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REVIEW AND SYNTHESIS

The multiple values of nature

A review of planting principles to identify the right place for the right tree for 'net zero plus' woodlands: Applying a place-based natural capital framework for sustainable, efficient and equitable (SEE) decisions

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Abstract

1. We outline the principles of the natural capital approach to decision making and apply these to the contemporary challenge of very significantly expanding woodlands as contribution to attaining net zero emissions of greenhouse gases.
2. Drawing on the case of the UK, we argue that a single focus upon carbon storage alone is likely to overlook the other 'net zero plus' benefits which woodlands can deliver.
3. A review of the literature considers the wide variety of potential benefits which woodlands can provide, together with costs such as foregone alternative land uses.

4. We argue that decision making must consider all of these potential benefits and costs for the right locations to be planted with the right trees.
5. The paper closes by reviewing the decision support systems necessary to incorporate this information into policy and decision making.

1 | AN INTRODUCTION TO THE PROBLEM

Determining 'The Right Tree for the Right Place' has been a consistent focus for academic research from the first publication with that title during WW2 (Minckler, 1941) up to the present day (Di Sacco et al., 2021). However, a review of this literature shows that the definition of 'right' varies widely across studies; further expanded by the more recent extension to find 'The Right Tree for the Right Place for the Right Reason' (Broadmeadow, 2020), in the right way (Buchan, 2021).

Definitions of the 'right' set of issues to consider in determining planting decisions have often been highly constrained and differ fundamentally across studies. Examples include a focus on raw materials for the timber industry (Minckler, 1941; Nieuwenhuis & Williamson, 1993); mitigating the impact of agricultural ammonia emissions (Bealey et al., 2016); reducing the salinity of waterways (Cleary et al., 2010); urban benefits (Grant, 2016; Mersey Forest, 2014; Summit & Sommer, 1998) such as heat mitigation (Morakinyo et al., 2020) and reducing the risks of skin cancer for commuting pedestrians (Langenheim et al., 2020); biodiversity (Betts et al., 2021); poverty reduction (Leakey et al., 2005), mental health benefits for visitors (Maes et al., 2021); the enhancement of farming incomes (Asaah et al., 2011; Jack & Santos, 2017); and, increasingly, greenhouse gas storage (Bradfer-Lawrence et al., 2021). All these analyses provide important information on a dimension of the consequences of silviculture and arboriculture, particularly with respect to afforestation and urban greening. However, such 'single focus' assessments are at best only partial guides to the complex panoply of impacts generated by afforestation; a criticism which, in principle, applies equally to major contemporary challenges such as providing habitat for biodiversity (Lee & Thompson, 2005) or addressing climate change (Fontaine, 2014; Fontaine & Larson, 2016).

Applying a single focus to desired objectives is a common mistake in decision and policy making, particularly when those decisions and policies concern the natural environment (UNEP, 2021). It is typified by cases such as the EU Common Agricultural Policy, whose founding focus upon food production has incentivised decades of environmental degradation across Europe (Davidson & Lloyd, 1977; Pe'er et al., 2020). While over-complicating a problem relative to the decision is itself a danger, it is simply unrealistic and potentially disastrous to apply a single focus perspective to decision making in a world where two complex systems, the environment and the economy, interlink with each other in multiple ways to deliver a wide array of positive and negative effects on human well-being. The consequences of such a simple approach alert us to the warning of H. L. Mencken that, for every complex problem there is an answer that is clear, simple—and wrong.

The apparent simplicity of the 'Right Tree for the Right Place' slogan, with its focus on species choice, contrasts with the reality of the inherent socio-political, environmental, ecological and economic trade-offs that are involved. As the title of a recent paper acknowledges, 'Tree planting is not a simple solution' (Holl & Brancalion, 2020); rather tree planting projects 'can be an important component of ensuring the well-being of the planet in coming decades, but only if they are tailored to the local socioecological context and consider potential trade-offs' (ibid.; p. 580). Indeed, we would argue that the very large variation in benefits and costs which can occur from changing planting locations has the potential to swamp the significant, yet relatively smaller variation induced by choice of species, an argument supported by recent research (Gregg et al., 2021). This means that the more pressing research question is not to determine 'The Right Tree for the Right Place' but rather 'The Right Place for the Right Tree'.

'The Right Place for the Right Tree' is a systems question and requires a systems answer. This is particularly the case for woodlands which are far more than simple collections of trees but distinct ecosystems. Responding to this, the literature has seen single focus assessments progressively supplemented by several wider analyses of woodland planting strategies (Bateman et al., 2003; Brancalion & Chazdon, 2017; Di Sacco et al., 2021; Evans, 1992; Garrity et al., 2006; Hale et al., 2015; IUCN, 2020; Kovacs, 2015; Maser, 1990; Nolan, 2016; Susse et al., 2011; TDAG, 2021). However, no single analysis has yet addressed a sufficient set of issues to deliver that systems answer. This review sets out to add to this literature through an overview of a comprehensive set of issues. The principles set down here should have wide application, but are illustrated using the UK as a case study, partly because of the severe decline in woodland prior to the establishment of the Forestry Commission in 1919 (Gambles, 2019). More pertinently, the UK forestry policy sector has recognised the need to incorporate the wider benefits and trade-offs of woodland within planting strategies, embracing the multi-purpose nature of woodland (Quine et al., 2013). For example, the UK Forestry Standard (UKFS), which dates back to the 1990s, provides a broad perspective on forest planning (Forestry Commission, 2021); an initiative which is extended through a variety of further policy documents (e.g. Forestry Commission Scotland, 2010; Forest Research, 2018; The Scottish Government, 2019a, 2019b, 2019c; Forestry England, 2020; UK Government, 2021). This case study is also prompted because of the recent decision to substantially expand UK forestry as a part of the Government's commitment to reach net zero emissions of greenhouse gases by 2050 (Houses of Parliament, 2019; Priestley, 2019). This comes at a time when global interest in forestry expansion is rising rapidly (COP26, 2021), but there is a historical low in the creation of new UK woodlands over the last century; while 5-year average planting to 1990 was well over 25,000 ha per annum, this had fallen to less than 10,000 ha by 2020 of which less than 15% was undertaken in England while more than 80% was planted in Scotland (Forestry Commission, 2020a). Using forests as a cost-effective route to directly remove greenhouse gases from the atmosphere (R. Soc., & R. Acad. Eng., 2018), the UK government plans to accelerate planting up to 30,000 ha per annum by 2025 (Ares et al., 2021) and follow UK Climate Change Committee (2019, 2020) advice to maintain or exceed this level through to 2050 resulting in a total expansion of 750,000 ha or 3% of UK total land area, increasing overall woodland coverage to 16%. This coincidence of an enlightened forest policy sector and central support for woodland expansion makes the UK a useful case study for principles which will have more general applicability.

In the following section, we show how the 'natural capital' framework for decision making provides the necessary multiple focus approach to planning for woodland expansion. In Section 3, we detail the range of issues which need to be included within such an analysis. Section 4 considers the decision support systems necessary to address these wider objectives and concludes.

