the overall biodiversity in a landscape if planted in an unfavourable location or of an inappropriate type or species for the intended management objective of the habitat. There is protective legislature and statutory designations in place to avoid potential negative impacts to priority habitats from woodland creation in Wales.

Planting on or adjacent to these areas may be discouraged. Special considerations are also needed for Section 7 Priority Species, including: red squirrels, water voles, barbastelle bat, West European hedgehog, brown hare, otter, pine marten, harvest mouse, dormouse, polecat, Bechstein's bat, noctule common pipistrelle, soprano pipistrelle, brown long-eared bat, greater horseshoe bat, lesser horseshoe bat. Other valuable areas include arable field margins, geological features, historic features, riparian habitats, lakes, ponds and canals. There is often a desire to conserve iconic communities by attempting to strictly maintain their species assemblages into the future; but this needs to be carefully weighed up against the potential gains in biodiversity or resilience to current and future environmental problems from allowing diversification (Bellamy et al. 2018).

2.3.2 Sustainable forest management

Appropriate measures can mitigate disbenefits from poorly planned woodland creation. The UK Forestry Standard (UKFS) requires that new forests are not planted on soils with peat exceeding 50 cm in depth or on sites that would compromise the hydrology of adjacent bog or wetland habitats. This would risk both priority habitats, species and carbon emissions. Protection is also given to soils against erosion by constraints onto the time of year in which management can be carried out, and by requirements for brash mats. Appropriate forest machinery and equipment may be required to minimise damage.

Forest establishment and management must also be carried out in a way that minimises risk to surrounding areas. The UK Forestry Standard is the authoritative standard for sustainable forest management in the UK. Woodland creation can introduce a new seed source that could spread to adjacent land, with negative consequences for important areas of open ground such as heathland. Forests can also encourage predatory species into an area, for example bird species that require open habitat are susceptible to predation from species that use woodland as cover. (Amar et al. 2011, Douglas et al. 2014; see also 7.1). Planting adjacent to protected and vulnerable habitats is therefore restricted (Forestry Commission 2018).

New planting must follow UKFS Forest & Water guidelines to prevent water course acidification, and planting in riparian buffer zones must follow specified guidelines. Where woodland creation is granted permission, the seasonal timing of planting and subsequent management is constrained to reduce impacts on protected species such as ground nesting birds and to minimise soil erosion.

2.4 Socio-cultural factors

Discussions surrounding woodland creation and biodiversity can be emotive politically, scientifically and culturally. Any assessment of overall changes in biodiversity will inevitably involve a subjective choice between species assemblages. However, studies exploring the socio-cultural aspects of woodland creation in the UK are lacking, with this being a common knowledge gap in ecological restoration research globally (Burton et al. 2018). Forestry strategy needs to be part of wider debates around land use and conservation priorities. Legacy effects of previous damage to valuable habitats and subsequent negative views towards forestry need to be addressed through discussion of the benefits of appropriate woodland creation, landscape scale decision making, and the collection of further evidence. Trade-offs and synergies with other land uses are context specific, and therefore local assessments which involve deliberative participation of land managers are argued to be necessary to make discussions and decisions around socio-cultural effects of woodland creation clearer (Burton et al. 2018; Slee, Polson, & Kyle 2014).

As outlined above, current policy has a presumption against woodland expansion on any land designated as a priority habitat type. It may be that more sophisticated net benefit assessment could support decision making. Based on their quality, some areas of priority habitat may be of huge biodiversity importance and thus a high priority for protection. Lower quality areas may be providing very little benefit. To use public funds in the most efficient way, woodland creation may offer a greater benefit (in terms of biodiversity and other associated benefits covered in these evidence pack) in these low-quality areas compared to improving the quality of the existing habitat. Decisions such as these are sensitive and value-laden, and are discussed further in Annex-4/ERAMMP Report-36: *Climate Change Mitigation* and Annex-6/ERAMMP Report-38: *Economics and Natural Capital Accounting.*

LAND SHARING VS. LAND SPARING APPROACHES

Given the varied effects of forest management outlined above, evidence around land sharing and land sparing approaches may be required (e.g. Paul & Knoke 2015). In places where continuous-cover and rotational forestry are both practiced on a large scale, continuous-cover approaches show significantly lower tree growth, and biodiversity benefits may need to be demonstrably better to justify the loss of carbon capture and wood production (Bianchi, Huuskonen, Siipilehto, & Hynynen 2020) . Both management approaches deliver benefits, and ‘either-or’ discussions should be avoided.

The "land sharing" model underpinning the UK Forestry Strategy, UKWAS, and the Welsh Government’s Woodlands for Wales Strategy assumes that all production forests should compromise timber yield with a wide range of measures to conserve biodiversity and deliver a range of ecosystem services. While a minimum level of environmental standards should be maintained in all woodlands (as in all farmland), conservation of woodland biodiversity as a whole, at a national or landscape scale, may not necessarily be achieved by seeking to deliver this equally amongst all woodlands. Instead a better benefit to cost ratio (more biodiversity per pound) may be achieved, to an extent, by a spatially explicit "land sparing" approach. Spatial scale is key. But identifying sites of greatest biodiversity value (beyond existing designated sites) and focussing conservation measures there, while sustainably intensifying timber production on sites of lower biodiversity potential, may give the greatest net benefit of biodiversity and timber production at the landscape/ national scale. Consideration then also has to be given to other services woodlands can provide such as recreation.

2.5 Global change

An additional major challenge is decision-making set in the context of global change. The consequences of interacting changes in drivers such as land-use, habitat fragmentation, pollutant deposition and climate change alter species distribution in uncertain ways. For example, this may mean that land use change may create new conditions which favour existing native species even more than they do now, favour non-native species over native species, favour native species that are presently absent or rare in an area or, interestingly yet problematically, result in so-called noanalogue conditions. These are environmental configurations that may never have occurred before and where managed or natural outcomes for community assembly are even more uncertain (Williams & Jackson 2007). The practical consequence is that woodland assemblages of the future may be different from those of the past but the timescales of ecosystem adaptation are uncertain. Moreover, managing for continuity of ecosystem service supply may require trading off valuation of native biodiversity against novel assemblages that comprise non-native taxa whilst also expecting novel dynamics as they ecosystems equilibrate to new conditions.

2.6 Overview

Woodland creation can have both positive and negative impacts on specific and general aspects of biodiversity. Well considered and appropriate woodland creation can benefit forest species and minimise disbenefits to other habitats. There are a broad range of potential forest types, species, and management approaches, each with different ecological profiles. It is important to distinguish between them when considering woodland expansion.

Given the complexities involved, there is often a lack of or unbalanced scientific evidence. Burton et al*.* (2018) report that there are very few published studies monitoring the change in biodiversity during the process of woodland creation from different land-uses (although note evidence for birds in woods created on farmland, 3.6.1.2), and much of the available evidence relates to existing woodland, with a bias in research and sampling towards coniferous plantations. Addressing this imbalance with further research monitoring woodland expansion and its impacts (trade-offs and synergies with existing habitats) is required. However, decisions may need to be made using the best available knowledge.

The following sections provide detailed evidence reviews for specific taxa.

3. PLANT DIVERSITY AND WOODLAND EXPANSION

This section considers plant diversity as a consideration in options for restoration and creation of woodland in Wales. The focus is on vascular plants due to the lack of evidence for bryophytes and lichens despite their critical role for biodiversity in Welsh woodlands internationally. Expansion of, for example, bryophyte-rich humid Atlantic woods in Wales is desirable however there are evidence gaps as to how feasible this is although a low disturbance regime is known to be required for existing sites.

3.1 Shade and managed disturbance

Understorey vascular plant diversity declines within increasing shade. For example, in a study of 103 broadleaved woods across Britain, and how they had changed between 1971 and 2002, marked reductions in understorey species richness were strongly correlated with increasing shade indicators – increasing shade-tolerant tree regeneration, increasing mean basal area of trees and shrubs, reduction in open habitats, reduction in plot and site-level signs of recent management (the largest reduction in management signs was actually across the 20 sites recorded in Wales, Kirby et al. 2005 – Appendix 9). These changes were thought to have resulted from greatly reduced disturbance following a period of unusually acute and geographically widespread timber removal at the end of WWII (Kirby et al. 2005). The apparent reversibility of these changes in understorey species richness was observed in a subset of the English sites impacted by the October 1987 storm (Fig 7.1b in Kirby et al. 2005; Fig 4 in Smart et al. 2014). This was inferred from the fact that species richness actually increased across the storm-impacted sites in contrast to non-impacted sites that on average declined. Hence much evidence supports the fact that vascular plant species diversity is increased by gap creation and more incident light at ground level following natural disturbances. These conditions can be created by active management (Harmer et al. 2010; Kopecký et al. 2013).

Changes in shade and disturbance act as a non-random filter on understorey plants. Many woodland plants are not strict shade-specialists and so are more likely to decline under a closing woodland canopy. This includes a range of nectar plants and butterfly larval food plants (Baude et al. 2016; Smart et al. 2000). Consequently, these are also less likely to flower even if they persist (Sparks et al. 1996). Whilst Baude et al. (2016) showed that broadleaved woodlands were second only to calcareous grasslands in their potential for nectar provision across the British countryside, appropriate light levels are required to maximise flowering and this requires natural or managed gap creation.

A suite of specialist woodland plants are shade-loving and so favoured by low light, higher humidity and reduced competition with faster-growing, shade-intolerant species. Beneficiaries of increased shade include many fern and bryophyte species and also many, but by no means all, of those species that are considered indicators of ancient woodlands in Britain (Kimberley et al. 2013; Glaves et al. 2009). Shade-tolerant plant species that can persist under a tree or shrub canopy may also be buffered against the competitive effect of shade-intolerant, nitrogen-loving species (Smart et al. 2016). This presumably results from the effects of light limitation on faster growing species that are otherwise not nutrient limited and could therefore invest resources into rapid biomass accumulation. We would therefore expect the buffering potential of a woody canopy to be most important in woodlands with high levels of soil nutrient availability. This could arise from exposure to nutrient surpluses from surrounding farmland (Didham et al. 2015) or from land-use legacies in secondary woodland (Perring et al. 2018; Vellend et al. 2007). The implication is also that reinstated disturbance might result in dominance by nitrogen-loving plants of low conservation importance, which suppress the abundance of species associated with intermediate or low productivity sites, many of which are of higher conservation priority, at least until cycles of woodland removal were effective in reducing fertility. At present this seems a plausible but largely untested scenario (Smart et al. 2014). There is also evidence that a sudden and severe episode of canopy removal may also have a detrimental effect on shade-tolerant woodland specialists. Brown et al. (2015) studied the consequences of clearfelling of conifers versus more gradual removal in 39 Woodland Trust sites across Britain. They found that the more open the canopy following conifer removal the greater the loss of typical woodland plant species leading them to recommend a more gradual thinning approach as opposed to abrupt clearfelling.

