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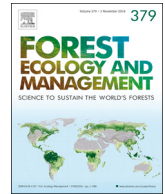
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Reviewing the evidence base for the effects of woodland expansion on biodiversity and ecosystem services in the United Kingdom



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ABSTRACT

Woodland expansion is internationally accepted as a strategy to reduce biodiversity loss, climate change and flood risk, but there has been limited assessment of the evidence for these benefits. In addition, despite UK targets for woodland creation, planting levels remain low. Furthermore, following the UK's decision to leave the European Union, a number of important decisions will be made about support mechanisms for different land uses, making evidence for the effect of woodland creation in different contexts essential. Following established guidelines, we undertook a systematic review of 160 articles to assess the evidence base for the effects of woodland expansion on ecosystem services in a UK context. We aimed to characterise the evidence base on the effects of such woodland expansion on biodiversity and ES and highlight where further research might be required. We found that the evidence base is dominated by research studying conifer plantations, and outcomes relating to biodiversity and regulating ecosystem services. By contrast, evidence for the effect of afforestation on multiple ecosystem services, cultural, and provisioning services is severely lacking. We argue that this weighting of evidence towards 'public goods' may contribute to the observed lack of planting, and that evidence for more tangible effects of woodland creation in relation other land uses is lacking. Implementation of woodland expansion could benefit from developing new incentives for planting woodlands, based on 'public money for public goods'. Future research should focus on the evidence gaps identified here, making use of context-specific, transdisciplinary, participatory methodologies which take into account plural values held in relation to the landscape.

1. Introduction

Woodland cover in the United Kingdom (UK) is currently 13% (Forestry Commission, 2017), less than half the European and global averages of 37% and 30% respectively (FAO, 2015). This relative lack of woodland is attributed to a complex history of exploitation by society and natural climate changes throughout the Holocene (Holl and Smith, 2007; Tipping et al., 2008). Since the end of WW1, woodland cover in the UK increased via the expansion of the public Forestry Commission estate, and a succession of grant schemes supporting private woodland planting. This increase was characterised by an initial dominance of conifer investment forestry planted mainly for timber, shifting towards increasing emphasis on broadleaved woodlands for multiple, predominantly environmental and recreational, purposes (Wong et al., 2015). Woodland types favoured for woodland expansion vary geographically across the UK. Welsh and Scottish forests are predominantly coniferous, while almost 90% of private and other woodlands in England comprises broadleaved species (Wong et al., 2015). In the UK and elsewhere, aims for afforestation are complicated by the fact that forest planting takes place on different lands, owned by different people with a diverse range of objectives and values (Thomas et al., 2015; Burton

et al., 2018).

Globally, forest loss and degradation have led to dramatic losses of biodiversity, carbon stores and ecosystem services (ES) (Ciccarese, Mattsson and Pettenella, 2012). As a result, numerous national and international policies aim for afforestation and reforestation, including the European Biodiversity Strategy for 2020 and the Water Framework Directive. Afforestation is an important strategy for climate change mitigation under the Paris Agreement, as well as wider targets for ecological restoration including the Aichi Targets, Bonn Challenge, and New York Declaration on Forests. In the UK, these high-level goals are incorporated into specific targets for woodland creation (Forestry Commission, 2009; Forestry Commission Wales, 2009; DEFRA, 2013; Scottish Government, 2017). It is internationally recognised that progress towards achieving these goals is uncertain, and that multiple challenges and barriers remain (Chazdon et al., 2017). Woodland expansion aims sit within wider land use challenges, including the need to ensure sufficiency and security of food supplies and the desire for ES multifunctionality, and there remains a lack of synergies between policies in these areas. Thus, at the UK level, there is a consistent gap between policy aspirations and actual levels of woodland planting, with reports showing that woodland planting in 2016 was at its lowest level

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for more than five years (Priestley and Sutherland, 2016). These persistent challenges are reflected in the identification of woodland expansion, in particular in relation to climate change mitigation, as a ‘wicked problem’ beset by conflicting goals, values and perspectives (Rittel and Webber, 1973; Shindler and Cramer, 1999; Duckett et al., 2016). This is due to the difficulty of implementing woodland expansion in the face of conflicting food and climate change policy goals, low acceptability of woodland planting among farmers, volatile stakeholder perceptions, and, in Scotland, grazing pressure from high deer populations (Duckett et al., 2016; Environment Climate Change and Land Reform Committee, 2017; Burton et al., 2018).

A review of evidence in a Scottish context has suggested that we need a clearer articulation of woodland benefits, improved evaluation of woodland creation schemes, and improved understanding of trade-offs with other land uses (Thomas et al., 2015). Research has also shown that there is a lack of synergies between the many policies and plans promoting woodland networks and corridors (Muñoz-Rojas et al., 2015). More generally, further research is required to elucidate the relationships between different (forest and non-forest) land use ES provision levels, whether these be synergies, trade-offs, or groupings in multifunctional ‘bundles’ (Cord et al., 2017). Therefore, research needs to make clear the effect of woodland expansion in different contexts, in order to provide robust, context-specific evidence. This is especially pertinent given the urgency of initiatives concerned with carbon sequestration and biodiversity protection, and the risk of rapid, poorly-informed actions leading to suboptimal or counterproductive outcomes. Assessment of the extent of current knowledge about the effects of woodland expansion in the UK is therefore necessary not only for national-level policy making but also as a case study of internationally-relevant challenges in land system planning and management.

We undertook a systematic review to assess the evidence base for the effects of woodland expansion on biodiversity and ES in the UK. The review had two main objectives: (1) To systematically collate and synthesise both academic and grey literature studying woodland

expansion in a UK context; and (2) To characterise the evidence base on the effects of such woodland expansion on biodiversity and ES and highlight where further research might be required. To achieve these objectives, we aimed to answer three research questions: (1) What knowledge do we currently have about the effects of woodland expansion on biodiversity and ES provision?; (2) What are the main gaps in this knowledge?; and therefore (3) What does this mean for developing strategies for woodland expansion that maximise biodiversity and ES provision? The first two questions directly address results from the review, while the third forms the basis for the discussion.

1.1. A note on the terms and definitions used

Many different terms are used to describe different approaches to woodland creation, and this can result in some confusion or misconceptions (Mansourian, Vallauri and Dudley, 2005). ‘Woodland expansion’ is used throughout this paper to encompass afforestation, reforestation, woodland creation, and forest landscape restoration. Afforestation and reforestation both involve the artificial establishment of trees, in the former case where no trees existed before (for at least 50 years), while the latter refers to planting or seeding on land that was recently forested but that has been converted to non-forested land (Mansourian, Vallauri and Dudley, 2005). Natural regeneration is considered a process of woodland expansion and can either be assisted (Chazdon, 2008) or can occur unaided if there are seed sources and browsing pressure is low (Forestry Commission, 2009). Ecological restoration is defined as ‘the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed’ (SER, 2004). Increasingly, Forest Landscape Restoration (FLR) is advocated on the basis that restoration has to address multiple and sometimes competing needs. It is defined as ‘a planned process that aims to regain ecological functionality and enhance human well-being across deforested or degraded landscapes’ (Lamb, 2014). Although every effort is made to use woodland expansion as a catch-all, these terms are occasionally used