2 | THE NATURAL CAPITAL APPROACH TO SUSTAINABLE, EFFICIENT AND EQUITABLE (SEE) DECISION MAKING

The progressive move towards bringing real-world complexity into forestry planning is driven by an ongoing adoption of environmental-economic systems thinking into policy decision making more generally (Millennium Ecosystem Assessment, 2005). Within the UK, this has been conceptualised through the incorporation of the natural capital framework (Bateman & Mace, 2020; Dasgupta, 2021; Maddison & Day, 2015; Natural Capital Committee, 2013; UK NEA, 2011) as the bedrock of policy formation regarding the relationship between the environment, economy and well-being, as reflected in the 25-Year Plan for the Environment (H.M. Government, 2018) and the Environment Bill (2020). Government advice on the practical enabling of the natural capital approach has recently been issued (Defra, 2020) while official Treasury guidance on the appraisal and evaluation of public spending projects has been reformulated to place natural capital principles at its heart (HM Treasury, 2020), an approach endorsed by the UNEP (2021).

The natural capital framework illustrated in Figure 1 provides a simplified overview of the connections between the environment, economy and well-being. It uses these links to identify three principles for better decision making: delivering sustainability, enhancing efficiency and improving equity. The interconnections between the environmental system and the economic system mean that policy action in one of these areas inevitably affects those others and therefore an integrated approach to policy creation is clearly important. However, for clarity, we initially consider these separately.

2.1 | Sustainability

Sustainability is typically defined in terms of ensuring non-declining opportunities for well-being across generations (UN, 1987). This

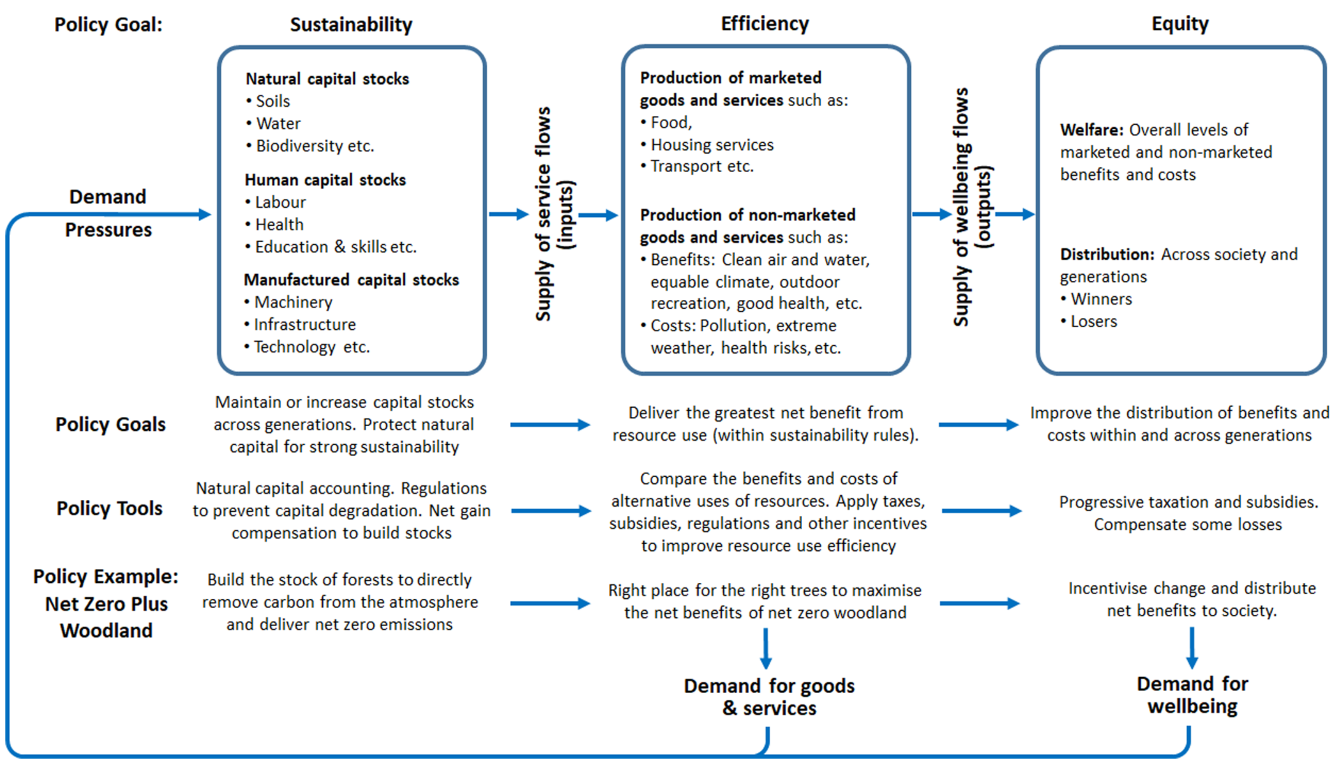

FIGURE 1 Natural capital framework for sustainable, efficient and equitable (SEE) policy formation and decision making with examples from UK woodland creation

requires, at very least, that the aggregate value of stocks of all capital assets (resources capable of providing the flows of services and production inputs that maintain welfare) should not decline over time. If all types of capital (natural, human, manufactured, etc.) are perfectly substitutable (i.e. the functions of one may be completely undertaken by another; for example timber replacing steel in buildings) then ensuring that the total value of capital does not decline over time is sufficient for sustainability; this is known as the 'weak sustainability' rule (Neumayer, 2012). However, research suggests that, even with foreseeable technological change, substitutability between natural capital assets (such as water, fertile soils, an equable climate) and other forms of capital may be moderate to low (Cohen et al., 2019; Fitter, 2013). Degrading non-substitutable 'critical' natural capital (Daly, 1991; Ekins et al., 2003; Pearce & Turner, 1990) may cross 'tipping points' beyond which resilience (ability to resist or recover) is compromised (Perrings, 2006). In such circumstances, a 'strong sustainability' rule is required (Neumayer, 2012). Here both the aggregate value of all capital is maintained across generations, and critical natural capital is kept within the planetary boundaries described by Rockström et al. (2009) as defining the 'safe operating space for humanity' (see Supplementary Material [SM]).

Attempts to assess the state of natural capital stocks are beginning to make their way into policy and institutions. Within the UK, the Office for National Statistics now publishes annual natural capital accounts (ONS, 2020a) while the UN System of Environmental Economic Accounting (UNSC, 2021) provides international standards such as the 'Inclusive Wealth' assessment of all forms of capital (Managi & Kumar, 2018) and the Gross Ecosystem Product (GEP) measure of natural capital service flows (Ouyang et al., 2020) for direct comparison with high policy-impact economic metrics such as Gross Domestic Product (GDP). These measures (and more conventional scientific quantifications of loss; Rockström et al., 2009) reveal the long-term decline in natural capital, accelerating from the middle of the 20th Century to the present day reaching levels which are now clearly unsustainable.