3.2 The importance of non-intervention and dead wood

Lack of disturbance and increased shade have also been linked with more ‘natural’ non-intervention regimes (e.g. Hambler & Speight 1995), in which understorey vascular plant species richness declines but which results in conditions favourable to specialised shade-tolerant biota including plants, fungi and invertebrates associated with dead wood. Restoration of these species groups and managing for their dispersal and establishment in new woodlands has to contend with the small size of many woodlands, lack of source species populations and the legacy effect of previous management resulting in low amounts and size distributions of dead woody debris plus high residual soil fertility. For example, Kirby et al. (1998) showed that there were much lower deadwood volumes in British woodlands and those found in long-continuity North American forests. Condition data from the National Forest Inventory indicated that 80% of British woodlands were unfavourable for deadwood volume in 2010-15, while in Wales 45% of surveyed sites had no qualifying deadwood present (Ditchburn et al. 2020a,b).

Long-continuity oak and ash woods in Wales are especially important for their Atlantic bryophyte and lichen flora favoured by high year-round rainfall, humidity and restricted temperature range (Bosanquet & Dines 2012; Ratciffe 1968). While bryophytes have been classified into those indicating ancient woodland in other parts of Europe (e.g. Mölder et al. 2015) it is not clear if this could be achieved or is necessary to aid conservation and woodland management in Wales.

3.3 The importance of shade and disturbance for forest specialist plants including Ancient Woodland Indicators

Not all of the species thought to be most characteristic of long continuity and ancient woodlands are strictly shade-dependent. Many are associated with better lit gaps and rides (Kimberley et al. 2013; Hermy et al. 1999; Peterken & Game 1984; Brown et al. 2015). The distinction seems to be that most of the species thought of as associated with older woodlands are not capable of tolerating persistently high grazing pressure or high soil fertility. Hence woodlands provide refugia for many plant species that are now less frequent in the wider farmed countryside (Kimberley et al. 2013). These differences in ecological tolerance are, however, species-specific leading to a continuum in the preference of species for ancient woodlands rather than a convenient hard and fast separation of ancient from non-ancient woodland species (Verheyen & Hermy 2002; Webb & Goodenough 2018; Gibson 1988). In practice, it is also worth pointing out that definitions and criteria for selecting Ancient Woodland Indicators also vary greatly across Britain (Glaves et al. 2009) so that evidence for impacts of management and other factors may be partly a function of the list applied. Regionally specific lists for Wales can be found in Appendix 4 of Glaves et al. (2009) and Castle et al. (2008).

3.4 Woodland creation and assembly of plant communities

Assembly of ‘typical’ or ‘desirable’ woodland understoreys will often require overcoming poor inherent dispersal and long distances to source populations plus land-use legacy effects. The latter include soil seedbanks, *in situ* vegetation and high soil fertility, all strongly influenced by previous agriculture (Coote et al. 2012; Harmer et al. 2001). Kimberley et al. (2014) showed, for example, that, for British broadleaved woodlands, traits related to dispersal and persistence among woodland vascular plants were most strongly filtered by conditions within each woodland – soil pH, C:N ratio and shade – but were also significantly influenced by area and age of the woodland. Brunet (2007) used a chronosequence approach to study the factors affecting plant species accrual in oak and sycamore plantations in Sweden over 20 to 80 years. As found by Kimberley et al. (2014) spatial factors acted to non-randomly filter immigrants. Isolated plantations were more likely to be colonised by well-dispersed species (adhesive or wind-dispersed seeds) while only plantations contiguous with existing ancient woodland increased in plant species richness to the levels of the adjacent woodland. This took 70-80 years to occur. Jacquemyn et al. (2003) also showed that vascular plant species richness was significantly lower in recent woodlands greater than 100 m from long-continuity woodland compared to recent woods adjacent to long-continuity woodland. Humphrey et al. (2015) summarized relevant literature comprising 28 spatial or temporal studies of vascular plant diversity responses to abiotic, temporal and spatial factors. They showed that while patch characteristics, that is abiotic and biotic conditions within the woodland, were important in 88% of studies, an effect of surrounding habitat was important in 80% and isolation in 74%.

While woodland plant communities (understorey and canopy) appear able to assemble spontaneously in 40 years in some circumstances (e.g. Harmer et al. 2001), specialist, poorly dispersing and rare species are likely to remain absent for extremely long periods especially where legacy effects of disturbance and increased soil P and N levels persist (Dupouey et al. 2002; Strengbom et al. 2001; Naaf & Kolk 2015). The flora was found to be richest before canopy closure, with steady development of ground flora but substantial turnover (Harmer et al. 2001). Thus, in any particular place, expectations of the timescale for achieving a plant community target need to be managed and informed by an evaluation of local factors. Moreover, objectives for woodland creation need to consider the woodland understorey as well as the tree canopy.

Likely timescales for dispersal of typical or desirable woodland plants into newly created woodland vary with climatic region, the favourability of the intervening matrix separating source and recipient woodlands (Svenning & Skov 2002) as well as distance to source populations. In a study of 49 woodland plant species in a sample of southern Swedish woods, mean migration rates ranged from 0.00 to 1.00 m per year with a median rate of 0.3 m per year. Ant-dispersed species had lower migration rates suggesting that if intensification in the wider countryside has also reduced ant populations or results in land between woodlands that is hostile to the movement of ants then this will reduce dispersal of dependent plants (Brunet & Von Oheimb 1998). Long-distance dispersal and establishment events can also occur but are often rare.

3.5 Creation of new woodland and extending existing woodland

Kimberley et al. (2015) showed that plant species diversity in broadleaved woodlands across Britain, which had reduced in extent since at least 1899, showed a stronger correlation with their earlier size than present day size indicating an extinction debt.[1] They concluded that there may still be time to arrest this ‘relaxation’ loss. Kolk & Naff (2015) carried out a similar study in NE Germany reconstructing land-use history for the past 230 years and correlating changes with modern plant species richness. They concluded that the extinction debt had been paid off in 160 years with reductions occurring quicker where fragmentation of woodlands in the landscape was higher. Taken together the results suggest that newly planted woodland or increases in extent of existing woodland encouraged by natural expansion in the absence of grazing, should focus on existing long-continuity woodlands or with a larger persistent species pool of woodland plants but where these woodlands have seen reduction in historical extent. Harmer et al. (2001) suggested that colonisation and plant community assembly is also likely to be more rapid if existing linear features are included, for example hedges rich in remnant forest species.

The existence of an extinction debt indicates that further species loss will inevitably occur as species pool size equilibrates to the reduced area of the woodland relative

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to its former size. However, the importance of this threat will differ depending on the size of the woodland. Losing a hectare from an already small woodland is likely to have a much greater impact than a hectare of much larger woodland. However, this also depends on the beta diversity of the woodland. For example, a highly heterogeneous long-continuity woodland could harbour specialised diversity in many distinct places across the forest environment such that loss of a particular area results in loss of habitat not replicated elsewhere in the woodland. This again emphasises the importance of place-specific assessment. Logically, planting new woodland in close proximity to, or encouraging expansion of, the existing wood might be expected to buffer these predicted losses from extinction debt (e.g. Brunet 2007). However, for this to be a likely outcome, conditions in the buffering woodland ought to be similar. If soil conditions reflect agricultural legacy and are very different in pH, macronutrient levels and seedbank composition then these are likely to make establishment more difficult (Kimberley et al. 2014; Govaert et al. 2020).

The extent to which newly created woodlands can accrue expected species assemblages also varies. Coote et al. (2012) showed that conifer plantation was much less able to support plant communities typical of semi-natural oak and ash woods. Ash plantations were significantly better although still with fewer typical woodland species. This suggests a need for managing expectations and tailoring creation and restoration targets to the type of woodland planted and its proximity to existing long-continuity woodlands.

Since plant diversity reduces with shade it is not surprising that much evidence indicates higher species diversity of trees, shrubs and herbaceous plants in woodland gaps, along rides and woodland edges. The impact of better lit conditions around the edges of woodlands interacting with high adjacent land-use intensity can however lead to reduced abundance of typical forest species in favour of nitrophilous species (Chabrerie, et al. 2013).

3.6 Timescales of impact

There is conflicting evidence about the timescales for plant community assembly after woodland creation; vascular plants show different responses to bryophytes. Many impacts are long term and so can take longer than the duration of most studies making it difficult to get evidence. As mentioned in Section 3, the timescale for the establishment of species, including plant communities, varies depending upon soil nutrient levels, climatic region, disturbance, potential for species dispersal, dispersal mechanism, distance to existing woodland and connectivity of woody linear features such as hedgerows.

Plant community biodiversity and other benefits could begin to appear in years 0-5 after woodland establishment and continue to develop over many years as the trees mature (ER4-Keenleyside et al. 2019). Restoration of full canopy cover from grassland could take 20-30 years but then transition from a flora of light demanding species to shade tolerant up to 40 years (Harmer et al. 2001). Even then, although there may be woodland plant communities (understorey and canopy) specialist, poorly dispersing and rare species are likely to remain absent for longer especially where legacy effects of disturbance and increased soil nutrients persist (Dupouey et al. 2002; Strengbom et al. 2001; Naaf & Kolk 2015). Another study showed that it took 70-80 years where new planting was adjacent to existing ancient woodland for species richness and composition to be similar to the existing woodland (Kimberley et al. 2014).

In general, the evidence shows that woodland vascular plant diversity increases where woodlands are subject to favourable disturbance regimes. However, a meta-analysis of 120 comparisons of managed and unmanaged stands across European forests indicated that the reverse applied to bryophyte diversity (Paillet et al. 2009, see also Edwards 1986). This difference is likely to be highly pertinent for Welsh Atlantic bryophyte assemblages in woodlands (Bosanquet & Dines 2012). It seems a reasonable supposition that accommodating both non-intervention areas where large woody debris is allowed to accumulate and areas where gap-dynamics are reinstated will require larger woods with a richer and more responsive existing biota associated with older woodlands or in closer proximity to ancient woodland. At present such differing objectives are accommodated among the existing archipelago of Welsh woodland sites.

3.7 Summary for plant biodiversity

In summary, much evidence about how to maximise vascular plant diversity in woodlands is consistent with Lawton’s (2010) call for habitat conservation to achieve bigger, better, more and joined up. However, the evidence also points to the importance of not translating these generalisations of the importance of extent, condition and connectivity into a series of one-size-fits-all interventions. Matching interventions to place-specific opportunities and constraints seems vital, especially if there is a shift in emphasis toward payment for environmental outcomes (e.g. Sidemo-Holm, et al. 2018). This is because in many situations land-use legacy, small woodland size, long-distances to source populations, poor dispersal and unfavourable management within and outside existing woodlands pose management challenges whose severity should help manage expectations for the timescales and resources required to achieve woodland restoration and creation objectives.