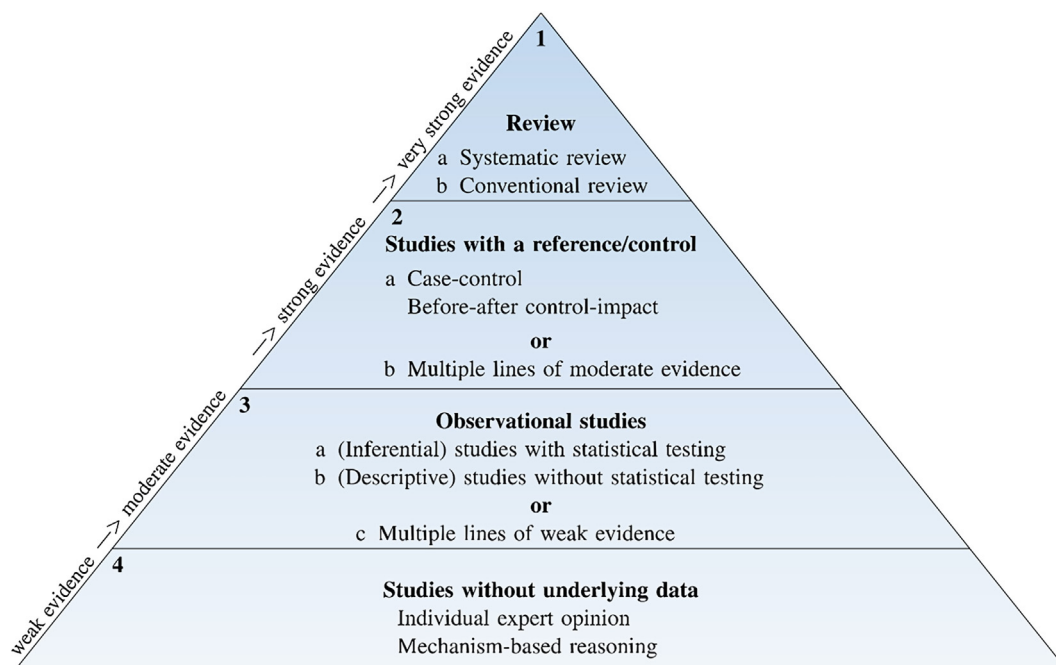


Fig. 1. The Level of Evidence (LoE) hierarchy ranking as produced by Mupepele et al. (2016). The pyramid ranks study designs according to the strength of the evidence they produce. Within this hierarchy, systematic reviews (LoE1a) are regarded as providing the highest evidence level, with studies based on mechanistic reasoning providing the lowest. In this case, the ‘strong evidence’ level 2a included either studies that took measurements prior to afforestation as well as during or after, or studies which considered a different (usually open) habitat in comparison to an afforested or woodland habitat. Modelling studies require a slightly different interpretation in terms of their LoE. Models which represent theories, without underlying data, are given the lowest level (LoE4). If they have data input to determine their parameters they achieve LoE3b, and if this data is tested and validated then this increases to LoE3a. To reach LoE2b or LoE2a, modelling studies must confirm their predictions on several unrelated datasets, or be built on data from controlled studies.

interchangeably.

2. Methods

We undertook a systematic review (SR) of both academic literature and unpublished ‘grey’ literature, following established guidelines (Collaboration for Environmental Evidence, 2013).

2.1. Data collection

Online searches were carried out on electronic databases, organisational websites and internet meta-search engines. Search terms were developed around the Population, Intervention, Comparator, Outcome (PICO) framework (Collaboration for Environmental Evidence, 2013). As ‘ecosystem services’ is a relatively new term, keywords that related to each ES category were included, to capture all relevant research prior to and since the emergence of ES research. A full list of search terms used for all databases and websites can be found in Supplementary Material 1. The academic electronic databases Web of Science and Scopus were searched, and the first 50 Microsoft Word document and PDF hits from Google Scholar and Google were examined for appropriate literature. Several organisational websites were also searched for relevant information. All literature returned by the searches underwent a three-stage filtering process, using pre-defined inclusion and exclusion criteria. All articles were initially filtered by title and then abstract. Following the abstract filter, full texts were assessed and either accepted or rejected from the final review. The SR identified studies conducted in the England, Scotland, Wales, Northern Ireland and Ireland, as well as the UK as a whole. Duplicates were removed. Some documents (26 in total) could not be accessed due to restrictions, or were book chapters that could not be sourced online or in available libraries, and these were also excluded. The entire filtering process was carried out by one reviewer. However, all progress and decisions made were discussed regularly with co-authors and the entire inclusion/exclusion process was recorded in a spreadsheet for transparency (Supplementary Material 1).

Supplementary data associated with this article can be found, in the online version, at <https://doi.org/10.1016/j.foreco.2018.08.003>.

2.2. Data analysis

The final set of articles underwent an iterative process of characterisation, data extraction and critical appraisal in terms of study quality. We categorised each study by a range of attributes including location, ES category, unit of study, woodland type, comparator/control habitat, and outcome measure. Papers were assigned to an ES category based on the Common International Classification of Ecosystem Services (CICES). Biodiversity was considered as a separate category. The quality of study design was further assessed using a tool developed to assess the strength of evidence of ES and conservation studies (Mupepele et al., 2016). This tool (Fig. 1) uses a hierarchy of evidence to rank studies based on their experimental design, and thus lent itself well to assessing a wide range of studies with different units of study and types of data. The entire study database, with references, can be accessed in Supplementary Material 2. The scope of the review was intentionally broad throughout, in order to show the spread (and any bias) of evidence available, resulting in a diverse set of studies with very different methodological designs and resulting types of data. This precluded the use of formal quantitative techniques or meta-analysis, and therefore we focus on a narrative synthesis of the evidence.

3. Results

3.1. Summarising the evidence base

An initial search was carried out on the 2nd February 2016. The

Table 1

A summary of the number of papers returned from the initial searches, and at each subsequent filtering stage.

Systematic review stage	No. of papers
Studies captured using search terms in electronic databases (including duplicates)	1552
Studies captured using search terms in electronic databases (excluding duplicates)	1347
Studies remaining after title and abstract filter	474
Studies remaining after full text filter	160

search was updated on the 7th April 2017 using the main databases (Web of Science and Scopus) from the first search. Table 1 shows the number of papers returned at each stage in the SR process.

Most evidence relates to the regulating and maintenance ES category (82 studies) and biodiversity (54 studies) (Fig. 2). Within both categories, there is a bias towards studies of conifer plantations (74 studies). There is very little evidence relating solely to provisioning (1 study) and cultural categories (2 studies). However, it is important to note that some studies which consider multiple topics include aspects which fit in those categories (21 studies). Given the weighting of the evidence, we focus on summarising findings from papers within the biodiversity and regulating and maintenance categories, before going on to assess evidence for ecosystem disservices, and papers which consider multiple ES. The guidelines followed by the review emphasise that SR methodologies should collate and synthesise data without adding interpretation (Collaboration for Environmental Evidence, 2013).