The UK Government's commitment to reach net zero emissions of greenhouse gases by 2050 describes one of several sustainability policies to which woodland creation can contribute. While the current impetus for tree planting globally is driven by climate change, woodland protection and creation can also assist in addressing biodiversity decline. Hence, woodland creation should not be viewed from the single focus perspective of greenhouse gas removal, but rather as a multi-functional 'Net Zero Plus' land use, where the 'plus' emphasises that woodland can deliver multiple benefits (of which greenhouse gas removal is a high value but not sole output), all of which should be considered alongside all costs. The challenges of choosing from a variety of options bring us to the issue of efficiency.

2.2 | Efficiency

While technological progress can increase the productive capacity of resources, few natural resources can be considered unlimited. These resource constraints mean that every time we make a decision and choose a policy or investment option, then we are foregoing alternative uses of those resources. These opportunity costs mean that there are no costless options in the world. It also means that every time we choose one option over another, we are revealing a value; the chosen option is more highly valued than the rejected option. Valuation is the essence of decision making. Of course, many decisions are made from the perspective of the private decision maker and may undervalue or ignore benefits to others. It is the task of the social decision maker to ensure that their decisions account for all the benefits and costs accruing to all members of society.

Ensuring decisions make efficient uses of resources is no minor task. A first challenge is to extend the appraisal of all the positive and negative effects of a potential decision as far as possible. Decisions affecting natural capital are particularly complex because changing one element of the environmental system invariably affects other elements. For example, planting trees to store carbon will almost always have effects upon biodiversity, food production, water systems, recreation and so on with some of these being co-benefits and others costs.

A second challenge is that these diverse benefits and costs often arrive in a plethora of non-commensurate units, such as tonnes of carbon, the market price of food, water quality measures, biodiversity assessments, visitor numbers, etc. Direct comparison of such metrics is meaningless for decision makers concerned with changes in social welfare. What is needed is conversion into measures that really matter, their contribution to the welfare of all affected individuals. The practical measurement of changes in welfare is a longstanding challenge but typically involves examining how much of one thing an individual will give up to gain another, with money being used as the fungible unit of exchange. This approach is not flawless as way of assessing underlying welfare, rather it is the least bad alternative. Given that resources are not infinite, trade-offs are inevitable. Furthermore, as money is usually the basis of policy analysis, this ensures that all benefits can be given due weight in the allocation of government budgets.

Economic valuation is relatively easy to undertake when the good in question is provided through a market and is therefore priced. To extend this approach to non-market items, including many environmental benefits and costs, a variety of valuation methods have been developed over the past half-century (Freeman III et al., 2014) and approved for official use (HM Treasury, 2020). These include approaches which examine the contribution of the environment to the production of goods and services (e.g. the role of pollination in producing food, see IPBES, 2016), values reflected in human behaviour (e.g. the sacrifice of time and money to visit recreational sites; see Freeman III et al., 2014) and related purchases (e.g. the amount house buyers are prepared to pay for homes in quieter locations; see Day et al., 2007), the amounts that survey respondents are prepared to pay for environmental improvements (e.g. increases in water bills for cleaner rivers, Metcalfe et al., 2012).

One area where robust values may not be available is in respect of biodiversity benefits. Biodiversity delivers a plethora of benefits, including pollination (and human well-being benefits gained from birdwatching [both 'use values']). One of the major roles of biodiversity is in maintaining ecosystem functioning through modulating atmospheric, water and soil cycles. Scientific understanding is still incomplete to the extent that we do not have the firm basis for robust economic valuation. A further problem arises with respect to 'non-use' benefits generated by biodiversity such as the 'existence value' of ensuring that wild species are safe from extinction. As such non-use values are often not reflected in observable behaviour, some advocate the use of survey based stated preference valuation techniques (Tonin, 2019). However, these methods have significant drawbacks as discussed by Bateman and Mace (2020).

We therefore have a measurement problem: we know that biodiversity is valuable, but we do not know how to measure that value robustly. We also know that biodiversity is under threat (Leclère et al., 2020) so allowing it to default to a zero value would lead to grave decision errors. Accordingly the Treasury Green Book promotes the approach of Bateman et al. (2013), noting that in cases where 'estimates of biodiversity value are insufficiently robust for use, an alternative is to use quantitative metrics of biodiversity change as objectives and calculate the costs of delivering those objectives' (HM Treasury, 2020; Section A1.29, p. 81). In effect, if those objectives are no net loss or net gain, this introduces the costs of sustaining biodiversity into investment appraisals (the practical details of which are under current investigation; HM Treasury, 2021, p. 22).

A further essential part of delivering efficiency of resource use is to consider alternative methods of delivering desired outcomes. With respect to investments affecting land use, perhaps the most obvious and often most effective permutation to consider is the impact of changing the location of actions. While this is complicated by the private ownership of land, necessitating different levels of incentivisation and policy action, changing the location of tree planting initiatives can radically alter the benefits and costs they generate. At the extreme, planting on certain peat soils could result in woodlands whose sequestration of carbon is outweighed by the emission of greenhouse gases generated by the drainage and drying of peatlands (Matthews, 2020). Conversely, appropriate targeting of tree planting can deliver landscapes which not only contribute significantly to greenhouse gas removal, but also generate high value added wider net benefits (Quine & Watts, 2009; Watts et al., 2010). Similarly, while the focus of current policy change is squarely upon the creation of new woodlands, the benefits of avoiding the loss of existing woods (e.g. to infrastructure planning) should not be forgotten (Di Sacco et al., 2021). However, alternative investments should also assess entirely different technologies for delivering objectives. A contemporary example of such thinking is provided by the UKRI Greenhouse Gas Removal (GGR) programme (UKRI, 2021). This considers multiple approaches to the delivery of GGR including large-scale tree planting, the management of peatland`s, enhanced rock weathering, the use of biochar and bioenergy crops.

How decisions will be implemented will also impact efficiency. Private ownership of many resources, and reliance on private individuals to act appropriately, is a major factor. The land use change required to meet net zero cannot be planted without the conversion of privately owned, mainly agricultural land. This requires substantial regulatory and policy change together with a radical shift in the focus of incentives. For the public decision maker, the objective here is to use both positive and negative incentives (e.g. subsidies and regulations) such that the optimum strategy for the private producer is one which also delivers a better outcome for society. The Public Money for Public Goods principle (Bateman & Balmford, 2018) now embedded within the UK 2020 Agriculture Act provides the basis for such a change, although to date a clear policy statement linking agricultural subsidies to this level of woodland creation has not been forthcoming.

2.3 | Equity

While efficiency analysis can be characterised as ensuring that we maximise the size of the cake generated by resource use, equity analysis assesses who gets the slices of that cake. Increasingly environmental policy is being used by decision makers as a vehicle for improving well-being for those who endure poor access to high-quality environments. Delivering to the UK Government's 'Levelling Up' agenda (MHCLG, 2021) is an explicit objective of the England Trees Action Plan (H.M. Government, 2021) while the Scottish Government's 'Just Transition' policy (Scottish Government, 2021) and the Welsh Government's 'Well-being of Future Generations Act' (Welsh Government, 2015) intimately connect environmental and equity policy objectives.