4. POLLINATORS

In Wales, pollinators provide an important function for wild plants and an ecosystem service in increasing production of some crop species. They also provide cultural and recreational benefits. Creating semi-natural woodland benefits many pollinator groups but is not always the best restorative solution for all pollinators. There is abundant observational evidence on the value of woodland for pollinators in intensive agricultural landscapes, including Wales-specific evidence. Wider research clarifies mechanisms by which woodland provides for pollinators, which has implications for woodland management.

In this review, “pollinators” refers to the diversity and abundance of insect species, both wild and managed, that visit flowers (in the UK almost all pollinators are insects). Pollinators transfer pollen between reproductive organs of insect-pollinated flowers, enabling fertilisation. Local species richness and abundance of pollinators are positively correlated (Hodgson et al. 2010), both of which can increase the rate of flower visitation and fertilisation (Klein et al. 2003; Garibaldi et al. 2013). However, more flower visits are a crude indicator of successful pollination (i.e. pollen deposition on stigmas; King et al. 2013). Furthermore, crop pollination is largely provided by highly effective and abundant pollinator groups. Key examples are honeybees, which are a managed species in the UK, and bumblebees (Breeze et al. 2011; Kleijn et al. 2015). As such, optimal management for pollinator diversity may differ from optimal management for crop pollination. Crop pollination is highly economically valuable, but pollinator diversity and abundance are also culturally and intrinsically valuable (Potts et al. 2016).

Pollinators are highly diverse, including subsets of bees, wasps, flies, beetles, butterflies and moths. Bees, hoverflies, butterflies and moths show evidence of decline in Great Britain (Fox et al. 2014; Powney et al. 2019; Thomas et al. 2004). Evidence is sparse in Wales, although abundance of 30 butterfly species has declined since 1970, with some recovery since 2002 (Hayhow et al. 2016). Further analysis under the Glastir Monitoring and Evaluation Programme (GMEP) revealed that declines were steepest for 7 habitat specialist species which are more restricted to semi-natural habitats (Emmett and the GMEP team 2015). Pollinator trends reflect trends in the resources that they depend on (Biesmeijer et al. 2006). For example, forage flowers for bumblebees have declined across Great Britain since the mid-1900s, possibly in association with changes in land use (Carvell et al. 2006). Woodland creation and management are likely to impact many resources that pollinators depend on. However, the magnitudes of impacts, and subsequent effects on pollinators, are not always clear.

4.1 Timescale of impact

Research on habitat creation and management for pollinators has tended to focus on open habitats, e.g. meadows and arable field margins under agri-environment schemes (AES), rather than woodland. Long-term set-aside of arable fields has been shown to affect abundance and species-richness of bumblebees, butterflies and moths within 3 years (Alanen et al. 2011), as have effects of grassland restoration on beetle communities (Woodcock et al. 2010). Woodland creation may impact pollinators during early stages, but a climax or equilibrium state may not be achieved even after decades. As such, the overall effects of woodland creation on pollinators are difficult to assess within the duration of the average research grant. This may partly explain the lack of clear evidence on the subject. There is a need for more experimental studies observing the impacts of woodland creation and management on pollinators over longer time periods. For cost-efficiency, such experiments may be best established at the same time as woodland creation initiatives.

4.2 Magnitude of impact

Pollinator species are highly diverse in their life strategies. Many groups or species are positively affected by woodland creation or management measures, but some are negatively affected. Recent work in Portugal demonstrates that overall species richness of moths is greater in wooded sites than in scrub or meadow sites, while the opposite is true for a subset of “non-forest” moths (Dantas de Miranda et al. 2019). Diaz-Forero et al. (2011) found that four bumblebee species in Estonia had significant or near-significant positive correlations with forestry, while two species had negative correlations. Research spanning hundreds of semi-natural grasslands in Sweden shows that nearby forest cover has positive effects on species richness of butterflies, including red list species (Bergman et al. 2018), although some species are negatively affected e.g. the agricultural pest *Pieris rapae* (Bergman et al. 2004). Forests have also been demonstrated to lessen the negative impacts of habitat fragmentation on butterfly species richness (Öckinger et al. 2012). International evidence suggests that flower visitation rate declines with distance from natural areas, especially woodlands (Joshi et al. 2016; Ricketts et al. 2008), as does fruit set (Garibaldi et al. 2011).

In Wales, the relationship between woodland and pollinator diversity and abundance is captured through national monitoring, e.g. the Glastir Monitoring and Evaluation Programme 2013-2016 (Emmett and the GMEP team 2017), the Environment and Rural Affairs Monitoring and Modelling Programme 2020-2022 (ERAMMP, https://erammp.wales/en) and the UK Pollinator Monitoring Scheme (PoMS, https://www.ceh.ac.uk/our-science/projects/pollinator-monitoring). A study of high nature value farmland, based on GMEP survey data, highlighted positive relationships between butterfly diversity and connectivity to broadleaved woodland (Maskell et al. 2019). More recent analysis confirms that woodland supports high abundance of pollinators, as well as high diversity. Across eight groups of pollinating insects, transect counts in broadleaved and coniferous woodlands were generally higher than counts from other habitats (Alison et al. n.d.; in prep). Broadleaved woodland ranked more highly than coniferous woodland across most pollinator groups - especially honeybees and butterflies.

A great deal of evidence demonstrates that woodlands provide nectar and pollen as forage for pollinators during their adult life stage. For example, broadleaved woodlands - alongside calcareous and neutral grasslands - are estimated to have the highest nectar productivity of habitats in Great Britain (Baude et al. 2016). Furthermore, studies analysing DNA content of honey, led by the National Botanic Garden of Wales, have highlighted the role of native woody plants including willow

(Salix), hawthorn and blackthorn (Rosaceae) and other trees (Ilex, Acer, Quercus) for honeybees in the early season (April & May, De Vere et al. 2017). Further DNA metabarcoding studies of pollen carried by hoverflies and moths have suggested frequent usage of Rubus, a genus associated with woodland and woodland edges (Lucas et al. 2018; Macgregor et al. 2019). However, beyond providing pollen and nectar for adult insects, woodland may provide for pollinators by offering:

- Nest sites and food for larvae (e.g. the brood of bees, maggots of flies, or caterpillars of butterflies and moths). Some bumblebees nest in cavities in trees; some solitary bees (especially leafcutter bees; family Megachilidae) nest in hollow twigs and stems which are probably abundant in woodland habitats. Caterpillars of a large proportion of moth species feed on leaves of shrubs and trees (Waring and Townsend 2009).
- Shelter and microclimatic refugia (e.g. for overwintering).
- Relief from artificial light at night, which impacts night-flying pollinators in well-lit areas (Macgregor et al. 2017, 2014).

Positive associations between insect abundance and woodland habitats have been presented alongside arguments in favour of “rewilding” marginal agricultural areas in Europe (Merckx and Pereira 2015). However, woodland creation is just one of many restorative solutions to pollinator declines. The apparent value of woodlands for pollinators in Wales could simply reflect (1) a lack of high-quality open semi-natural habitats, and (2) improved grassland swards that have few floral resources. A study across Switzerland, Italy, Germany and southern England highlighted that open semi-natural habitats (e.g. extensively managed grasslands) support higher numbers of wild bees and honeybees than interior areas of woodland (Bartual et al. 2019). While woodland creation may benefit pollinators, creation of open semi-natural habitats on improved land can also be used to great effect (Alison et al. 2017, 2016). Even low-effort interventions, e.g. to increase cover of flowering clover in improved grasslands, could provide non-negligible increases in forage for pollinators (Baude et al. 2016). Such interventions allow farmers to provide some resources for pollinators while minimising the impact on agricultural yield, although semi-natural habitats are still required to provide nesting habitat (Rundlöf et al. 2014).

4.3 Spatial context and connectivity dependence

There is limited evidence on the nuances of where woodland creation or management are best placed to benefit pollinators. However, internationally, studies have quantified the scales over which woodland can increase pollinator abundance in surrounding agricultural landscapes. This is an important consideration in Wales; although 88% of Wales’ land is used for agriculture, only 5% of Wales’ agricultural output comes from crops (Armstrong 2016). If the aim is to maximise benefits to crop production, priority should be given to restoration of small woods on unproductive land that is near to pollinator-dependent crops.

There is clear evidence that effects of woodland on crop pollination are distance-dependent. For example, Joshi et al. (2016) observed that positive effects of proximity to woodland on flower visitation were apparent within 500m. Similarly, Bergman et al. (2018) found that, for the majority of the 30 most common butterfly species in their study, there were strong positive responses to the amount of forest cover within 200–500m (although an earlier study only found effects at scales >2km; Bergman et al. 2004). Diaz-Forero et al. (2011) found that some bumblebee species responded positively to forest at small spatial scales (250-500m) while other species responded negatively at large spatial scales (1-2km). Ricketts et al. (2008) found that visitation rate declined more steeply than pollinator richness with increasing distance from natural or semi-natural habitats (half maximum at 0.6km and 1.5km respectively). Garibaldi et al. (2011) observed continuous declines in visitation and fruit set up to >3km from natural habitats. In general, when creating or restoring any habitat, the existence of source populations will limit colonisation potential (Hanski 1994), particularly for rare and less mobile species.

4.4 Woodland types and management regime

Dense, intensively managed conifer plantations have little value for pollinators. For example, a study of plantation forests in Ireland found that nearly 80% of hoverfly species recorded were associated with open space habitats rather than closed-canopy forest (Gittings et al. 2006). Furthermore, some hoverfly species were positively associated with the presence of broadleaved woody vegetation and wet habitat features (e.g. ditches and water saturated ground).

Guidance on woodland management for pollinators produced by DEFRA (DEFRA and Forestry Commission 2014), Buglife (Falk and Buglife 2019) and Butterfly Conservation (Clarke et al. 2011) emphasises the importance of maintaining woodland edges, rides and clearings. Good management of rides and clearings for pollinators will maximise the area receiving sunshine, prevent “wind tunnels”, introduce sown wildflowers, and introduce or encourage broadleaved native shrubs and trees. Sallow (or goat-willow; Salix caprea) and hawthorn (Crataegus monogyna) are particularly likely to benefit pollinators because they provide pollen and nectar in abundance in the early season (Nowakowski and Pywell 2016) and are food plants for caterpillars of a wide variety of butterfly and moth species (Clarke et al. 2011; Waring and Townsend 2009). Maintenance of a variety of successional stages within woodlands, e.g. through coppice management, helps to ensure that forage is available for insects throughout the year.

4.5 Risks of woodland creation and how to manage them

Woodland creation can pose risks to a wide range of outcomes, including agricultural productivity, landscape cultural value and soil carbon (on some soil types). Similarly, woodland creation can pose risks to pollinators, especially to species associated with open semi-natural habitats. Negative effects of woodland creation could result if woodland form obstacles to bees foraging in open habitats, for example (Goulson et al. 2010). As such, managing risks to pollinators involves understanding the starting point of woodland creation; on species-rich wildflower meadows, woodland creation is likely to have little benefit or even detrimental results for pollinators. On the other hand, in Wales acid grasslands appear to be a low starting point in terms of pollinator abundance (Alison et al. n.d., in prep). It is also important to consider the end point of woodland creation; from most starting points, dense stands of conifers are unlikely to provide increases in resources for pollinators. It is also important to recognise that the ideal start and end points for pollinators will sometimes differ from those for e.g. soil carbon or productivity.