3.2. Biodiversity

Most studies of biodiversity focus on birds (22 studies), invertebrates (16 studies) and ground flora (5 studies), highlighting evidence gaps for the effects of new woodland on other taxa (e.g. mammals). For studies of conifer plantations (30/54 studies), several broad findings based on the evidence can be identified. In the early stages of Sitka spruce plantations studies have recorded a shift in ground vegetation from small stature herbs to more competitive grasses as plantations grow (Oxbrough et al., 2006; Buscardo et al., 2008). The shading effect of dense non-native coniferous plantations at later stages has been found to significantly reduce ground vegetation cover, but this is noted to reflect the density of the tree canopy more than the species planted (Wallace et al., 1992). A more diverse woodland flora has been found to develop in some conifer stands over time particularly where lower stocking densities allow a greater amount of light through to the forest floor (Essex and Williams, 1992; Wallace et al., 1992; Wallace and Good, 1995). A comparison of plantations with different species mixes and stand ages across Ireland illustrates differing effects on vegetation communities (French et al., 2008). In line with previously outlined findings, closed canopy sites dominated by Sitka spruce (*Picea sitchensis*) had very low numbers of ground flora, and were found to be striking in their lack of diversity. By contrast, Japanese larch (*Larix kaempferi*) stands at all ages supported an abundant and species rich community of bryophytes, but these were mainly fast-colonising generalists, essentially meaning that the vegetation community beneath larch represented ‘moorland-with-trees’ (a function of the previous land use) as opposed to a true woodland flora. A mature Ash (*Fraxinus excelsior*) stand was found to support a vegetation community closest to native woodland, aided by a location on base rich free-draining soils and proximity to mature native woodland. This aligns with the assertion that it is likely that the biodiversity of newly established stands depends on the availability and colonisation ability of native woodland species (Thomas et al., 2015)

There is widespread evidence for carabid species (*Carabidae*) turnover following the establishment of conifer plantations, with open

Category	Metric/Service	Level of Evidence (LoE)			
		Very Strong	Strong	Moderate	Weak
Biodiversity	Community structure		(2) (1)		
	Connectivity/Species movement			(5) (1) (2)	
	Functional diversity		(1)		
	Species abundance		(1) (6) (1)	(4) (5)	
	Species abundance and breeding success		(2)	(4) (1)	
	Species abundance and diversity		(9) (2)	(1) (2) (3)	
	Species density		(1)		
Regulating and Maintenance	Regulation of chemical composition of atmosphere and oceans	(2)	(1) (5) (3) (4)	(2) (2) (5)	
	Regulation of temperature and humidity, incl. ventilation and transpiration		(1)	(1)	
	Control of erosion rates		(1)		
	Hydrological and water-flow regulation (including flood control and coastal protection)	(1) (2)	(1) (6)	(2) (1)	
	Decomposition and fixing processes and their effect on soil quality		(1) (4)	(1)	
	Weathering processes and their effect on soil quality	(1) (1)	(4)	(1)	
	Regulation of the chemical condition of freshwaters by living processes	(2) (2) (1)	(2) (5) (1)	(1) (9)	
	Bio-remediation by micro-organisms, algae, plants and animals		(1)		
Provisioning	Biomass			(1)	
Cultural	Characteristics of living systems that are resonant in terms of culture or heritage			(1)	
	Characteristics of living systems that enable aesthetic experiences			(1)	
Multiple		(7) (1)	(2)	(3) (7) (1)	

Key	Conifer plantation		Native woodland		Agroforestry	
		Broadleaf plantation		Riparian		Multiple

Fig. 2. The biodiversity metrics and CICES ecosystem service categories considered by the 160 papers. Each paper is represented by a symbol relating to the type(s) of woodland it studies, and is assigned a Level of Evidence (LoE) based on its study design (see Fig. 1).

ground species becoming less common, and generalist and forest specialists becoming more so (Day and Carthy, 1988; Buse and Good, 1993; Lin, James and Dolman, 2007; Karen et al., 2008). There are differing conclusions as to whether this turnover significantly changes overall

carabid diversity compared to previous or comparative land uses. Day & Carthy (1988) found that although species richness and alpha diversity were lower in forested compared to moorland plots, this was not statistically significant. By contrast, Buse & Good (1993) found the

greatest abundance, species richness and diversity occurred in non-afforested sites. Nonetheless, there is strong agreement between studies that fostering and maintaining diversity in forest structure and species is essential for maintaining overall carabid diversity, with rides, clear-fell areas and early successional habitats allowing open ground specialists to continue to thrive (Buse and Good, 1993; Lin, James and Dolman, 2007; Karen et al., 2008; Spake et al., 2016). When taking into account all the stages of the forest cycle, as well as the effect of locality, it has been concluded that on the whole forests can be as species rich (in terms of carabids) as surrounding open habitats (Coll and Bolger, 2007). A recent study of carabid functional diversity finds that, as increasing canopy cover generally drives down functional diversity, management which emulates natural disturbance regimes through gap creation and close-to-nature forestry will be beneficial to carabids (Spake et al., 2016).

Spider and bird species also demonstrate turnover with afforestation. Pre-afforestation land use has an influence on both these communities, with evidence suggesting that improved grasslands are most likely to benefit from afforestation, showing increased species diversity and richness, whereas wet grasslands and peatlands are more sensitive due to their more distinct species assemblages (Oxbrough et al., 2006; Wilson et al., 2012). Peatlands, although low in overall bird diversity, are home to rarer habitat specialists, and are found to be most sensitive to afforestation (Wilson et al., 2012). There is strong evidence that edge effects generated by conifer plantations negatively affect a number of open ground bird specialists, particularly in upland contexts (Lavers and Haines-Young, 1997; Buchanan et al., 2003; Douglas et al., 2014; Graham et al., 2015). However, similar to findings from studies of carabids, diversity in stand age and forest structure can have positive effects, with most bird species abundances being positively related to the extent of shrub cover at the edge of plantations (Calladine, Bielinski and Shaw, 2013), and black grouse (*Tetrao tetrix* L.) abundance being positively associated with younger pre-thicket forest (Pearce-Higgins et al., 2007). One long-term study carried out over a 12-year period of coniferous afforestation on an upland grassland showed significant turnover of both vegetation and bird species, with a significant increase in the overall number of breeding bird pairs (Sykes, Lowe and Briggs, 1989).

Several studies consider the effect of coniferous afforestation within the wider landscape, and once again, the effects differ depending on the context and species considered. In lowland farmland, no detectable differences in total farmland bird species richness or abundance were found between farmland sites with very little forest cover and those approximately one-third afforested (Pithon, Moles and O'Halloran, 2005). By contrast, conifer plantations have been associated with declines in presence and breeding performance of both ravens (*Corvus corax*) (Marquiss, Newton and Ratcliffe, 1978) and golden eagles (*Aquila chrysaetos*) (Marquiss, Ratcliffe and Roxburgh, 1985; Watson, 1992; Whitfield et al., 2001). Similar studies of red kites (*Milvus milvus*) in Wales found no firm evidence for an effect of afforestation on species occurrence (Newton et al., 1981). Still, it is important to note that many of these studies are based on the effect of plantations planted in the 1970s, which were often densely planted and heavily dominated by non-native conifers. Modern plantations are encouraged to have more sympathetic designs, such as areas of open ground, riparian buffers and areas of broadleaf woodland, and thus may not have the same effect.