Inequality is both a present and intergenerational issue. Society's preference for present over future benefits (and its preference for future rather than present costs) is captured within the discount rate (Prest, 2020); the rate at which future returns are related to their present value equivalent so that decisions regarding the future can be made in the present. If concern for future generations increases, for example in the context of climate change, this should be reflected in a reduction in the rate at which future benefits and costs are discounted, as reflected in official guidance (H.M. Treasury, 2020). However, the extended nature of forestry investments means that even at low discount rates the present value of much delayed benefits, such as carbon storage and timber production, can appear relatively minor. While discounting reflects preferences, it cannot be allowed to impinge on the requirements for sustainability set out previously; a sustainable society is sustainable at all points in time, including the future (Hunt et al., 2012). Objectives such as tackling climate change and reversing biodiversity loss are sustainability issues and must be achieved.

We now turn to apply these principles to provide a comprehensive assessment of the issues that need to be considered to move from a single focus approach to tree planting to an application of the natural capital framework to delivering net zero plus woodlands (although these issues also apply to trees outside woodlands, hedgerows and urban trees).

3 | NATURAL CAPITAL PRINCIPLES FOR PLANTING 'NET ZERO PLUS' WOODLANDS

This section outlines the issues which should be considered when deciding the location and type of planting necessary to deliver net zero plus woodlands. The sustainability requirements of the natural capital approach mean that the ability of woodlands to enhance climate change resilience and reduce biodiversity loss must be incorporated as objectives. Efficiency requires that we incorporate into decision making the wide diversity of benefits and costs which woodlands can generate, irrespective of whether those values arise within or outside markets. By appraising all relevant benefits and costs and using economic valuation techniques to place them on an equal footing with market priced goods, we show the true value of woodlands relative to alternative land uses. Equity concerns mean that all these positive and negative impacts should be assessed not only in terms of their magnitude, but also in terms of who in society they affect. This allows us to examine the distributional effects of alternative land use decisions carried out in differing locations. Finally, while the above discussion has treated the issues of sustainability, efficiency and equity separately, the delivery of all three requires their integration. Efficiency must be pursued within the constraints of sustainability while there may be trade-offs between equity and efficiency. Bringing all three issues into consideration together requires improvements in the way that decisions are made, and policies formed. But the investment in decision making necessary to implement these improvements will be repaid many times over through the delivery of a sustainable, efficient and equitable society.

All the factors discussed below, from those that appear most objectively measured (such as the timber values produced by woodland, or the agricultural value lost when land use changes) to the most subjective (say the aesthetic value of landscapes) are actually only known with some degree of uncertainty. This uncertainty is itself sometimes only incompletely understood (Beven, 2008; Pappenberger & Beven, 2006). This is a challenging but unavoidable problem made even more important because woodland creation is a process that happens in the present yet affects long periods of the future, where uncertainties may prove a very important aspect in determining outcomes. Because of this we advocate a 'portfolio' approach to assessment of the various benefits and costs described below (Costanza et al., 2000; Knoke, 2017; Lahtinen et al., 2017; Matthies et al., 2019). The uncertainties regarding each benefit and cost are themselves important information that should be incorporated into decision making. The portfolio approach then combines this with the benefit and cost information to provide a more sophisticated appraisal of the potential outcomes of different decisions. This allows the identification of a portfolio of decisions which together maximise the overall desired outcome; for example, the value delivered to society. In this approach, a set of decisions which yield the maximum possible social value, but with high uncertainty and greater downside risk, might well be rejected for an alternative portfolio of decisions offering greater environmental and socio-economic resilience (Oliver et al., 2015).

We initially employ straightforward comparisons between overtly different yet key species, such as Sitka spruce and oak, subsequently introducing material on mixed species forestry which is the focus of much current interest. Similarly, our review considers private financial benefits such as timber production before expanding to consider public benefits including greenhouse gas removal and storage, biodiversity, water, recreation, physical and mental health, landscape amenity, cultural values, noise reduction, air quality and a discussion of intrinsic value. Turning to the costs side again we consider private values such as the opportunity cost of afforestation in terms of foregone alternative output (such as food production). Related to this are incentive costs such as public sector payments to subsidise woodland creation. Other issues include employment and a variety of the same issues listed under the benefits category—for examples cases where the landscape amenity impacts of forestry turn from positive to negative. However, prior to this we first consider the factors affecting tree growth as this not only determines timber production, but also influences many other woodland benefits and costs (Hale et al., 2015).

3.1 | Tree growth: Species, spatial and temporal effects

Species, space and time bring together the three biggest determinants of both tree growth rate and the wider effects of woodland creation. Altering these three factors can radically change the benefits and costs generated by a woodland. These factors should be jointly determined; however, for clarity, we initially consider each separately.

3.1.1 | Species effects

Ideally, the choice of species should consider site conditions (such as climate and soils) as well as planting objectives (e.g. timber production, carbon storage, habitat creation or a combination of these and the other factors discussed subsequently). Considering timber production, growth rates are typically measured as the number of cubic metres of timber produced annually from each hectare of woodland (i.e. m^3/ha/year.) with the peak timber growth rate attained by a stand of trees over its lifetime being known as its Yield Class (YC). This rate can vary substantially between species and Figure 2 illustrates timber growth time courses for representative stands of one of the most common commercial conifers, Sitka spruce, and a broadleaf, oak; both growing at typical rates for the UK of YC12 (i.e. 12 m^3/ha/year.) for the former and YC4 (4 m^3/ha/year.) for the latter. Both species exhibit a classic sigmoidal (S shaped) growth function with the increase in cumulative timber production being initially low when the stand is young but growing rapidly and reaching a maximum as the stand reaches full development, after which the rate of growth declines as the stand matures (when respiratory and decomposition losses approach respiratory gains). Comparison of the