If forests are created and managed for intensive production, chemical insecticides and similar pest control measures have the potential to negatively impact pollinators. A report from the Forestry Commission identified that some chemical insecticides (e.g. used to control pine weevil) can pose a high risk to non-target insect species (Willoughby et al. 2004). The same report suggests a range of alternative management measures, as well as advice to minimise non-target impacts of pesticides. These include mixed cropping and biological pest control, which may have neutral or even beneficial results for pollinators.

Woodland creation could increase the abundance of diseases, pests, predators and competitors, some of which are non-native. Outcomes for pollinators are difficult to predict due to the complexity of possible species interactions. Risks could be exacerbated if goods and/or trees are imported for woodland creation and management. One possible risk would be increases in the Asian hornet *Vespa velutina*. This species has been repeatedly sighted in Southern England in recent years and poses a threat to honeybees. It is possible that honeybees themselves act as non-native competitors to native pollinators. However, one large international study found that pollination by managed honey bees supplemented, rather than substituted for, pollination by wild insects (Garibaldi et al. 2013).

5. SOIL MICROBIOME

Both prokaryotic and eukaryotic microorganisms make up a significant component of the soil microbial biomass and play key roles in a variety of soil functions. In temperate forest systems the fungi have received perhaps the most research attention, due to their known ability to degrade recalcitrant plant biomass through the production of various extracellular enzymes. In addition to these saprotrophic activities, mycorrhizal fungi which live in close associations with plants also are known to play key roles in nutrient mobilisation and transport. More recently, with the development of molecular methods, there is increasing awareness of the abundance and potential important role of other microbes such as the bacteria in woodland systems, though there is a general lack of understanding as to their potential functional roles (Llado et al. 2017).

5.1 Land-use and land-use change

With respect to overall biodiversity metrics, George et al. (2019) in the GMEP soil survey of Wales reported on soil diversity across all broad habitat types for microbes and mesofauna at a national scale. The study indicated woodland species richness was intermediate between cropland and heathland bog across a land use and soil pH gradient. There were divergent trends for bacteria and fungi diversity (Operational taxonomic units; OTU richness) relative to archaeal (and soil animals - see next Section) with greatest diversity of bacteria and fungi recorded in intensively managed, high pH soils such as cropland. In contrast, there was greatest OTU richness for archaeal (and soil animal) species in the unimproved, acid soils. Differences between upland and lowland woodland followed these trends for bacteria and fungi with greater OTU richness in lowland woodland relative to upland woodland. (There was no difference in soil animal OTU richness between upland and lowland woodlands.) Overall, as for other groups of species this emphasises the point that different groups have different requirements and that soil biodiversity as an overall concept has little meaning without stating which taxa or functional group is being discussed. It should be noted, this study was a broad landscape scale survey, not examining afforestation per se, and these results likely reflect differences in soil abiotic properties and land management intensity consistent with where upland and lowland woodland are located, rather than forestry effects per se. Importantly, uncertainty still surrounds whether simple metrics such as microbial biodiversity indices, or indeed biomass can be reliably translated to ecosystem benefits.

Specific studies addressing afforestation are notably few in number. Historically there have been several studies examining afforestation with respect to pine plantations. In a review (Chen et al. 2008), it was concluded that afforestation of grassland with coniferous trees reduces soil microbial biomass, soil respiration and enzymatic activity, signifying reduced soil fertility. The same study identified a shift from bacterial dominance to fungal dominance following ~20 years of pine plantation in contrast with adjacent grassland, though this was assessed using culture-based methodologies that may not be fully quantitative. Mitchell et al. (2010) used a long-term field experiment where Birch (*Betula pubescens*) had been planted into heather moorland 20 years previously. Total microbial biomass (as measured using PLFA[2]) was lower under 20-year-old Birch compared to heather moorland and the fungal community diversity (as measured by PCR-DGGE[3]) was altered under Birch.

5.2 Woodland types and management regimes

Desie et al. (2019) used Biolog EcoPlates to show a greater microbial functional diversity in topsoil under deciduous forest (*Quercus*, *Fagus*, *Carpinus*) compared to topsoil under Spruce. These methods, though functional in nature, have limitations in that they are known to only reflect change in organisms which are cultured within the assay. Several studies have also examined the impacts of different tree species on molecular measures of soil microbial diversity (e.g. Ayres et al. 2009). This study found different communities of bacteria and fungi under trembling aspen, lodgepole pine, and Engelmann spruce in native trees in North America. Whilst many other studies of this type exist globally, no synthesis of findings has been reported. It is apparent however that different trees, through modifying soil abiotic properties (indirect mechanisms) can have large effects on soil microbial communities (Tedersoo et al. 2016; Prescott & Grayston 2013). The direction and extent of change with afforestation on different soils will likely therefore be dependent on existing soil conditions prior to establishment and type of tree planted. Direct interactions between tree species and symbiotic fungi such as the ectomycorrhizal ericoid or arbuscular mycorrhizal fungi are also proposed (Baldrian 2017), yet we were unable to find studies that had directly quantified the influence of indirect versus direct effects of trees in structuring the soil microbiome.

5.3 Functional relevance of microbial biodiversity change

Singh et al. (2009) showed that afforestation (Pine) of pastures in New Zealand changed bacterial methanotroph abundances and was consistent with higher rates of methane consumption in the forested soils. Indeed, soil bacterial methylotrophy could be a key ecosystem service benefit of afforestation of agricultural land more generally, with another paper also demonstrating key linkages between rates of methane consumption and community structure in Scottish land use contrasts (Nazaries et al. 2013). Additionally this study demonstrated that the magnitude of woodland benefit depends on the contrasting land use type, with bogs and moorland have higher methane emissions than grassland (note that a more thorough review is required looking at consistency in afforestation effects on greenhouse gas (GHG) fluxes specifically - our focus here is specifically on biodiversity). More studies of this kind are likely needed to prove generality, and ideally looking at multiple microbial functions. It is known that methane consumption is generally restricted to a few specific bacterial lineages, so it is unclear if other more phylogenetically distributed functions (e.g. broad carbon cycling) will be affected to the same degree. Other important processes such as N_2O emissions have been shown to be reduced by afforestation of grassland in Ireland (Mishurov & Kiely 2010), however we were

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[2] Phospholipid-derived fatty acids (PLFA) - chemotaxonomic markers of bacteria and other organisms
[3] Polymerase Chain Reaction (PCR) - Denaturing Gradient Gel Electrophoresis (DGGE)

unable to identify studies which had linked such effects to changes in the microbial community.

5.4 Effect of afforestation on soil biodiversity mediated by change in soil properties

Global studies utilising molecular techniques reveal that in general soil microbial communities are predominantly structured by abiotic properties, with tree species specific effects only usually apparent from controlled manipulations, or highly localised studies on the same soil type. For bacteria, factors relating to soil pH are highly associated with community diversity, composition and structure. Fungal communities are known to be broadly sensitive to pH variation, but it has been proposed that other plant and soil organic matter related factors may be more important. In the absence of detailed meta-analyses of afforestation effects on microbial communities, it may be possible to make site-based predictions of likely change through predicting soil abiotic effects. Several meta-analyses on soil abiotic effects have been undertaken, with one revealing global soil pH reduction alongside decreases in nutrient cations (Ca, K, Mg), and increases in soil C:N with afforestation (Berthrong et al. 2009). This contrasted with a survey of afforestation across Northern China which revealed a complex, site and species specific effect (Hong et al. 2018). Here it was found that afforestation lowers pH in alkaline soil but raises pH in acid soil, the extent of which was modulated by tree species.

5.5 Summary and outlook for afforestation effects on soil microbes

Overall it is difficult to use existing evidence to explicitly predict effects of afforestation on soil microbial communities, as there just isn't the breadth of research literature available relevant to the Welsh context. However there are opportunities to make some likely predictions by leveraging existing Welsh/UK datasets, and using them to explore effects of forests on soil properties in relation to other geographically constrained land use types. Such findings could then be extrapolated to make likely predictions on soil biodiversity responses. Since existing datasets such as the Welsh GMEP survey (George et al. 2019) and British Countryside Survey (Jones et al. 2020) also have associated molecular biodiversity information, it would be possible to then test and validate the predictions. If successful, there is still the challenge of translating biodiversity information to functional information pertaining to soil ecosystem services. However new whole genome sequencing approaches are now becoming available which could be used to monitor functional effects following the onset of trials, linked to process measures such as GHG fluxes.

6. SOIL AND SAPROXYLIC INVERTEBRATES

Invertebrates include a huge taxonomic and functional diversity of different organisms. There is a wide literature on the effects of trees and woodland types on different broad soil and litter-dwelling invertebrate groups (e.g. nematodes, collembolans, mites, isopods [or woodlice], annelids [or earthworms]) and evidence of impacts on their abundance and diversity is generally strong and well accepted. There are also saproxylic invertebrates (i.e. those with a direct dependence on dead trees or wood) that are an important component of biodiversity in wooded habitats.

6.1 Land-use as dominant driver of soil invertebrate diversity

Land-use has a dominant effect on the abundance, composition and diversity of soil invertebrate communities, largely through determining the flow of energy (inputs of organic matter and nutrients cycling) into the soil, and the level of physical disturbance to the soil. The creation of new woodland will likely represent a substantial shift in land-use and land-use change has clear impacts on soil invertebrate biodiversity. The magnitude and timing of impacts of woodland creation on soil invertebrate biodiversity will depend strongly on the previous land-use.

Microarthropod soil and litter communities are strongly influenced by land-use. In Wales, George et al. (2017) looked at microarthropod communities (predominantly mites and collembolans) from a national soil survey under the Glastir Evaluation & Monitoring Programme (GMEP). While variability within land-uses or habitats was high, as may be expected in a national survey, total microarthropod abundance was found to be greatest under lowland and upland woodland vegetation classes (proxies for land use). A follow-up paper, which used a molecular approach to assess diversity in soil, showed that the mean and variability of the richness of animal OTUs (Operational taxonomic units) was similar under all vegetation classes except Crops/Weeds, where it was lowest (George et al. 2019). (Note that this is the opposite of the OTU richness for bacteria and fungi which was greatest in Crops/Weed habitat class.) A large-scale study of mites across habitats in Ireland (Arroyo et al. 2013) highlighted that abundance and alpha diversity of Oribatid mites was greatest in coniferous woodland and rough grazing habitats compared to intensive agricultural habitats.