The effect of growth stage has been found to have a more significant effect of bird assemblages than species mix, with species associated with younger woodland stages being typical of open, un-forested habitats, which were turn strongly influenced by the pre-afforestation habitat (Wilson et al., 2006). In older stages, assemblages were characterised by woodland generalist species such as the Chaffinch (*Fringilla coelebs*), Coal tit (*Periparus ater*), Robin (*Erithacus rubecula*) and Wren (*Troglodytidae*). Other studies indicate that availability of young woodland is particularly important for several bird species. A national comparison of Tree pipit (*Anthus trivialis*) and Lesser redpoll (*Acanthis*

cabaret) abundance with changing woodland cover over time shows that abundance declines with a decrease in the availability of young woodland (Burgess et al., 2015). Studies of Black grouse (*Tetrao tetrix*) also indicate distributional shifts in populations, with population declines where pre-thicket forest matures, and establishment of new populations near to new native pinewoods (White, Warren and Baines, 2013). Populations were found to be greatest where new native woodland comprised approximately 30% of the land area and averaged 5 years old (Scridel, Groom and Douglas, 2017).

A greater number of studies have begun to consider biodiversity in relation to other types of woodland in recent years, but there is a lack of controlled, field-based evidence for the effect of native woodland expansion on biodiversity. Studies suggests that earthworm (*Lumbricus terrestris*) communities are larger under re-established native woodland than surrounding moorland (Butt and Lowe, 2004) whereas studies of moths in Woodland Grant Scheme sites found lower species abundance and richness compared to mature woodland (Fuentes-Montemayor et al., 2015). A collation of vegetation surveys from over 100 years of broadleaf woodland colonisation in two abandoned fields in lowland farmland also found substantial turnover of individual species, with total flora estimated to be richest just before canopy closure, but it is noted that the methodology is limited by a patchy record and different survey methods used throughout the study period (Harmer et al., 2001). Most evidence for native woodland is derived from reviews, landscape scale GIS or modelling methodologies, primarily focusing on the potential for increasing woodland connectivity to enhance biodiversity at the landscape scale. Reviews state that regenerating native woodland allows natural colonisation by plants and fungi, with anecdotal evidence stating that regenerating woodland in a Scottish national nature reserve supports species such as blaeberry (*Vaccinium myrtillus* L.) and bog myrtle (*Myrica gale* L.) that replace dominant grassland species (Bunce et al., 2014; Armstrong, 2015). Earlier GIS-based work focused on functional connectivity suggested that targeting new native woodland adjacent to ancient woodland patches increases core habitat area and functional network size, enabling faster colonisation of woodland species (Bailey, Lee and Thompson, 2006; Hope, Humphrey and Stone, 2006; Quine and Watts, 2009). In terms of the effect of planted conifer woodlands in facilitating species movement, wood ant populations have been found to make use of newly formed non-native plantations to expand, showing that these plantations can provide a suitable habitat (Procter et al., 2015). More recent modelling approaches such as circuit theory and individual based modelling suggest that using spatially targeted woodland creation to fill regional 'bottle-necks' has potential improve species expansion response to climate change (Hodgson et al., 2011), but that it is difficult to accommodate multiple species when targeting woodland creation (Synes et al., 2015). Some strategies, such as creating small woodland patches next to larger patches of existing woodland, can provide benefit to the widest range of species (Synes et al., 2015)

3.3. Regulating and maintenance

Studies relating to the regulation of water are also dominated by studies of conifer plantations. Previous reviews have concluded that afforestation can alleviate flooding via three main mechanisms: (1) greater water use compared to other land uses; (2) greater hydraulic roughness compared to more open habitats, having a slowing effect on flood flows; and (3) a soil 'sponge effect', with more organic matter and tree roots, and less soil disturbance, allowing woodland soils to hold more water (Nisbet and Thomas, 2006; Nisbet et al., 2011). There is a collection of strong evidence to suggest that conifer plantations have higher water use when compared to a variety of other land uses, and that this is associated with reductions in peak flows and reduced 'flashiness' in forested catchments (Hornung and Newson, 1986; Johnson, 1998; Heal et al., 2004; Nisbet et al., 2011). A single study looking at naturally regenerated Scots pine woodland concludes that it is likely to

have a similar magnitude of water use compared to non-native plantations (Haria and Price, 2000). Fewer studies explicitly consider the effect of broadleaf plantations or naturally regenerating native broadleaf woodland on flood flows, but two reviews conclude that although these types are expected to have slightly lower water use than coniferous woodland, they can still have a dampening effect on flood flows when compared to other more open land uses (Roberts and Rosier, 2005; Thomas and Nisbet, 2007). Riparian and floodplain woodland is found to be particularly effective at reducing peak flood magnitude (Broadmeadow and Nisbet, 2004; Dixon et al., 2016).

Scale is a key issue when thinking about the effect of woodland on flood control (Nisbet and Thomas, 2006; Nisbet et al., 2011). Given current evidence, the smaller the area of woodland in a catchment, the less the effect on reducing flood peak, and there continues to be little support for a significant effect on extreme flood flows at a wider landscape level (Nisbet et al., 2011). Nevertheless, there is evidence for a forest impact on flood flows at a local level, and for smaller flood events (Nisbet and Thomas, 2006), as well as recent studies suggesting that small areas of floodplain woodland in the upper and middle catchment can have a large effect on reducing peak flood magnitude (Dixon et al., 2016).

The largest number of studies of regulating and maintenance ES (29) relate to regulation of the chemical composition of the atmosphere through carbon sequestration and storage. Evidence for the effect of afforestation on soil organic carbon (SOC) is dominated by chronosequence studies of Sitka spruce (*Picea sitchensis*) plantations. A previous review of the effect on SOC of converting grassland to forestry found inconclusive evidence, citing inherent problems of soil heterogeneity and few relevant UK datasets as an issue (Reynolds, 2007). Overall, studies suggest an initial loss of SOC due to a combination of site-specific factors (e.g. site disturbance, drainage, higher root activity/respiration, thicker litter layer), followed by a recovery and/or increase with stand age, or by the second rotation (Byrne and Farrell, 2005; Zerva and Mencuccini, 2005; Saiz et al., 2006; Black et al., 2009). A long-term study of naturally regenerating native woodland found a significant increase in SOC (Powelson et al., 1998), whereas an Ireland-wide study found no significant change in SOC between afforested (either coniferous, mixed, or broadleaf) and non-forested (paired pre-afforestation habitat) sites (Wellock, LaPerle and Kiely, 2011). Taking into account C stored in the forest floor has a positive effect, with conifer stands in particular having significantly larger C stores than broadleaf or mixed stands (Wellock, LaPerle and Kiely, 2011).

Studies which also consider aboveground biomass in the growing trees (i.e. total ecosystem carbon) show significant increases in overall carbon, with woodlands becoming more significant overall sinks as stands age, (Wellock, LaPerle and Kiely, 2011; Peichl, Leava and Kiely, 2012). Recent analysis of silvopastoral systems suggest that they may be able to achieve a higher level of overall carbon storage than equivalent areas of either woodland or pasture (Beckert et al., 2015; Upson, Burgess and Morison, 2016). A small number of studies consider non-carbon GHG dynamics. The evidence is limited, site specific, and hard to generalise. The initial disturbance caused by conversion of a grassland to broadleaf planation was found to increase nitrous oxide emissions, but this effect decreased to approximately one third the previous grassland level of emission with increasing woodland age (Mishurov and Kiely, 2010). Elsewhere, conversion of grassland to Sitka spruce (*Picea sitchensis*) caused an increase in nitrous oxide emissions, but a decrease in methane emissions, whereas an Ash (*Fraxinus excelsior*) plantation have no clear effect on either GHG (Benanti et al., 2014). Another study considering three transitions (bog planted with pine (*Pinus*), grassland planted with pine, and birch (*Betula*) regenerating on moorland) found that afforestation resulted in a stable and consistent sink of methane in all cases (Nazaries et al., 2013).