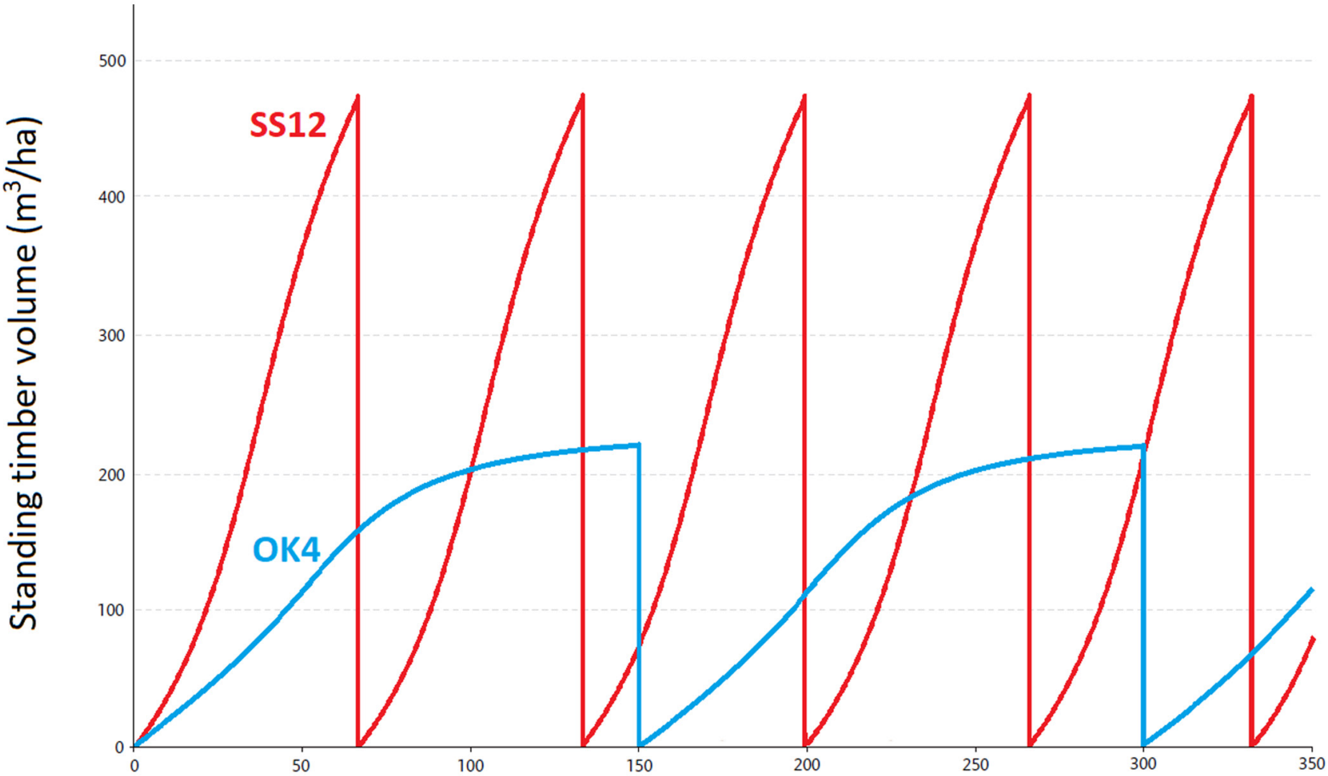

FIGURE 2 Time courses of the cumulative growth of timber for representative conifer (Sitka spruce; SS) and broadleaf (oak; OK) stands growing at typical rates for the UK (YC12 for SS and YC4 for OK) thinned for optimal timber revenues and felled at the maximum mean annual increment (MMAI) of timber production (59 years for SS12 and 150 years for OK4).
Source: Data taken from Edwards and Christie (1981); Morison et al. (2012); Matthews et al. (2016)

two curves highlights the much more rapid timber growth generated by a productive conifer stand of Sitka spruce when compared to a slow-growing broadleaved species such as oak. There is therefore a preference for conifers over broadleaves when viewed from a commercial, market-value, perspective where timber revenues are a major objective. This commercial perspective affects management (see SM), notably through the process of 'thinning' (the removal of smaller trees to maximise timber revenues). Figure 2 shows time courses where the rotation length (the time between planting and felling) is set at the maximum mean annual increment (MMAI, also known as Maximum Sustainable Yield) of timber production, although more commercially orientated management will often result in earlier felling than this (for assumptions, see Bateman et al., 2003).

As Figure 2 shows, MMAI-based rotation periods for broadleaf species can be more than twice as long as for conifers growing in similar conditions. Furthermore, conifer timber production is considerably greater over any period. MMAI felling for a YC4 stand of oak occurs some 150 years after planting at which time just under 250 m^3/ha has been produced. However, in the same period, a YC12 stand of Sitka spruce would already have been felled twice, yielding three to four times as much wood and be well established on a third rotation. Nevertheless, in recent years, objectives have increasingly changed from a single focus upon timber production to wider benefits, with an accompanying emphasis upon mixed species forests (Bravo-Oviedo et al., 2018; Cannell et al., 1992; Pretzsch et al., 2017).

3.1.2 | Spatial effects

Looking across the landscape of developed countries such as Britain, one could be excused for thinking that trees prefer to grow on steeply sloping hillsides or to cluster on poor soils near the top of mountains; this is not the case. Trees are typically pushed into such adverse locations by competing land uses, principally agriculture, which dominates sites with favourable conditions. However, left to their own devices, trees generally grow best in good soils, at lower elevations where they can benefit from warmer temperatures, with ample but not excessive rainfall and sheltered locations with lower windspeeds (Bateman & Lovett, 1998). So, in the UK, trees grow faster on land with a south easterly aspect, warmed by the southerly sun and sheltered from the prevailing westerly wind (a factor which becomes steadily more important at higher elevations where the optimal aspect swings further towards the east to protect from the wind; Bateman et al., 2003). Figure 3 uses Forestry Commission data to model changes in Sitka spruce YC across Wales. This highlights the very substantial variation in growth rates across locations, with low growth rates in the mountainous areas around Snowdonia in the northwest and running south along the central uplands of the country, but with rates more than doubling when we consider lowland coastal areas where temperatures are warmer, and soils are more fertile. Analyses show that these relationships hold across species, with broadleaves revealing a similar pattern, albeit at lower yield classes (Bateman et al., 2003). The fact

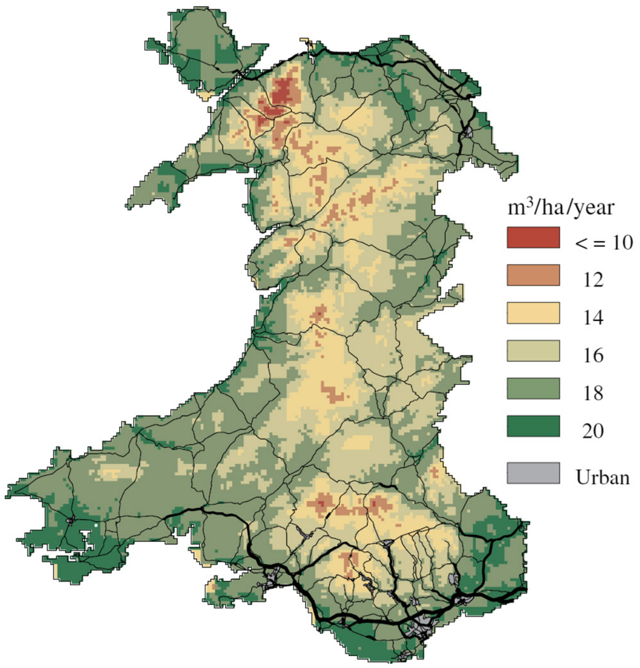

FIGURE 3 Predicted timber yield class for Sitka spruce planted in Wales.
Source: Bateman et al. (2003)

that forests are often confined to upland and low productivity locations reflects the power of external forces, most obviously, the much faster financial returns to agriculture.