Chronosequence studies of afforestation support changes in diversity moving to woodland systems. For example, there were increases in abundance and species richness of Oribatid mites moving from heather moorland to birch woodland (Osler et al. 2006), and similar findings were shown for afforestation of moorland by Scots pine (Horwood and Butt 2000). Generally, microarthropods tend to have greater biodiversity in forest habitats compared to other land use types and this is likely to be driven by greater heterogeneity in both resources and microhabitats, and more limited disturbance. Woodland creation, whether on agricultural or unmanaged land, will therefore alter the composition of soil microarthropods and is likely to increase their diversity.

Earthworm populations are also strongly influenced by land-use. The abundance and diversity of earthworm communities in agricultural soils is greatly influenced by land management practices; cultivated systems tend to contain a relatively low abundance of earthworms (50–200 individuals per square metre), somewhat depending on time since cultivation and crop type, and 3–5 species per square metre, whereas grassland or pasture systems generally contain much greater abundances (300–600 individuals per square metre) and 4–7 species per square metre (Rutgers et al. 2009; Schmidt et al. 2011). Forest soils, particularly broadleaf woodland on mineral soils, can also have substantial earthworm populations. The mean species richness of earthworms in forest sites in Ireland was 4 per square metre; this study included five broadleaf and five coniferous woodlands (Keith et al. 2012). The conifer woodlands, however, tended to be on organic soils. Across all sites a total of 10 earthworm taxa were recorded in both broadleaf and coniferous woodland (Schmidt et al. 2011); this is similar to the number of taxa found in different woodland types found by Ashwood et al. (2019). Vanbergen et al. (2007) found that, at the landscape scale, soil faunal richness (Collembola and Lumbricids) had a unimodal relationship with the percentage of forest cover.

There is evidence indicating the impacts of forest planting on earthworm biodiversity depend on previous land-use. Work by Ashwood et al. (2019) used a chronosequence approach to examine earthworm communities and soil quality in broadleaf woodlands that had been created on former agricultural land. While the range of earthworm abundance and biomass was similar between arable and woodland sites, earthworm diversity (as measured by Shannon H diversity) was greater in woodlands. In the same study, pasture grassland had a greater earthworm abundance than the arable and woodlands but similar diversity to the woodlands. Ashwood et al. (2019) also showed that, compared to arable, there were clear functional shifts under woodland, with greater proportion of anecic (deep-burrowing) and epigeic (surface/litter dwellers) under woodlands. In summary, the impacts of woodland creation on earthworm abundance and diversity will depend on whether the previous land use is agricultural or unmanaged.

6.2 Tree species composition and soil invertebrates

Tree species composition is an important determinant of soil invertebrate abundance and diversity, despite broad land use type being a dominant driver. There is a large body of literature looking at the effects of individual tree species, and tree species composition, on soil invertebrate populations.

Nematode worms are a key component of the soil food web with high diversity and representing a range of trophic groups. Mitchell et al. (2012) looked at the impact of five tree species (*Betula pendula*, *Betula pubescens*, *Sorbus aucuparia*, *Quercus petraea* and *Pinus sylvestris*) on the density of soil nematodes in a mesocosm experiment in NE Scotland. Nematode density was greater under *B. pubescens* and *Q. petraea* in the absence of *Calluna* understory, though the difference was not significant. A novel study by Cezarz et al. (2013) used clusters of trees (including *Fagus sylvatica*, *Fraxinus excelsior*, and *Tilia cordata*) to examine the effects of tree identity and diversity effects on the soil food web, through nematode community analysis. This study showed that tree species identity had a dominant impact on nematode trophic composition, via differences in root and litter traits between the tree species (Cezarz et al. 2013).

Studies looking at effects of tree species have often taken place using common garden experiments (e.g. Muys et al. 2005; Reich et al. 2005). For earthworms, Muys et al. (1992), showed differences in the taxonomic and functional composition of earthworms in a forest experiment In Belgium. This experiment was planted on former meadow grassland and included tree species such as Alder (*Alnus glutinosa*), Ash (*Fraxinus excelsior*), Lime (*Tilia platyphyllos*), Oak (*Quercus palustris*), Sweet Cherry (*Prunus avium*). Oak and Sweet Cherry on tended to have the greatest dissimilarity in effects on abundance of three earthworm functional groups (Muys et al. 1992). Similarly, Reich et al. (2005) used a common garden experiment with monocultures of 14 tree species; this study found, across all experimental plots, a positive relationship between litter calcium concentration and earthworm biomass.

While individual tree species can determine earthworm composition, there are generally clear and consistent effects of broad tree types (i.e. conifer v deciduous) on earthworm communities in forest soils (e.g. Reich et al. 2005; Schelfhout et al. 2017). This appears to be driven by differences in litter quality and subsequent effects on soil characteristics. In an analysis of forest plots across Europe, De Wandeler et al. (2018) showed that tree litter characteristics had a significant impact on earthworm biomass, being negatively related to the proportion of evergreen leaf litterfall. In a meta-analysis of land-use change effects on earthworms, Spurgeon et al. (2013) found that effect sizes of earthworm biomass change were mostly negative moving from grassland to coniferous woodland; effect sizes of earthworm biomass change from grassland to deciduous woodland were, however, spread from negative to strongly positive.

Although the functional identity of tree species in woodlands has been shown to have the strongest impact on invertebrates (compared to tree species richness *per se*), there is argument that tree species richness is beneficial to wider biodiversity. Ampoorter et al. (2020) looked at data from a European forest diversity experiment (the same experimental platform as De Wandeler et al. 2018) and compared factors driving diversity and abundance/activity of different groups (including bats, birds, spiders, earthworms, microorganisms, understorey plants). It was found that a metric of ‘multi-diversity’ was positively correlated with tree species richness.

6.3 Deadwood

Deadwood in forest systems provides resources and habitat for saproxylic invertebrates. Hodge and Peterken (1998) noted that 34% of scarce woodland invertebrate species (264 out of 771) require deadwood. In particular, beetles (Coleoptera) constitute a large proportion of saproxylic invertebrate species in forests. Saproxylic invertebrate diversity is generally considered to be under threat throughout Europe, due to increased removal of deadwood from landscapes and shifts toward intensive commercial forestry (Davies et al. 2006). Woodland creation and management will determine the supply of deadwood and its dynamics, thereby impacting habitat suitability for saproxylic species and resulting levels of diversity.

The volume of deadwood varies between different types of woodland. For example, ranges of standing deadwood volume were reported at 4–20 m^3 ha^{-1} in conifer plantations, 40–50 m^3 ha^{-1} in native pinewoods, 20–31 m^3 ha^{-1} in managed semi-natural broadleaved 50-100 year old forest, and 50–130 m^3 ha^{-1} in unmanaged semi-natural broadleaved woodland (Hodge & Peterken 1998). However, in the recent survey of Woodland Ecological Conditions in Wales, 45% of native woodland stands were found to have no deadwood within them and 28% less than 10 m^3 of deadwood per hectare (Ditchburn et al. 2020b). This suggests that existing woodlands need to be better managed for provision of deadwood if the promotion of saproxylic invertebrate diversity is considered a priority.

A meta-analysis by Lassauce et al. (2011) examined the correlation between deadwood volume and saproxylic species richness, reporting a positive relationship. However, this study found that total deadwood volume is not a sufficient indicator of saproxylic biodiversity and, in temperate systems, the positive link between deadwood volume and beetle richness was too weak. It was concluded that past management and the types of deadwood were likely more important, and also that saproxylic richness may not respond linearly to deadwood volume. These general findings were supported in a more recent study by Sandström et al. (2019) which reported a systematic review on the effects of dead wood manipulation on abundance and diversity of saproxylic insects and other groups. Enrichment of deadwood through creation (i.e. using *in situ* trees as a source) and addition (i.e. using wood from external source) had positive effects on abundance and richness of saproxylic insects, including rare species. This study also found that burning benefited saproxylic abundance and richness more efficiently than creation or addition of deadwood, with similar effect sizes from approximately half the enrichment of deadwood volume (Sandström et al. 2019). Consequently, quantity is probably not as important as qualitative aspects of deadwood stocks such as structural diversity and presence of deadwood at different stages of decay.

Deadwood also acts as a habitat and resource for earthworms in forest systems but it is not typically assessed in studies of earthworm diversity (Ashwood et al. 2019). Ashwood et al. (2019) recorded 7 earthworm species present in deadwood microhabitat of an oak-dominated broadleaf woodland.

6.4 Timescale of impacts

The impacts of forest planting on invertebrate diversity will depend on the time since planting (i.e. woodland maturity). There are likely to be short-term impacts (i.e. after several years) on populations of species already present. It is over a decadal timeframe, however, that significant changes in soil invertebrate biodiversity are likely to take place, as species are able to migrate into new habitat. This will also depend on the landscape context of woodland creation and management.

In the meta-analysis of impact of land use transitions from grassland to woodland on earthworms (Spurgeon et al. 2013), the effect size of changes in earthworm abundance was highly variable within different classes of time since management change. However, effect sizes tended to be positive earlier in the sequence (4 to 10 years) and more negative later in the sequence (11 to 15 years, 25 to 60 years, 60+

years). These early positive changes in abundance may be linked to the cessation of grazing and associated alleviation of soil compaction. Ashwood et al. (2019) found that, compared to arable land use, that young secondary woodland (50 to 60 years old on former agricultural land) had the highest species richness and diversity of earthworms.

The length of time that it takes to evidence significant change in soil invertebrates is also partly determined via changes in soil properties. For example, earthworm diversity is also controlled by inherent soil properties such as pH and texture. In highly organic soils where pH is low, such as in heath and moorland, earthworms are present but typically at low densities and dominated by particular epigeic species (e.g. *Bimastos rubidus*, *Lumbricus eiseni*). Desie et al. (2019) showed that changes in soil pH, arising from the transition between deciduous and coniferous species cover, were key determinants of earthworm biomass.

7. BIRDS

7.1 General issues for woodland intervention effects on birds

There is good evidence for effects of woodland management on birds, but the response speed of woody vegetation is generally slow, creating challenges for monitoring. In addition, bird population responses to management change may take several years to be detectable, both due to intrinsic rates of increase being low and to the confounding effects of harsh winters and other environmental variation. Impacts can be large, locally, and very significant at the population level, for specialist woodland species, but for generalists the low area footprint of woodland relative to other habitats means that this habitat will have a relatively small influence.

Management can be divided into creation of new woodland and the management of existing woodland, with the latter divided into protection and active management. In most cases the time lags between intervention and woodland response are long and uncertain. This often means that the ability of an intervention to deliver long-term outcomes are assumed rather than supported by well-replicated observational evidence.

A misconception with some people may be that woodland left to natural processes would be better for birds. This is likely only to be true on a timescale of several centuries, in which natural processes can establish in mature forest. On a decadal timescale, management of woodland, especially of the planted or heavily modified woods found in the UK, will be needed to maximize habitat quality, to establish features such as canopy gaps and heterogeneous or dense understorey development. This is especially important because the species that are accepted as conservation priorities are those that have been found in woods in recent decades, so are inevitably those that prefer the habitats that they provide. The bigger question of which species should be prioritized is beyond the scope of a review like this, which can only realistically consider the priorities that have been set by independent policy processes.