In terms of modelling approaches, various national scale models predict that afforestation can sequester significant amounts of carbon, especially fast growing Sitka spruce (*Picea sitchensis*) (Nijnik et al.,

2013), but also broadleaf or native woodland (Perks et al., 2010; Sozanska-Stanton et al., 2016). In a comparison of UK climate mitigation actions, afforestation of acid grasslands with broadleaf woodland is predicted to sequester carbon at the highest rate compared to a range of other options (Sozanska-Stanton et al., 2016). The economic value of sequestration potential depends on choice of discount rate, yield class, and social value of carbon (Bateman and Lovett, 2000; Brainard, Bateman and Lovett, 2009). Scenario analysis suggests that increasing planting on lowland agricultural land would be more beneficial than the current trend for planting on low quality upland land, but this assumes a loss of carbon from initial site preparation and drainage (Brown and Castellazzi, 2014).

3.4. Cultural and provisioning ES

The single paper focusing purely on a provisioning service uses a mathematical model to estimate the effect of an agroforestry system on sheep yields and timber production. Results suggest that such a system can be financially viable, but that it is very sensitive to discount rate and timber prices (Doyle, Evans and Rossiter, 1986). Only two papers consider purely cultural ES, considering the impact of afforestation in rural case studies in Ireland (Dhubháin et al., 2009; Carroll et al., 2011). Perceptions of forestry and afforestation differed significantly between regions, with one case study with a longer history of forest cover showing positive values for amenity and recreation, whereas another had negative perceptions linked to the dominance of Sitka spruce, which was considered to have negligible amenity value (Dhubháin et al., 2009). Another case study comparison showed similarly mixed reactions to afforestation, with large, dense blocks of coniferous plantations exacerbating feelings of social isolation in one region, while in the other contributing to greater landscape diversity and feelings of inspiration (Carroll et al., 2011). A more locally nuanced approach to forest planting is suggested to achieve greater social acceptance of future afforestation (Carroll et al., 2011).

3.5. Disservices

Some evidence points towards the potential negative effects of afforestation, in particular relating to the regulation of the chemical condition of freshwaters. There is strong evidence to suggest that coniferous woodlands are more effective at scavenging acid pollutants compared to other land uses, and thus that they can have an acidification effect on soils and freshwater (Jenkins et al., 1990; Waters and Jenkins, 1992; Rees et al., 1995; Allen and Chapman, 2001). Whether broadleaf or native woodlands have the same effect is still the focus of investigation, with the only two field-based studies having contrasting, location specific results (Gagkas et al., 2011; Ryan et al., 2012). However, reduced atmospheric deposition, soil buffering capacity, and sustainable forest management initiatives such as the use of more diverse species mixes and riparian buffer strips are expected to reduce acidification (Ferrier, Whitehead and Miller, 1993; Curtis et al., 2014).

In addition, despite the potentially beneficial effect of flood attenuation, research has highlighted that there could also be a negative effect of afforestation on water yield during dry weather. There have been contrasting results relating to groundwater recharge but modelling of a range of potential afforestation scenarios (including both conifer and broadleaf woodland) has predicted a maximum decrease in groundwater levels of less than 0.3 m, concluding that an increase in woodland cover may not exacerbate water stress (Zhang and Hiscock, 2010). Allen & Chapman (2001) reviewed the effect of forest cover on groundwater resources and concluded that generally afforestation (again considering multiple types of woodland) can reduce groundwater yield through interception and transpiration, with both potential positive and negative effects dependent on site specific factors, including land cover, rainfall, infiltration, evapotranspiration and spatial distribution of the water table.

3.6. Relationships between ES

A small proportion of studies (21) consider more than one ES, and could not be assigned to just one service or metric (Fig. 2). Within these studies, there are varied combinations of topics considered, with study designs dominated by reviews, spatial analyses and mixed-method approaches. Overall, these findings highlight that there are a limited number of studies and no consistent method for assessing the effect of afforestation on more than one ES at a time. This is an important finding given the need to understand relationships between ES for effective policy making. Further developing this challenging topic is beyond the scope of this review, and is an important area for future research. This should build upon valuable work already carried out to assess relationships between ES (Lee and Lautenbach, 2016).

3.7. Study design

There has been an increase in the number of papers focusing on afforestation over time, with field-based methodologies dominating (86 studies), but there has been a diversification in the methodologies employed over time, with social studies in particular being a relatively recent occurrence (Fig. 3). Table 2 summarises the methodologies employed and assesses their applicability to ES research. Given the long term nature of woodland development, and the likelihood that any resulting ES provision will vary over time, there is an obvious lack of long-term monitoring studies, with literature reviews, modelling, and chronosequence studies which examine the effect of different stand ages on the species or ES in question, perhaps aiming to fill this gap.

4. Discussion

Given the research questions outlined in the introduction, we conclude by considering how the evidence base outlined in the results can be used and built upon to design strategies for afforestation which

maximise biodiversity and ES provision. First, the evidence base outlined by the review is compared and contrasted to wider international findings, and evidence gaps are highlighted. Following this, considering the evidence base outlined by the review, this final section aims to reflect upon how this base might be used to develop effective strategies for afforestation in the UK.

4.1. Strong evidence for public goods

At a global scale, reforestation has been identified as the land-based strategy with the greatest potential for climate mitigation (Griscom et al., 2017), and the UK level evidence reported here supports this. Despite this, the assumption that faster growing tree species (such as Sitka spruce (*Picea sitchensis* (Bong) Carr.) planted extensively in the UK) will be most beneficial to carbon sequestration has recently been challenged (Körner, 2017). These productive forests typically serve the timber industry, and thus any carbon sequestered has shorter residence times than slower growing species, with wood products only considerably contributing to carbon sequestration if their overall use rises (Körner, 2017). For the carbon pool to significantly change in the long term, the maintenance of slower growing old-growth stands, which also increase and protect soil organic carbon (SOC) stocks, is expected to be more beneficial (Körner, 2017; Schwartz et al., 2017). Although evidence for carbon sequestration in old-growth stands in the UK (as a function of woodland/land use history) is limited, international evidence clearly shows the importance of old-growth stands in this and other contexts (Luyssaert et al., 2008; Körner, 2017). However, in countries that are net importers of timber, such as the UK, it is important that strategies for climate mitigation consider the protection and maintenance of old-growth stands alongside efficient management of faster growing tree species to meet demands for wood products and prevent moving issues of carbon sequestration to other countries. Overall increases in the amount of carbon sequestered are likely to be as a function of expanding woodland cover, retaining old growth stands