3.1.3 | Temporal effects

The long periods involved in tree growth make time a major factor in forestry decision making. Rotation lengths are inversely related to YC. In the case of Sitka spruce while MMAI suggests felling at around 67 years for YC6, this declines to as little as 45 years for YC24 (Bradley et al., 1966), although earlier for financial and operational reasons is not uncommon for such commercial species (Henderson & Bateman, 1995). For oak, MMAI suggests felling at just under 100 years for YC2, falling to 70 years for YC8 (ibid.; Lemaire, 2014).

If we were to ignore the impact of time, then the commercial comparison between planting conifers and broadleaves for timber revenues only might appear finely balanced. As noted above, conifers can produce three or four times the volume of timber that broadleaves can over the same period. However, prices for broadleaf timber can exceed those of softwoods by a similar or even greater degree such that revenues might be considered comparable if we were to ignore their timing. But we are far from indifferent to time as mentioned earlier (Section 2.3), placing lower weight on delayed benefits and costs when calculating their present value. This preference is incorporated within investment analyses through 'discounting' (HM Treasury, 2020) with the weight relating future returns to their present value equivalent known as the discount rate. While governments typically adopt a relatively low 'social discount rate' reflecting society's concern for future generations, the 'private discount rates' revealed in the decisions of forestry investors are substantially higher, emphasising preferences for faster returns (Ferguson, 2018; Sauter & Mußhoff, 2018). Either rate is typically more than enough to give the edge to conifer over broadleaf woodlands where analyses are restricted to commercial returns alone (see SM).

Time also matters when other factors change the relative benefits, costs and risks associated with any venture. Conventionally, the impacts of inflation can be ignored where the nominal price (i.e. that paid for in the market) of wood move roughly in unison with those of other goods. Such assumptions are usually valid where the balance between supply and demand is likely to stay constant as is reasonably likely to be the case with softwoods. However, this is less obviously true for certain hardwoods, most obviously tropical species where forests continue to be harvested unsustainably. Here rises in the real price of such trees seem more likely and should be built into analyses. The same logic applies to those ecosystem services whose supply is degrading over time; assuming present-day supplies and values will remain constant is a major error in such cases.

One of the major and increasing drivers of ecosystem service change over the lifetime of a forest is of course climate change (IPCC, 2022), and the UK climate is already changing rapidly (Kendon et al., 2021). Figure 4 provides estimates of the impact on tree growth rates of the mean expected change in UK climate to 2060 (with climate change predictions taken from UKCP09; Met Office et al., 2017). The relationship between tree growth measures and climate-related variables (such as temperature and precipitation) is derived by looking at the spatial variation in growth across the country (e.g. Figure 3). This relationship is then used to estimate the response in yield as climate change alters temperature and rainfall. The upper row of Figure 4 shows impacts on Sitka spruce. To date, such species have thrived in the wet and cool temperate conditions prevalent along the western and upland areas of the country. Drought susceptibility is expected to reduce growth rates for such species as a result of climate change (Davies et al., 2020). Conversely, native and European species such as oak are expected to respond positively to such changes as shown in the lower row of Figure 4. Mixed species planting also provides greater resilience to drought than monocultures of exotics such as Sitka spruce (Bravo-Oviedo et al., 2018; Pretzsch et al., 2017). However, the impacts of climate change in the UK are likely to be complex, with warmer and wetter winters, hotter and drier summers, and more frequent and intense weather extremes (Met Office, 2021; Murphy et al., 2019). This complexity makes the prediction of climate change impacts upon trees challenging and reliance solely on average effects may disguise important events arising from more frequent and intense periods of extreme weather. Given such challenges, drawing from a broader palette of different species might provide a more resilient alternative to reliance upon the relatively restricted set of species currently dominating UK woodlands (Reynolds et al., 2021).

Climate change will not stop at alterations to temperature and rainfall distributions. Climate-induced increases in the frequency and intensity of weather extremes raise the risks of both windthrow (tree damage and uprooting due to high wind speeds) (Jolly et al., 2015; Klaus

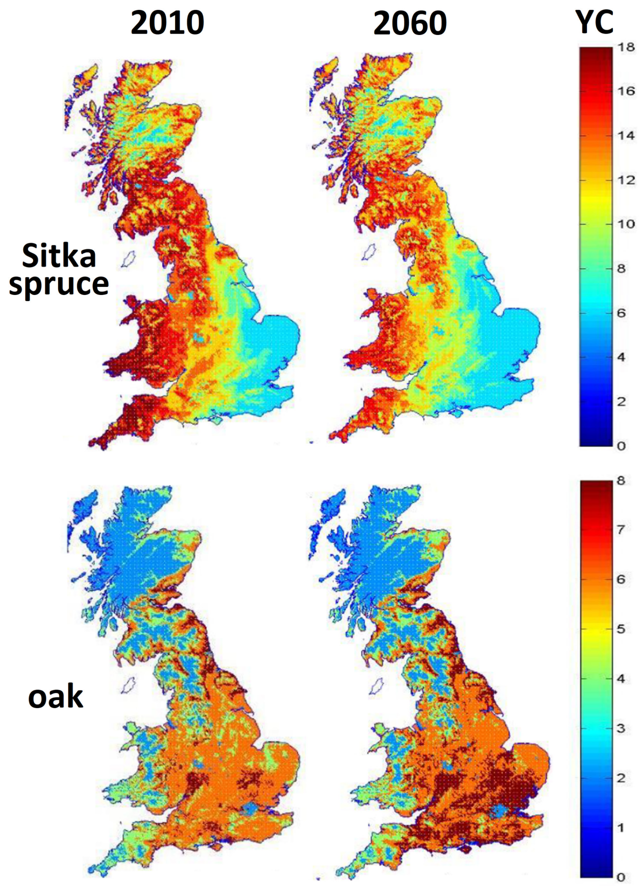

FIGURE 4 The impact of climate change on UK tree growth 2010–2060: Sitka spruce and pedunculate oak yield class (YC; m^3/ha/year).
Source: Bateman et al. (2014)

et al., 2011; Saad et al., 2017; Wohlgemuth et al., 2008) and wildfire (Betts & Brown, 2021; Marrs et al., 2019; Wohlgemuth et al., 2008). Projections of future UK fire risk suggest that the percentage of summer days generating the highest level of fire risk will increase from a 1981–2010 average of about 10% to 27% under a 2°C warming scenario and 55% under a 4°C rise by 2060 (Belcher et al., 2021; Perry et al., 2021). This raises an obvious risk to the net zero agenda; one wildfire can wipe out decades of greenhouse gas removal. Species selection is relevant here with conifer stands generally being at higher risk than broadleaved woodland (Forestry Commission, 2014). Management can modify fire risk with high density stands, especially those with a heavy understory, being at significantly greater risk of fire than those woodlands with an open structure, well-managed understory, fuel breaks and a patchwork of different species (Forestry Commission, 2014; Gazzard et al., 2016; Wentworth & Shotter, 2019).