As reflected in the literature on woodland creation that is described below, woodland area and landscape context are important influences on bird usage of woodland patches. This is partly because woodland specialists are likely to need large, contiguous areas of habitat into which sufficient numbers of breeding pairs can fit to sustain a local population, and partly because birds in general are mobile and respond to landscape variation at large spatial scales (e.g. Pickett & Siriwardena 2011). Further, given the fragmented nature of much woodland habitat in Wales and the UK more generally, many of the species that are associated with woodland would have been found in edge habitats in the prehistoric past and are also found today in habitats such as hedges, gardens and scrub. Their populations in woodland per se are therefore likely to be deeply connected to and influenced by the surrounding habitats for example as sources or sinks, or in respect of gene flow.

7.2 Woodland creation on farmland

Woodland creation involves a complete change in habitat coverage, so impacts on species presence and abundance are inevitable. Research considering created mature woodland has yet to be conducted because of the inevitable decades of time lag. However, farm woods of around ten years old attract scrub, hedgerow and open-country bird species (Vanhinsbergh et al. 2002), while the same woods, at c.30 years old, attract more woodland species, but with rather little difference in total species composition (Dadam et al. 2020). Differences in community structure from the 1999 survey were small, but Simpson's diversity was marginally higher and the more mature habitat in 2019 supported higher densities of 37 species, but lower ones of 23 species. Many decreasing species were those more associated with scrub or open habitat, but also included species that have declined nationally (Dadam et al. 2020).

New woodland location and characteristics have significant effect on colonization and use by birds. Dadam et al. (2020) characterized woodland connectivity in terms of areas (weight of surrounding habitat) and numbers of nearby patches (numbers of point sources), but these were fairly highly correlated, so the potential to discriminate between these effects was limited. The connectivity analyses showed that, although patterns varied between species, woodland connectivity generally had a negative effect on abundance at the local scale, but a more mixed effect (and often a positive one for specialists) at the landscape scale. This suggests that the use of farm woodland patches by birds during their daily activity is lower where there is more nearby woodland, possibly because this habitat is more mature and provides better or more resources. Conversely, where there is little surrounding woodland locally as an alternative source of resources, perhaps birds use farm woodlands more. At the landscape scale, there is then some evidence that specialists (in particular) are more likely to colonize new woodland plots in more heavily wooded landscapes. Adding woodland to less wooded areas at the local scale that have more broadleaf woodland and less coniferous woodland at the landscape scale is likely to deliver larger local populations.

More complex plot shapes (longer perimeters per unit area) were associated with lower abundances for 33 species, including 12 specialists (Dadam et al. 2020), suggesting that there was no strong preference for edge habitats across the assemblage, although deviation between edge and core habitats would be expected to be greater in mature woodland.

Larger new woodlands support higher local abundances of most species, as would be expected, but the pattern is generally for the increase to level off (Dadam et al. 2020), so there would be optimal plot sizes and numbers of plots for each species, given a particular target woodland area. With more mature woodland, however, the richness of woodland specialists (i.e. typically conservation targets) continues to rise at the expense of that of generalists, for larger woods up to 120ha (Gardner et al. 2020). Therefore, with a very long-term (multiple-decade) focus for priority species, larger woodlands would be recommended. Note, however, that, despite their lower biodiversity, smaller woodlands can deliver multiple ecosystem services better than larger woodlands of similar age, via the effects of non-avian biodiversity (Valdés et al. 2019).

Whytock et al. (2017) considered a wider range of woodland ages, but a smaller range of woodland covers in the landscape, but found similar results: birds were surveyed in 101 secondary, broadleaf woodlands aged 10–160 years with 80% canopy cover and in landscapes with 0-17% broadleaf woodland cover within 3000 m. Local patch characteristics were relatively more important than landscape characteristics for bird communities, and biodiversity responses to habitat creation depended on local- and landscape-scale factors that interacted across time and space.

In a specific Welsh AES context, woodland establishment under Tir Gofal was tested for effects on bird population growth rates by Dadam & Siriwardena (2019), showing mixed effects. Of ten mostly generalist species, there were significant or near-significant effects for five species, of which three were positive. Note that these woodlands would have been less than 20 years old at the end of the period of evaluation.

Dolman et al. (2007) reviewed the evidence for patch area and composition effects on woodland birds globally, finding that larger woodlands support more woodland bird species, and that woods located within sparsely wooded landscapes are less valuable to specialist woodland species. Species found in small woods generally also occur in large woods, but small woods may be preferred by a few edge species and are more variable in bird assemblage composition. They concluded that the metapopulation dynamics of specialist species with poor dispersal (typically those of most conservation concern) shows that creating or buffering large woodlands is more efficient than a greater total area of small fragments. Connectivity among smaller fragments appears to benefit widespread generalist species. However, this study considered patterns among contemporaneous, mature woodland, not among newly created habitat, so the conclusions would relate to a hypothetical, very long-term context with no gross environmental change, if they were applied to inform new woodland planting. This evidence should therefore be weighed against that from direct studies of woodland creation, where there are apparent conflicts in consequent recommendations for best practice, considering the timeframe and species range that are of interest for target-setting.

In general, woodland creation will inevitably boost woodland bird populations, assuming that they can find the new habitat. Time lags, especially for specialists, are very long, so a focus of multiple decades is required, opening a potential issue with unknown interactions with climate change. There is good evidence around landscape context influences to guide plot placement and size/shape, but detailed responses and ideal plot types/locations will vary between species. Larger woodlands, or extending existing patches, is likely to be more effective for target species, but this will not always be true for all species and service provision.

The risks of woodland creation relate to non-woodland issues: (a) the habitats that are replaced and (b) effects of the new habitat boundary or heterogeneity that is created. With woodland creation on farmland or moorland, the replaced habitat typically has low biodiversity value (few species present) and many species associated with boundary habitats are also found in woodland. Increased heterogeneity at the landscape scale is also often a positive influence on species abundance, although not for all species, and for community diversity as a result (Pickett & Siriwardena 2011). However, woodland creation facilitates the presence of predators in otherwise treeless landscapes, with potential negative effects on other species. This can be due to real or perceived predation risk, as per the common avoidance of tall habitat structure by ground-nesting birds (e.g. Chamberlain & Gregory 1999). The principal evidence for negative effects on population viability involves upland-nesting wading birds and plantations (Amar et al. 2011, Douglas et al. 2014). This indicates that careful consideration of geographical context is important to minimize impacts on sensitive species of conservation concern.

7.3 Woodland fencing to reduce browsing

A major cause of woodland bird decline in recent decades is believed to be loss of understorey (and tree and shrub diversity) due to browsing by increasing populations of native and introduced deer (Donald et al. 1998, Hewson et al. 2007, Fuller et al. 2007, Holt et al. 2011, Fuller et al. 2014). In Wales, sheep grazing of upland woodland is also likely to be a negative factor. From Fuller et al.'s (2014) review, browsing probably does not affect birds in canopy, but there is strong evidence that it reduces nesting and foraging habitat in the low shrub layer below 2 m and reduces the herbaceous component of the field layer, increasing coarse grasses and sedges; impacts on young coppice regrowth are particularly strong. There is good evidence that deer have hence reduced numbers of Willow Warbler, Garden Warbler, Song Thrush, Nightingale, Dunnock and Bullfinch (Fuller et al. 2014). However, there is less evidence for the mechanism, i.e. direct evidence for effects on food supplies or nesting (Fuller et al. 2014).

Eichhorn et al. (2017) used terrestrial laser scanning to quantify woodland structure in an English region and one in the Welsh borders and found a 68% lower density of understorey foliage (0.5-2 m above-ground) in high-deer woodlands, although total amounts of foliage detected across the full canopy did not differ between deer density levels, because high-deer sites were 5 m taller overall, for uncertain reasons. The results suggest that reduction of deer pressure is likely to have a strong impact on woodland structures and aid in restoring the complex understorey habitats required by many birds. In principle, this could be achieved by culling deer (a landscape-scale intervention) or by fencing individual woodlands.

Following studies in North America, Holt et al. (2011) showed that deer activity can alter woodland bird assemblages with a replicated split-plot exclusion experiment in English coppiced woodland. Deer browsing strongly altered vegetation structure such that significantly more ground and understorey foraging birds were found where deer were excluded, including both generalist and specialist species, including Nightingale *Luscinia megarhynchos* (Holt et al. 2011). No significant positive responses to browsing were detected. For one model species, blackcap (Sylvia atricapilla), there were also negative effects of browsing on singing behaviour and body condition (Holt et al. 2013). The effects of deer browsing apply in both spring and winter (Holt et al. 2014). It is noteworthy that this study was set up as a test of the effects of deer browsing, but also functions effectively in providing evidence of the benefits of fencing, albeit only for small plots, rather than whole woodlands. The effects were detected after two-nine years after fencing was introduced.

Fuller et al. (2014) surveyed birds in 300 broadleaved stands (median area 3.3 ha), thought to be typical of lowland England, structured *a priori* by deer abundance (high, medium, low), but there were few clear patterns in the relationships between abundance of individual bird species and deer abundance. Given the weight of independent evidence of effects here, this lack of effect probably reflects a mismatch between the scale of sampling and that of variation in deer, or that deer effects are patchy within broad areas of overall deer density.

Under Tir Gofal and specific application of grazing reduction options, Dadam & Siriwardena (2019) found a clear pattern for positive effects on bird population growth rates: reducing/excluding grazing had significant or near-significant effects for 4/16 species and managed stock grazing had significant or near-significant effects for 7/16 species, all of which were positive. This provides good evidence for community-level benefits of the interventions on the timescale of the life of Tir Gofal, from 1999 to 2014. This has also been considered under Glastir by GMEP, considering habitat selection of Glastir woodland, but revealed little evidence for selection or avoidance of managed woods (http://gmep.wales/). However, the timing of sampling here was such that effects would have had little time to develop. Issues with habitat data may mean that these tests would benefit from being repeated.

7.4 Woodland thinning

The literature review by Fuller et al. (2014) examined the evidence of the impacts of woodland management on woodland habitats and bird populations. There was clear evidence that silvicultural management can be a major influence on local bird community composition, but the largest impacts of management derive from the scale and rotation length of harvesting systems. Both clearfell and coppice produce a larger proportion of young growth stages which favour bird species depending on open ground or dense low shrub growth. Normal silvicultural thinning does not tend to stimulate sufficiently large changes in the understorey to affect most birds. The field survey also reported by Fuller et al. (2014), which used field sites that were stratified on the basis of explicit documentation of interventions (or lack thereof) at the stand level in the previous 20 years, further revealed few clear patterns of management effects on birds. Where interventions had occurred in the study plots, they were considered likely to be mainly by canopy thinning and small patch felling.