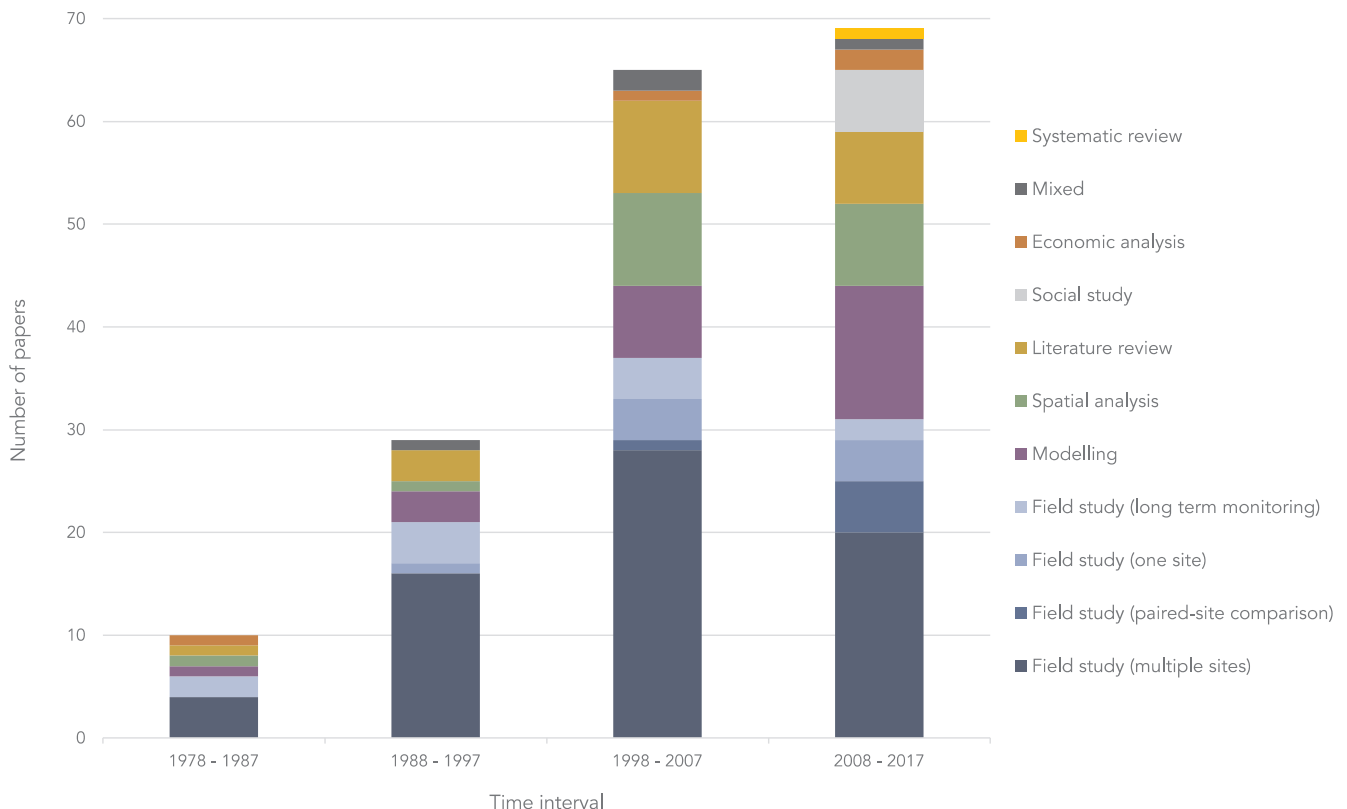


Fig. 3. Changes in the methodologies used to study afforestation over time.

Table 2
The strengths and weaknesses of the methodologies employed by the 160 studies, in relation to the ES categories they consider.

Study methodology	Regulating & maintenance				Total	Strengths and Weaknesses
	Biodiversity	Provisioning	Cultural	Multiple		
Field study					9	Site specific, difficult to generalise findings
One site	7				65	Consider effect across many sites with different settings e.g. pre-afforestation land use, soil type, tree species, stand age. Chronosequences attempt to tackle effects through time.
Multiple sites	29		3		12	The ideal type of study for long-term woodland development, but limited in number with very varied methodologies
Long term	5	1	1	1	23	Offer tools for exploring the effect of long-term land use changes and their effect. Address complex interactions. Ideally parameterised with data from field studies, but often lack testing/validation
Modelling	5				18	Consider effect across landscapes. Attempt to consider synergies, bundles, trade-offs
Spatial analysis	13		2		19	Collation of wide range of evidence
Review	3		7		1	Place economic value on services. Useful for cost-benefit analysis
Economic analysis	12				6	Consider values and preferences associated with afforestation, often across ES categories
Social study	1	2	4	4	4	Varied and inconsistent design. Some consider multifunctionality/multiple benefits
Mixed						

and greater use of timber products in buildings.

In terms of SOC, our results align well with other reviews. Both European (Bárcena et al., 2014) and global (Guo and Gifford, 2002; Laganière, Angers and Paré, 2010) meta-analyses found a strong influence of former land use, with afforestation having a more positive impact on SOC on cropland soils than pastures or natural grasslands. A more recent global meta-analysis taking into account sampling depth and carbon-nitrogen interactions found a significant increase in stocks on both cropland and pasture 30–50 years after afforestation, with stocks before this being either depleted or unchanged (Li, Niu and Luo, 2012). Given these findings, together with established UK guidelines which prevent afforestation on deep peat soils (a peat layer greater than 50 cm) (Forestry Commission, 1998), it seems likely that woodland expansion strategies can benefit soil carbon stocks.

The strong evidence we find for woodland expansion's contribution to water flow regulation and flood control aligns with international reviews showing reduced water yield with afforestation (Farley, Jobbágy and Jackson, 2005; van Dijk and Keenan, 2007). Afforestation is more likely to regulate local 'flash' floods than major events, and there are a wide variety of factors that are important to take into account when determining the overall impact on water resources, including tree species physiology, plantation design and management, the benchmark against which changes are assessed, and the water system (or catchment) configuration (scale of afforestation, timing of impacts, location in the catchment) (van Dijk and Keenan, 2007). The knowledge gap for the effect of native woodland is found elsewhere, with a study of the key drivers (e.g. species composition, tree canopy status) of natural forest water use highlighting gaps in current functional knowledge regarding water use by many forest tree species (Aranda et al., 2012). In terms of the potential disservice of acidification of freshwaters, the evidence presented here suggests that this effect is location specific, depending on the soil buffering capacity, tree species, and level of atmospheric acid deposition. Thus, acidification is not an inevitable consequence of afforestation (Hong et al., 2018), especially when taking into account guidelines in the European Water Framework directive to reduce acid pollution, and national guidelines to avoid planting in acid-sensitive catchments (Forestry Commission, 2014a).

4.2. Biodiversity

Forest 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 and Doncaster, 2017). The results presented here illustrate this, with a variety of methodologies used, and a lack of long-term monitoring of changes in biodiversity with afforestation. This review highlights the patchiness of evidence available for the effect of woodland expansion on biodiversity, with evidence limited to a small number of bioindicators (e.g. carabids, birds) and biased towards dominant woodland types in particular areas of the UK (e.g. non-native conifer plantations in Scotland). New experimental designs are aiming to address these challenges. For instance, a large-scale UK-based natural experiment utilising historic woodland creation sites (Watts et al., 2016) has shown that bird species responses to woodland creation depend on local and landscape-scale factors that interact across time and space (Whytock et al., 2017).