Climate-induced environmental change is also likely to increase the risks to trees from pests and disease (including fungi, bacteria, viruses, parasitic plants, nematodes, and insects such as aphids and bark beetles) as they change their breeding capabilities and geographical distribution (Frankel et al., 2012; Linnakoski et al., 2019; Wainhouse & Inward, 2016). These risks can be further exacerbated through poorly regulated trade in forest products, seed and saplings and the emergence of new pests cannot be ruled out (Frankel et al., 2012; Wainhouse & Inward, 2016). Set against a background of ongoing damage from existing pests such as grey squirrels and deer (Defra and Forestry Commission England, 2004; Forestry Commission, 2019, 2020b; Forestry Commission England, 2014; Forestry Commission Scotland, 2014; Scottish Natural Heritage, 2016) and an unprecedented increase in tree disease (Nguyen et al., 2016; Pain, 2020; Santini et al., 2013), these challenges to the UK's future forests should be incorporated within decision making.

3.2 | Woodland benefits

The benefits and costs arising from woodland are many, diverse, always related to the spatial and temporal issues raised above and often varying by species and management. This and the following section provide an overview of those benefits and costs. A number of these are financial benefits (notably timber) or costs (establishment and felling costs, foregone agricultural output values) as reflected in market prices. However, woodlands also yield a variety of non-market values, in particular benefits (such as carbon sequestration and storage, biodiversity, water environment benefits, open-access recreation, etc.), most of which can be brought into conventional policy and decision making through their translation into economic values. Many of these benefits have an equity dimension. Individuals' access to environmental recreation and its physical and mental health benefits, high-quality water environments, urban benefits and other cultural services are often unevenly distributed across society, depending upon the socio-economic status of different groups. Economic cost–benefit analysis explicitly allows for the incorporation of equity issues within decision making. This can be achieved through means such as the reweighting of benefits and costs according to the income of those upon which they fall, noting that 'the value of an additional pound of income is higher for a low-income recipient and lower for a high-income recipient' (H.M. Treasury, 2020).

Rather than propose some inevitably contentious (and almost certainly over-simplistic) prioritisation of woodland impacts, we will focus first on the market and then non-market benefits of woodland before, in the following section, presenting the market and non-market costs. However, the plurality of woodlands defies perfect ordering along these lines; as mentioned above, the same item can be a benefit or cost, large or small, depending on the nature and location of a woodland. The following ordering is therefore necessarily contextual, but the quantification and valuation of these items within decision analysis determines the extent to which we break the tyranny of single-focus assessments and embrace the diversity of woodlands.

3.2.1 | Timber production and other financial benefits

Attractive financial incentives, such as zero capital gains tax on sales of commercial forests, combined with growing interest in the environmental and recreational benefits of woodlands, have led to strong and sustained growth in UK woodland prices over the past decade (Investment Property Databank, 2020). Alongside these capital gains, commercial felling values are also free of tax making timber production the major financial operating return from forestry (Forestry Commission, 2020a). As such, timber provides the major market priced benefit of woodlands and those prices provide a key input within decision analyses.

Timber values dictate much of the management of commercial woodlands and an important element is the 'price-size curve' which affects both hardwood and softwood felling revenues (Ryan et al., 2016; Whiteman et al., 1991). Because a one tonne single tree trunk can be put to so many more high value end-uses than one tonne of small stems, the per tonne price rises steeply with the volume of each tree levelling off when a trunk passes the girth at which its end-uses are not restricted by size with a per tonne value typically more than five times greater than the same weight of wood composed of small stems (ibid.). This relationship means that overall felling revenues are increased when a forest manager progressively removes smaller stems as the forest ages allowing remaining trees to increase their girth and effectively move along the price-size curve. This process, known as 'thinning' reduces the overall per hectare timber volume of the forest and incurs some direct cost but, from a profit maximising perspective, is typically more than compensated for by the higher price per tonne of the remaining trees.

As mentioned previously, prices also vary across species, with hardwoods generally commanding substantially higher prices than softwoods. However, conifers produce greater timber volumes much faster than hardwoods. The high discount rates typifying private forester decisions mean that conifers are almost always preferred when viewed from a purely commercial market price perspective. It is only when we adopt a wider economic perspective, where all the net benefits of woodland relative to other land uses are considered, and where lower, long-term social discount rates (HM Treasury, 2020) are used to reflect greater concern for delayed benefits and costs, that the benefits of broadleaved woodlands are highlighted.

A commonly cited financial argument for expansion of domestic forestry is to substitute for imports and indeed the UK is second only to China in terms of the value of its net imports (imports less exports) of forest products, with net imports worth US $8.6 billion in 2018 (Forest Research, 2020). However, import substitution arguments require scrutiny. More than 200 years ago, David Ricardo (1817) pointed out that countries enjoy mutual gains when they focus on producing those goods for which they enjoy a comparative advantage (the ability to produce at a lower opportunity cost than other countries) and then trade these goods with each other. Provided that countries do not exclude themselves from mutually beneficial trade agreements (Springford, 2021), market forces should be sufficient to ensure such comparative advantages are realised.

Quite distinct from the import substitution argument, a case for encouraging some fibre security level of domestic production can more plausibly be made where potential supply interruptions might generate wider costs (Bateman, 1992). Indeed, it was the blockade of timber supplies to the UK in the first World War which in part prompted the creation of the Forestry Commission in 1919 (Gambles, 2019). The challenges of wartime blockades can, thankfully, be reasonably discounted but the combined forces of Brexit and the Covid-19 pandemic have coincided with significant supply gaps in the UK as reflected in unprecedented rises in timber prices (Combe, 2021). No current thorough analysis of the duration of supply gaps is available, however in a prescient early assessment Pearce (1990) argued that a small positive valuation of domestic production above that provided by the market could be justified as a bulwark against supply interruptions.

A final issue concerns employment and consequent incomes. Forestry operations directly employ some 16,000 people across the UK (Forestry Commission, 2020a) and this is likely to grow with size of the forest estate, any shift towards alternative management strategies (such as continuous cover forestry; Garfitt, 1995; Stokes & Kerr, 2009; Susse et al., 2011; Helliwell & Wilson, 2012; Vitková & Ní Dhubháin, 2013; Vitková et al., 2013) and the husbanding of the wider benefits of woodland. In many of the areas where this industry dominates, there is a lack of alternative employment. However, this does not necessarily mean that forestry should be expanded just because of related employment. There are always opportunity costs associated with any decision and the low level of employment activities during long rotation periods means that, per hectare, the employment impacts of forestry as opposed to alternative (typically agricultural) uses of the same land are mixed (Bell, 2014; Confor, 2018a, 2018b; Fairweather et al., 2000; Malkamäki et al., 2018). The argument in favour of forestry appears stronger if we add in downstream jobs in sawmilling and other processing (73,000 in the UK) and the pulp and paper industries (a further 62,000 UK jobs); however, a fair comparison with agriculture would require a similar extension to consider downstream food sector jobs. Furthermore, it is arguable that some timber industry jobs might be supported through imports in the absence of a domestic forest sector. Nevertheless, the total forest and timber sector employment of over 150,000 jobs clearly depends significantly on that home resource and may generate significant distributional benefits in low employment rural areas. In the absence of a rigorous contemporary study of net employment implications of the UK sector, we are unable to conclude what the overall value of this aspect of forestry might be.