Eichhorn et al. (2017) considered management effects in a subset of the full sample of plots in Fuller et al. (2014), measuring vegetation structure using terrestrial laser scanning. Managed woodlands exhibited smaller differences from controls than high-deer plots (i.e. no difference in foliage density), but including a lower quantity of stem material at heights from 2 to 5 m. This supports the conclusion that (thinning) management interventions, as currently practised, have limited and inconsistent effects on the habitat and birds.

7.5 Coppicing

Fuller et al. (2014) reviewed the evidence for coppice management effects on birds and found that the principal impact on bird habitats, supported by strong evidence, was the temporary provision of dense young woody vegetation approximately 3-8 years after cutting. Young vegetation in coppice woodland is denser than equivalent vegetation on replanted areas after clearfelling. The short rotation of coppice woodland also leads to a higher proportion of young growth stages than high forest with clearfell. This dense low vegetation principally benefits warblers and other shrub-nesting species. The few years immediately after harvest also provide conditions suitable for some ground-nesting species.

7.6 Harvesting and replanting

Fuller et al.'s (2014) review found that there was strong evidence for the effects of clearfelling. These effects depended on the proportion of timber trees that are harvested and on whether replacement is through natural regeneration or replanting. Most information on complete clearfell and replant is from conifer plantations, where newly cleared ground provides habitat for certain ground-nesting birds. This then develops after replanting to dense low shrubby vegetation, often containing bramble and birch amongst the crop trees. This stage through to canopy-closure provides habitat for many shrub-nesting species.

Considering group felling, selective felling and shelter wood, each of which remove only a proportion of the canopy trees, Fuller et al. (2014) found that, if the proportion of canopy removed is small (<40%) this is likely to have a similar (lack of) effect on bird habitats to canopy thinning. Where a large proportion of the canopy is removed (>80%), the effects are likely to be similar to that of clearfell but the retention of some mature broadleaf trees may maintain some habitat for particular canopy-feeding birds, although this would depend on the tree species involved. However, there was little specific evidence to support this conclusion.

7.7 Restoration of native woods (e.g. Plantations on Ancient Woodland Sites, PAWS)

This intervention largely involves the removal of non-native trees and encouragement of natural regeneration of native tree species, to provide a more varied age structure. Fuller et al. (2014) reported the impact of plantation conversion on bird habitats is determined largely by the proportion of canopy removed. Where the plantation is removed completely by clearfelling, the results are similar, so a temporary increase in open and low shrub habitat potentially benefiting associated bird species. Where the plantations contain native trees, usually only the non-natives are removed, resulting in changes to bird habitats similar to thinning and potentially having similar effects. Fuller et al. considered the evidence for these effects to be of medium strength.

7.8 Deadwood

Fuller et al. (2014) found little information from the UK on the effect of dead wood retention or provision on bird habitats. Removal of brash or fallen trees can remove nesting cover for some bird species and the creation of standing dead trees or snags has the potential to create suitable nest sites for hole-nesting birds, but many of these will also nest in holes in live trees and artificial boxes. Snags also provide food such as the larvae of bark beetles for some birds. Only rather rare species such as

Lesser Spotted Woodpecker and Willow Tit are very likely to benefit from creating dead wood, but the specific evidence for such effects is sparse.

7.9 Integrated woodland management - Woodland Improvement Grants

The East Midlands Woodland Improvement Grant (WIG) project aimed to improve populations of regionally important woodland bird species in a sample of woodlands under this management. The original East Midlands woodland bird project ran from 2008-2014 and was an advisor-led project which aimed to encourage woodland owners/managers to increase woodland management to improve habitat suitability for breeding birds. The management mainly involved manipulating stand structure, including reducing canopy density to stimulate lower understorey growth, reducing dominance of conifers on sites where present, and improving ecotone structure along permanent open space such as rides and glades. A sample of 64 sites from nine estates were selected for baseline monitoring during 2010 to 2012 to enable the WIGs’ effectiveness to be evaluated in repeat surveys in 2019 (Bellamy et al. 2020).

Birds were surveyed using both territory mapping (nominally recording numbers of breeding pairs) and point counts (recording presence/activity of individuals). WIGs had significant effects on woodland structure, but not all in the predicted direction, e.g. conifers were reduced but understorey density was also reduced, suggesting that methodological differences may affected the before-after comparison (Bellamy et al. 2020). There were no clear results for effects of individual management interventions (thinning, ride management, restructuring, ride widening and creation of dead trees) on changes in bird numbers, but the study design was focused on the WIG scheme as a whole, so the various influences were confounded to some extent. Overall, however, there was a significant or near significant positive effect of grant status on the target species with the data from both bird survey methods. From point counts, there was an increase in abundance on WIG sites but decreased on non-WIG sites. From territory mapping results, there was a smaller decline on WIG sites than on non-WIG sites (Bellamy et al. 2020). These results were somewhat heterogeneous at the level of individual species, partly reflecting differences in absolute abundance and sensitivity to the survey method. The point count data will more sensitive to variations in bird activity and use by passing individuals, whereas the territory data should be more stable, but will have provided less power and variation with which to detect small effects.

Overall, this study shows broadly positive effects of WIGs on target woodland birds, but also illustrates some of the challenges with detecting impacts with periodic monitoring of stochastically variable species abundances and potentially mismatched scales of monitoring, management and species’ habitat use. The evidence provided for bird responses over 7-9 years is fairly strong, but that for specific mechanisms underlying the effects is only weak.

7.10 Continuous Cover Forestry (CCF)

Fuller et al. (2014) included a specific study of upland conifer plantations with a sitka spruce component in Perthshire, Argyll, Borders and North Wales, quantifying differences in species richness and abundance of breeding birds under Continuous Cover Forestry (CCF), and large scale clearfelling and restocking. Ranking the forest types in descending order of species richness gave: CCF with shrub understorey>CCF without shrubs>young pre-thicket clearfell>mature clearfell. Many ‘forest birds’ were most abundant, or recorded only, within CCF (e.g. Willow Tit, Wren, Wood Warbler, Blackcap, Wren, Redstart and Hawfinch). A small number of ‘young-growth’ species were most abundant in pre-thicket. The review of woodland management in the same report found that CCF tends to favour bird species associated with closed canopy woodland. These patterns support the value of CCF for biodiversity as an option for forestry, and that this value is greater than conventional forestry practice.

8. MAMMALS

8.1 Habitat needs of Section 7 Priority Species

Many of the Section 7 priority mammal species in Wales use woodlands as a habitat. For some species habitat use is entirely constrained to woodland. For example, red squirrel feeds primarily on seeds and nuts of woodland trees and shrubs, lives arboreally, and nests and breeds in tree canopies. Mature woodlands are the preferred habitat (Gurnell and Pepper 1991). Dormice are similar in breeding and resting requirements but hibernate on the woodland floor and will make use of a wider range of foods (seeds, berries, buds) (Juškaitis 2007). Whilst utilizing woodland, it is the scrub structures which are the important habitat for dormice, this includes hedgerows and areas of scrub outside of woodlands (Bright and Morris 1991). Woodland is the primary habitat for the pine marten, and also for the four woodland specialist bats: barbastelle, Bechstein's, noctule and brown long-eared bat (Anon 2005). These bat species use trees as roost sites and woodlands as foraging sites (Mitchell-Jones 2004). Similarly, pine martens use tree cavities as nesting and resting sites but hunt for their main food source, field voles in grassy opening in the woodlands (Coope 2007; Caryl et al. 2012a; Caryl et al. 2012b).

Woodlands and woodland edges are primarily used foraging by the greater and lesser horseshoe bats, and by common and soprano pipistrelle although the latter will use trees as roost sites (Mitchell-Jones 2004). It is also the edges of woodlands which are used by polecats and hedgehogs, species with more generalist habitat requirements (https://www.vwt.org.uk/wp-content/uploads/2015/04/polecat-leaflet.pdf). Of the riparian mammals, water vole and otter, it is otter which makes use of woodland structures. Shrub and regeneration close to watercourses are prime areas for breeding for otters and deadwood features provide resting sites (Chanin 2003). Water vole habitat is grassland and wetlands, these types of habitat are often left unplanted and open within woodlands and in these situations often provide good habitat. Two further priority species mammals, harvest mouse and brown hare are primarily open habitat, grassland species.

8.2 Management of existing woodlands

There is good evidence for effects of woodland management on mammals in temperate and boreal forests, with 71 studies covering 8 different woodland interventions documented by Bernes et al. 2015. Summaries of this knowledge are not numerous, however. Overall, the impact of different forest disturbances on the abundance of small mammals (i.e. positive or negative) appears to be species-specific (Zwolak 2009; Bogdziewicz and Zwolak 2014) and the reduction in abundance of mammal species associated with mature woodlands (e.g. snag and cavity dependent species) commensurate with the intensity of management applied (Vanderwel et al. 2009).

For the Welsh priority species mammals, maintenance of mature and ancient, semi-natural woodland (ASNW) features including veteran trees is important e.g. plentiful seed supply from mature trees for red squirrel (Bryce 2005); veteran and ancient trees providing deadwood for woodland specialist bats (Carr et al. 2020); complexity of understorey structures for pine marten (Caryl et al. 2012b). Minimum intervention is therefore preferable for the conservation of these species as thinning often removes older growth features. However, thinning can have positive effects for priority woodland mammals such as increasing the bat species richness and activity by creating suitable habitat for commoner bats (e.g. common pipistrelle) (Carr et al. 2020). Providing younger growth stages (regeneration and shrub) through removal of the overstorey has been shown to be of critical importance to dormice (Goodwin et al. 2018) and of significant use to pine marten in more fragmented habitats (Caryl et al. 2012b)

8.3 Creation of new woodland

It would be anticipated that woodland expansion will directly affect many aspects of habitat quality for woodland mammals. Of the few studies which have investigated such broad scale land use changes, none to small positive benefits for woodland mammals have been indicated as a result of woodland expansion. Despite differences in mammal diversity not being detected between plantations and other habitats (Stephens & Wagner 2006), greater mammal abundance (but not species richness) was detected by Felton et al. (2010) in plantations when compared to land purely composed of pasture, and by Moore et al. (2003), for small mammals in newly planted woodlands on farmland compared to both hedgerows and agricultural land. However, no clear positive effects on woodland mammals were detected when woodland expansion was followed over eight years (Lindenmayer et al. 2008).