There is an obvious bias towards studies of the effect of conifer plantations on an also limited set of indicator species, with this being attributed to the history of afforestation in the UK referred to in the introduction. Globally there has been much debate about the implications of plantation forests for biodiversity (Brockerhoff et al., 2008), and it has been suggested that much of the literature reporting lower biodiversity in plantation forests has been based on inappropriate comparisons such as with natural or native forest rather than alternative human land uses (Stephens and Wagner, 2007). This is not the case for the UK evidence, with most comparisons being made to previous or adjacent land use (e.g. either grassland or agriculture). The

effect of plantations on biodiversity has been found to vary considerably depending on the original land cover, with positive effects most likely to occur when plantations are established on degraded or intensively used lands (Bremer and Farley, 2010). This agrees with our findings that afforestation on improved grasslands has a more positive effect on species diversity than afforestation on semi-natural grasslands. Generally, given the evidence presented here, 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. The lack of strong evidence for the effect of naturally regenerating native woodland on biodiversity means that no firm conclusions can be made, and this is an important area for future research.

Given the likely species turnover from open-ground specialists to generalists and forest-specialists shown by this review, and evidence for the development of woodland flora over long time-scales, any assessment of overall changes in biodiversity will inevitably involve a subjective choice between species assemblages. A shift in focus of the biodiversity metrics used could be more informative for land use change decisions. Most of the evidence here is based on plot-based samples for single taxa, defined as alpha diversity. It has been suggested that measures of beta (spatial) and gamma (total) diversity may be better suited to assessing land use change at the landscape level (von Wehrden et al., 2014). Long-term studies could also help us to consider the time over which species groups/assemblages fluctuate after land use change, with slower response by specialist species compared to more generalist species. There is also increasing focus on metrics such as functional diversity, or the roles that groups of species play in an ecosystem (Aerts and Honnay, 2011). Only one paper found by this review uses functional diversity as a metric (Spake et al., 2016), and we suggest that future research should focus more on metrics such as this. There are also increasing numbers of attempts to link biodiversity to ecosystem services, with the majority of relationships being found to be positive, although many are still poorly understood (Harrison et al., 2014). This is an important area for further research.

4.3. Evidence gaps

The significant lack of evidence for the effect of afforestation on provisioning and cultural ES is interesting given that the primary objective of many woodlands is for fibre (i.e. timber), and that they also play major cultural and recreational roles in the UK and beyond (Ward Thompson and Aspinnall, 2011). We propose that biomass and timber are perhaps taken for granted in terms of ES provision, due to their very tangible outputs, and that yields are most commonly recorded in other forms (e.g. in national inventories rather than academic research papers). Recreation is a commonly used indicator for cultural ES from woodlands (Scarpa et al., 2000; Edwards et al., 2009; Quine et al., 2011; Moseley et al., 2017). It may not have been picked up by this review due to the focus of the search strategy on woodland creation, as it is easier to measure recreation use in established woodlands than to monitor its development over time. Methodologies to measure other cultural ES (i.e. aesthetic, heritage, symbolic, existence values) have only begun to be significantly developed in recent years (Chan et al., 2016; Kenter, 2016), and the lack of evidence may be compounded by the fact that people may more easily recognise values for established woodlands, but less so for newly created ones. This knowledge gap is a common problem beyond the UK. The social aspects of ecological restoration, in particular negative ones such as farmers' worries about loss of land, have not been studied as frequently (Bullock et al., 2011), and cultural ES were not reported on in a global meta-analysis studying the effect of ecological restoration on biodiversity and ES, as cultural services were not measured explicitly by any of the restoration studies reviewed (Rey Benayas and Bullock, 2012).

4.4. What does this mean for developing strategies for afforestation that maximise biodiversity and ES provision?

Given the multiple objectives for afforestation in the UK and internationally, there is a need to develop strategies which promote synergies and minimise negative trade-offs between ES and with other land uses. Considering the evidence base outlined by this review, this final section aims to reflect upon how this base might be used to develop effective strategies for afforestation in the UK.

4.4.1. Incentivise public goods with strong evidence behind them

The results of this review show that evidence for the effects of afforestation is mostly biophysical, and can be classified as relating to 'public goods', in that it shows substantial but general and largely intangible benefits to society through climate mitigation and flood risk reduction. It has previously been recognised that the strong desire for change (e.g. policy aims for afforestation) is driven by public good values, and that the public good argument may simply not be effectively operationalised, with incentives such as subsidies and grants not being put behind the evidence to encourage planting (Slee, 2006). To date, payments as part of the Common Agricultural Policy (CAP) have had a very strong influence on land use in the UK and Europe more widely, with agricultural production being prioritised and subsidised (Van Zanten et al., 2014). CAP reform could create new financial incentives for afforestation, drawing on the evidence base for carbon sequestration and flood regulation benefits in particular. However, reform such as this has been noted to present a formidable challenge, with money for woodland benefits having to compete with other grant systems (Slee, 2007; Slee et al., 2014). Spatially explicit research on locating plantation forestry in New Zealand suggests that where net private benefits are negative, public support such as Payment for Ecosystem Services (PES) should be implemented (Barry et al., 2014). PES would differ from traditional woodland grant schemes by providing a more continuous stream of income for the ES provided. The recently published 25 Year Plan for the Environment emphasises that whatever may follow the CAP post-Brexit will be strongly focused on a natural capital approaches, with a new Environmental Land Management System which will pay farmers public money for public goods (HM Government, 2018). This shows that there is a significant opportunity for public funds to be put behind the evidence presented here for the benefits of woodland expansion.

4.4.2. Further develop integrated modelling approaches to assess land multifunctionality

Policy objectives for afforestation in the UK aim for 'multiple benefits', but there is very limited spatially-explicit evidence for the effect of afforestation on more than one ES at a time. Given the increasing focus in ES research on developing integrated modelling approaches (Costanza et al., 2017) and assessing ES relationships (Cord et al., 2017), this is a clear area for further research. Within woodlands, the potential for multifunctionality is high. A European-wide study has shown that there is a high unrealised potential for multifunctionality in EU forests, with this being dependent on management (van der Plas et al., 2017), while a UK review has shown that diverse management is beneficial to ES provision (Sing et al., 2017). Nonetheless, despite the high potential for multifunctional woodlands, at the landscape scale, the complexity of trying to achieve multifunctionality is huge. As a target, it is generally considered at the landscape scale, meaning that attempts to achieve multifunctionality must tackle complex interactions among multiple land covers, land uses and stakeholders (Mastrangelo et al., 2014). Amongst these stakeholders, there are a multitude of different values, and balancing the wellbeing of diverse stakeholders often involves different types of trade-offs, some of which may be 'taboo trade-offs' between morally incommensurable values (Daw et al., 2015). Further complexities relate to methodological shortcomings. Landscape assessments are usually based on the transference of ES

values obtained at ecosystem level (Mastrangelo et al., 2014). This issue is reflected by the results of this review, with most evidence being collected at the site scale, despite ES assessments requiring information relevant or applicable to larger scales. The choice of ES assessed is also often dependent on the availability of data and models, and assessments rarely incorporate stakeholders' visions and preferences meaningfully (Mastrangelo et al., 2014), meaning that the ES that really matter to local people are not always evaluated. There is also much debate relating to alternative strategies of land sharing (making farmland more friendly for biodiversity) and land sparing (making more space for unfarmed habitat). Discussion in this debate has become polarised based on misinterpretation of the many definitions and related concepts, and it is argued that insights from use of the model should be integrated with social and political knowledge, while recognising that choices made relating to land use change will always be underpinned by ethics (Phalan, 2018). Cord et al. (2017) highlight the potential for scenario approaches to further explore the biophysical constraints of landscapes and potential limitations for multifunctionality, but also acknowledge that significant challenges remain in terms of integrating the biophysical focus of many studies with stakeholder preferences. Given the lack of research on stakeholder preferences illustrated by this review, we next consider why this may be the case, and how further research might tackle this.

4.4.3. Social barriers

As previously noted, woodland expansion, in particular in relation to its potential role in climate change mitigation, has been described as a 'wicked problem' (Duckett et al., 2016). Stakeholders have different perspectives and goals, both in relation to the successful implementation of woodland planting, and to climate change in general (Duckett et al., 2016). Despite significant evidence for public goods arising from afforestation, barriers to woodland creation have been shown to be mostly social. Studies have shown that landowners may be reluctant to plant trees for many reasons. Despite decades of Forestry Commission grants for new woodland and attempts to create voluntary carbon markets for woodland (Forestry Commission, 2014b), the desired levels of planting have not been achieved (Thomas et al., 2015). There is a wide cultural gap between forestry and farming in the UK (Scambler, 1989; Duesberg et al., 2013; Wynne-Jones, 2013; Slee et al., 2014) and a bureaucratic application process, as well as a lack of information and advice, have been further cited as discouraging factors (Lawrence and Dandy, 2013; Lawrence and Edwards, 2013; Moseley et al., 2014). The forest ownership structure has had a major role in this divide, with rights to trees on tenanted land often vested in the landlord, resulting in alienation of tenants from the farm woodland on their land (Wong et al., 2015). As a large majority of land and forests in the UK and Ireland are owned privately, woodland expansion requires the involvement of private landowners, a large number of whom have been found to have generally negative attitudes to woodland creation (Lawrence and Dandy, 2013). The public good argument can also not be assumed to be inherently effective in generating action. The focus on woodland for carbon sequestration in Wales has been characterised as a distraction from the development of better governance strategies that learn from literature on farmer behaviour and uptake of previous environmental schemes (Wynne-Jones, 2013). Real or perceived trade-offs of new woodland with the ES or profitability of other land uses may have a role in holding afforestation back, and disagreements are often rooted in the core values and behaviours of land managers (Slee et al., 2014). Research shows that 'nudge' type approaches, along with deliberation with stakeholders, may help to overcome misconceptions (Moseley et al., 2014). These can include providing defaults and prompted choices e.g. adding woodland creation (with an emphasis on climate change mitigation) to application forms for grants for land management (Moseley et al., 2014).

4.4.4. Trade-offs and synergies

Taking into account these barriers, elsewhere it has been argued that the focus of previous research on biophysical potential for multifunctionality has obscured the importance of social factors, such as taboo trade-offs, or incommensurable values (Daw et al., 2015; Cord et al., 2017). The emerging Forest and Landscape Restoration (FLR) agenda argues that we lack the knowledge needed to operationalise and implement restoration successfully at different scales whilst also addressing the needs and aspirations of landholders, and that however much evidence supports the potential value of afforestation, social acceptability often lags behind (Chazdon et al., 2017; Ghazoul and Chazdon, 2017). With very few studies relating to public or land holder preferences for afforestation and land use change, this review confirms this knowledge gap. However, the limited number of findings do suggest that a more locally focused approach to afforestation may help to ensure that strategies take account of public preferences.

Any type of land use change is expected to generate winners and losers, with conflicts based on stakeholder values, and it is argued that there is increasing need for deliberative and participatory research methods to understand these conflicts (Martinez-Harms et al., 2015; Valluri-Nitsch et al., 2018). Significant challenges remain in terms of choosing standard values for decision making around ES (Cord et al., 2017), and attempts to develop these at the national scale have been criticised, due to the difficulties in reaching consensus developing appropriate indicators for particular settings (Slee, 2007). Trade-offs and synergies with other land uses, and between ES, are going to be context specific (Chazdon et al., 2017) and therefore local assessments which involve active participation of land managers are argued to be necessary to make discussions and decisions around socio-cultural effects of afforestation clearer (Slee et al., 2014). Recent work advocating the use of relational and shared values between stakeholders will be particularly beneficial (Chan et al., 2016; Kenter, 2016), as may working with visions, or 'positive scenarios of ideal futures', which can help to highlight areas of common ground and initiative discussion and collaboration between stakeholders (Burton et al., 2018).

4.4.5. Implications

Overall, given the strong evidence for public benefits from afforestation, together with social barriers and a lack of evidence for socio-cultural effects, there is a need for more context specific, participatory research with multiple stakeholders to better assess trade-offs and synergies generated by afforestation in different contexts.

Although not a novel idea, an increasing number of different areas of research, as well as practical landscape scale initiatives, have emerged in recent years which can guide further research or actions in this area. In a Scottish context, it has been argued that locally focused action research and collaborative learning will help to better understand and resolve conflicts (Slee et al., 2014), and the Regional Land Use Partnerships piloted as part of the Scottish Land Use Strategy have been an attempt to put this into action (The Scottish Government, 2016). Lessons from these may be applicable to the UK as a whole and more widely. Many other landscape scale initiatives internationally are piloting similar ideas, increasingly focused on involving local stakeholders in dialogue and decision making for sustainable development, for example Model Forests (Bonnell et al., 2012) and UNESCO Biosphere Reserves (Ishwaran, Persic and Tri, 2008). Globally, the FLR agenda proposes a framework for integrating agricultural and restoration/environmental policies, conceding that there will be a mixture of 'muddling through' with the best available evidence (Sayer, Bull and Elliott, 2008) whilst also developing cross-level environmental governance (Brondizio, Ostrom and Young, 2009; Chazdon et al., 2017). Overall significant challenges remain in terms of improving understanding and coordination at local levels, while also coordinating actions at a national level to ensure that policy goals for afforestation are met.

5. Conclusion

This review has characterised the evidence base for the effect of woodland expansion, encompassing afforestation, reforestation, woodland creation, and forest landscape restoration, on ecosystem services in the UK. Currently the largest body of evidence exists for the effects of conifer plantations, and public benefits such as carbon sequestration and water regulation. Evidence gaps need to be filled in relation to: a broader consideration of other taxa and metrics for biodiversity; natural regeneration; native woodland; farm woodlands; cultural and provisioning ES and particularly multiple ES. We recommend that site specific and, if possible, long-term research should be carried out on naturally regenerating and new farm and community woodlands in particular. The public good argument needs to be more effectively operationalised in order to meet planting targets, perhaps through new forms of incentives relating to Natural Capital or Payment for Ecosystem Services (PES) schemes. In addition to this, we argue that context specific, participatory research and implementation may be the best way forward in terms of assessing the effect of woodland expansion, and in making the best decisions for land-use in the future.

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