8.4 Woodland size and fragmentation

Larger woodlands are considered to be better particularly for large mammals which are area sensitive and occur at low densities (Volenec and Dobson 2020) and a relationship of increased abundance of mammals with increased areas of small reserves and fragment has been reported (Lawrence et al. 2018). Otter's aquatic ranges are reported as c.17 ha, and in Wales pine marten require home ranges of 20km^2 and even red squirrel require 6 ha (Neill et al. 2009; McNicol 2017; Bryce 2005), with a target of 200 ha indicated for red squirrel reserves.[4]

Woodland fragmentation effects have been shown to be significant for mammals e.g. dormice and yellow- necked mice (Bailey 2007). Strategic woodland planting can increase connectivity at a landscape scale, which can aid species dispersal (Humphrey et al. 2015). For example, afforestation with conifers in Britain was shown to facilitate dispersal and connections between previously isolated forest fragments for red squirrel (Hale et al. 2001). However, with woodland connectivity comes risk of invasion by unwanted species which may threaten biodiversity. Grey squirrel is one such woodland species which is estimated to be able to colonise new habitat at a rate of 5.7–8.2 km/year and can lead to the loss of red squirrels through transfer of the pox virus (Welsh Government 2018). Woodland expansion can create opportunities for grey squirrel invasion not only through extending existing woodland patches but by placing woodlands in contact via other habitats capable of supporting

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[4] https://www.forestresearch.gov.uk/documents/2431/ukrsg_prioroity_site.pdf [sic]

grey squirrel movement (Stevenson et al. 2013). It is already recognized that to sustain the red squirrel population in Wales, grey squirrel control is needed within red squirrel woodlands and in the buffer areas surrounding the three core, red squirrel areas: Anglesey and adjacent areas of Gwynedd, Clocaenog Forest, and networks of conifer forest in the Tywi area in mid-Wales (Welsh Government 2018). Designing woodland expansion which provides more habitat for red squirrel but which does not also enhance the spread of grey squirrel, is challenging (Shuttleworth 2012). Like red squirrel, the decline of water vole in the UK has been attributed to the spread of another mammal with similar habitat requirements, the American mink. However, in the case of water voles, decreasing the fragmentation of its habitat is thought to help make populations more robust to the impacts of mink predation (Rushworth et al. 2000).

8.5 Edge effects

Reducing fragmentation and creating more woodland in the landscape may results in greater length of woodland edge and enhanced edge effects. Much research has focused on open habitat bird species for which, generally, edge effects are negative (e.g. Lamb et al. 2016; Wilson et al. 2014) However for mammals, edge effects can be both beneficial and detrimental. In Wales, polecat and pine marten and possibly harvest mouse may benefit as studies have found the abundance of mammalian predators and small mammals (the prey) consistently increased in habitat edges (Salek et al. 2010). Edge effects can be detrimental for open habitat species e.g. hare, (Hummel et al. 2017) but also for woodland mammals, particularly carnivores, where land use change e.g. woodland expansion, brings woodland edges into closer contact with humans (Woodroffe and Ginsberg 1998; Kowalczyk et al. 2015).

9. SUMMARY OF ANNEX-1

Woodland expansion can have positive and negative effects on biodiversity, depending on a number of variables explored in this section, including location, habitat type and condition, intended forest type and management. To mitigate disbenefits from woodland expansion site-based evaluations are necessary, careful forest design planning and tailored management of new woodland sites. Expert value judgements may be required to establish which elements of biodiversity and ecosystem services are prioritised at both local and national scale. However these local judgements must sit within in a strategic landscape, regional and national framework to ensure all habitats are conserved.

Existing policy protecting priority species must be respected and going forward these may need to reviewed due to the ongoing challenge of climate change. Consideration needs to be given to how to balance these species with many common species which are important in the delivery of beneficial ecosystem services and some current exclusions may be overly restrictive.

A general consideration is that all woodland habitat types, whether created by specific interventions or not, will have associated biodiversity and the choice between them is largely a policy decision. Once targets are chosen, such as red-listed or habitat-specialist species, management can be targeted effectively, but there are no absolute of ‘good’ or ‘bad’ for birds, or any other taxa.

Good evidence relating to woodland creation is available for:

The size of an individual woodland has an influence on the biodiversity it contains and its potential resilience. Small woodlands support edge species but may not provide sufficient conditions for woodland interior specialists, due to light levels, humidity, and foraging area. There is a well-established species-area relationship (Connor and McCoy 1979). This relationship is more important for smaller woodlands, with increases in area having a greater effect than increasing the area of larger woodlands. The definition of a ‘small’ woodland also depends on the focal species being considered. Creating woodland that extends and buffer areas where there is already an ancient semi natural woodland component, where possible using site-native species, provides the highest biodiversity gains.

Creating large woodland areas provides more internal woodland conditions which benefits woodland interior specialists and provides benefit through buffering from the effects of adjacent landuse. Larger woodlands also allow for heterogeneity e.g. open glades, incorporate other semi-natural habitats and a variety of topographic and water features, which has high biodiversity benefits and supports resilience.

There may also be a point where more woodland habitat does not lead to greater numbers and diversity and increasing woodland size has diminishing return. This will be different across taxa and species.

The shape of the woodland is also important, with longer thinner forests supporting lower levels of biodiversity than more compact or circular. This is dependent of on the relative abundance of edge specialist versus woodland interior specialist species in the regional species pool.

Maximising the area of woodland habitat available to species in the landscape by connecting fragmented woodlands, trees and hedgerows, allows species to migrate and supports ecological resilience. Reduction in woodland fragmentation is generally beneficial to woodland mammals but managing the increased dispersal of grey squirrel into red squirrel areas requires careful management. However, there are concerns that pest and diseases could spread through green corridors, although, this depends on the dispersal mechanism of the organism.

If woodland creation can only be of small areas of woodland, these should be focused in landscapes which are already relatively well wooded, although creation of woodland elsewhere, e.g. intensive agricultural landscapes, can have benefits e.g. for pollinators and birds.

Woodland creation can have strong positive effects on biodiversity, although time lags can be extreme. Generalist species have less specific habitat requirements and are expected to respond first. Specialists may take longer to colonize an area e.g. woodland-associated birds and flora. Temporal lags in species response could mask the ability to observe progress towards conservation success. To take account of this, a shift away from measuring ‘total species’ as a measure of success, towards checking for more detailed milestones (e.g. arrival of generalists, successful breeding of generalists, arrival of specialists, self-sustaining populations of specialists) is recommended.

Deciding where to locate new woodland will depend on several factors, not only where woodland may establish most successfully, but on the biodiversity value of the underlying habitat to be converted. There will need to be a consideration of trade-offs to facilitate ‘net biodiversity gain’. Quantitative evidence of trade-offs may not be readily available and requires long-term, landscape scale monitoring or experiments.

There is some evidence of the positive effects of woodland creation on mammals in the short term but this relates to abundance of individuals rather than the number of species (richness). Larger woodlands are considered better for certain mammals. Reduction in woodland fragmentation is generally beneficial although reduction in edge habitat may bring benefits and disbenefits depending on species considered. Increasing connectivity of woodlands in red squirrel stronghold areas requires careful planning and management.

There is good evidence for the benefits of woodland management:

Appropriate management can increase the biodiversity of existing woodland, in particular through generating open space, and increasing structural, species and genetic biodiversity.

Fencing to control levels of deer can support natural regeneration, where this is an aim, and enhance the shrub layer (understorey) of woodlands and as such have positive effects on a variety of woodland specialist species. Likewise, squirrel control can be essential for growing high-quality broadleaf woodlands which achieve the highest profits, however if this is not carried out on a large enough scale the costs can be prohibitive.

Maintenance of woodland rides, open spaces and structural diversity is critical for pollinators, flora, and all taxa of woodland edge specialists and those that need open space for part of their lifecycle. Increasing woodland edges can have disbenefits for woodland interior specialists including some mammals, where the effects can be both beneficial and detrimental depending on their trophic level and whether they are open habitat or woodland species.

The restoration of plantations on ancient woodland sites (PAWs) on ancient woodland sites presents a fundamental opportunity to increase biodiversity. Many ecological features remain on PAWs sites and they can recover with restoration, even as the plantation reaches maturity. The approach to restoration is important, with gradual opening of the canopy and change essential to conservation and preventing further damage and biodiversity loss.

Forest management that mimics natural disturbances (close-to-nature and combined objective forestry) delivers greater biodiversity benefits through diversifying species and age classes of even-aged stands. Continuous Cover Forestry (CCF), where suitable, reduces many of the negatives associated with clear fell management, although increased management frequency can also have negative impacts to recreation and wildlife

Good evidence for woodlands and biodiversity is available for:

The quality of woodland habitat may be even more important than size for biodiversity. Higher biodiversity value is associated with native and ancient semi-natural woodland often because of the longevity of such habitats and development of a complexity of structures and microhabitats often required by specialist woodland species, but conifer woodlands can contribute positively to biodiversity, especially diverse CCF where it offers heterogeneity and diversity of habitat features.

Mature forests and veteran tree species support higher levels of biodiversity than younger stands. Support may be needed to preserve mature and ‘over-mature’ trees to allow them to reach veteran status. Ecological succession from mature trees near the end of their life to younger trees which also support the same habitat can be supported by management.

In many situations land-use legacy, small woodland size, long-distances to source populations, poor dispersal and unfavourable management within and outside existing woodlands pose colonisation and management challenges, therefore we need to manage expectations for the timescales and resources required to achieve woodland restoration and creation objectives.

Woodland when compared to other habitats supports a greater abundance of individual species rather than a greater richness of species. Mammal abundance increases over time with woodland expansion and larger woodlands are considered better for certain mammals.

Given the wide range of the evidence, the importance of not translating these generalisations of the importance of extent, condition and connectivity into a series of one-size-fits-all interventions is clear. Matching interventions to place-specific opportunities and constraints seems vital, especially if there is a shift in emphasis toward payment for environmental outcomes (e.g. Sidemo-Holm, et al. 2018).

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Beauchamp, K., Jenkins, T.A.R., Alison, J., Bathgate, S., Bell, C., Braban, C., Broome, A., Bursnell, M., Burton, V., Dickie, I., Doick, K.J., Evans, C.D., Fitch, A., Griffiths, R., Hall, C., Healey, J.R., Jones, L., Keith, A.M., Kerr, G., Kuyer, J., Maskell, L.C., Matthews, R.W., Morison, J., Nicoll, B., Nisbet, T., O'Brien, L., Old, G.H., Pagella, T., Perks, M.P, Robinson, D.A., Saraev, V., Smart, S.M., Smith, A.R., Siriwardena, G.M., Swetnam, R., Thomas, A.R.C., Tye, A., Valatin, G., Warren-Thomas, E.M., Wong, J. & Emmett, B.A. (2020). Environment and Rural Affairs Monitoring & Modelling Programme (ERAMMP). ERAMMP Report-32: National Forest in Wales - Evidence Review. Report to Welsh Government (Contract C210/2016/2017)(UK Centre for Ecology & Hydrology Project 06297)

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ERAMMP Programme Office
UKCEH Bangor
Environment Centre Wales
Deiniol Road
Bangor, Gwynedd
LL57 2UW
+ 44 (0)1248 374500
erammp@ceh.ac.uk

www.erammp.cymru

www.erammp.wales