3.2.2 | Greenhouse gas removal and storage

When viewed from a natural capital SEE perspective, most of the benefits and costs associated with woodland are efficiency issues, some of which have equity dimensions. However, two issues are more appropriately treated as sustainability concerns (although the most important sustainability concerns will depend upon global context): biodiversity; and greenhouse gas removal and storage. These relate to existential issues: the necessity of preventing wholesale extinction of wild species and associated losses of crucial ecosystem functioning and services; and the essential requirement to tackle climate change. This does not mean that we should be insensitive to the costs of delivering these goals; cost-effectiveness ensures against resource waste and allows saved resources to be efficiently allocated to the delivery of other benefits. Woodlands can either exacerbate or positively contribute to addressing both biodiversity loss and climate change challenges depending on the policy, planning, species-mix and management decisions taken.

Within cost–benefit analyses of public investments, the theoretically correct approach to incorporating greenhouse gas emission and sequestration is through the social cost of carbon (see SM). This approach is adopted in the UK (and USA) although in some other countries the (typically lower) marginal abatement cost (MAC; the per tonne cost of abating one tonne of carbon dioxide equivalent emission) is applied.

Woodlands affect greenhouse gases in numerous ways. As forests grow, they sequester carbon from the atmosphere and fix it in living biomass, including organic carbon in the soil, to produce carbon sinks which can be of global significance (Luyssaert et al., 2008; Magnani et al., 2007). Woodland management, including the establishment of new woodlands, thinning and felling, also alters net sequestration (Winjum et al., 1992). If a new forest is left unmanaged and unfelled, the carbon storage will increase substantially up until the woodland reaches full maturity, but over a long period of time the balance between gains and losses may approach a new equilibrium. The overall carbon storage can be increased if the trees are felled and replanted, and the derived timber used for products that store carbon. The lifetime of those products determines how soon that embodied carbon is returned to the atmosphere. While timber used for construction and furniture can last for long periods, wood used for packaging, paper and of course fuel will soon release its embedded carbon back to the atmosphere. As with almost all aspects of forestry, species plays a role here. While most species can be used for a variety of ends, hardwoods are generally more durable than softwoods (Van Acker et al., 2003). This results in some systematic differences in end-uses and generally hardwood carbon release times are somewhat longer than those for softwoods. The net greenhouse gas balance effects of using wood as a fuel depend very much on the impact this has upon consumption of alternative fuels. Clearly burning wood emits carbon dioxide and so this will be a higher emission alternative to other renewables such as solar or wind energy (although the embodied emissions in construction of these facilities should be considered). However, if wood fuel is used to substitute high fossil carbon alternatives such as coal, gas and oil, its greenhouse gas profile is more advantageous. In short, the devil is in the detail here and simple assumptions should be questioned.

Further contributions to net emission reduction arise where those products substitute for other high-emission alternatives, for example, where timber substitutes for concrete in building or where wood fuel replaces non-renewable high carbon fuels (Leskinen et al., 2018). Forests may also affect greenhouse gas emissions through their displacement of other activities. For example, if agricultural production is relocated to facilitate forest planting, this may have a neutral, positive or negative impact on emissions. Furthermore, forests affect soil carbon stocks, with the effect varying across soil type and prior use but eventually settling towards a new equilibrium over time. Each of these factors need to be considered to accurately evaluate the greenhouse gas removal and storage potential of forests.

Considering first the carbon storage benefits which arise during tree growth, the faster and higher timber yield of some conifers relative to most broadleaves typically means that they sequester carbon from the atmosphere more rapidly. Figure 2 depicts this advantage in terms of timber production; however, many broadleaves do perform better in terms of carbon storage than this might suggest. This is because they often have a higher carbon content than conifers due to a higher timber density, and additionally because there is more biomass in non-timber components in broadleaves (an average of 55% compared to 45% in conifers in the UK; Morison et al., 2012). The density varies from as low as 0.33 $t\,m^{-3}$ for Sitka spruce to around 0.56 $t\,m^{-3}$ for oak (Bateman, 1997; Lavers, 1969). So if Sitka spruce grows at its UK average of YC12, then it produces 390 m^3/ha of standing timber after about 60 years; conversely, oak typically grows at YC4 yielding just 134 m^3/ha over the same period. However, when assessing carbon benefits, the YC12 Sitka spruce stores 64 tC/ha in timber while the YC4 oak stores around 37 tC/ha, about 60% of the former (Morison et al., 2012). If the other components of the biomass (branches, roots) are considered, the percentage increases to approximately 70%. Nevertheless, while Sitka spruce will be felled and replanted at or before this age, the YC4 oak will take a further 90 years to reach MMAI felling age; by which time a conifer forest would be well established on its third rotation and will have contributed substantial timber to the carbon pool in wood-derived materials. Evaluated over this full 150-year oak rotation, the oak may have stored around 130 tC/ha in above-ground biomass, while the Sitka spruce stand will have sequestered well over 270 tC/ha at the two harvests and in the growing stand (ignoring thinnings).

Similarly, within-species rates of carbon storage are directly related to yield class, with higher growth implying greater storage in tree tissue. Thinning takes advantage of the price–size curve by removing smaller trees to increase the girth and timber quality of those remaining. This maximises timber felling revenues, but very significantly reduces carbon storage (Matthews, 2020). This direct trade-off between the market value of timber production and the non-market value of carbon storage is a clear illustration of the need to adopt a natural capital perspective when developing woodlands policy.

Within a given climatic zone, the impact of woodlands upon soil carbon depends primarily on soil type and prior use (with further factors such as temperature and water availability becoming important as we move across climatic zones; Guo et al., 2021). For long established woodlands, soil carbon content may be near a steady-state value, although this can require centuries (Ashwood et al., 2019). However, where new woodlands are established on soils which have had their carbon content depleted by prior use (e.g. ploughed arable farmland) trees can increase that carbon content over time (Bárcena et al., 2014; Matthews, 2020), although this process can be quite slow. In contrast, there may be little change if grasslands were the prior use (Bárcena et al., 2014). An important exception to this occurs where trees are planted in waterlogged peaty soils. Here the drainage of such sites prior to planting can initiate substantial release of carbon which can continue for many decades (potentially centuries). This can result in greater release of carbon dioxide than will be stored in the new trees (Bateman & Lovett, 2000; Germer & Sauerborn, 2008; Günther et al., 2020; McCalmont et al., 2021). While the last century saw extensive planting on peat soils, increased awareness of this problem means that afforestation of deep peats is now effectively prohibited in all but the most exceptional cases (Forestry Commission, 2021; Forestry Commission and Natural England, 2021; Woodland Carbon Code, 2021). Considering other soil-based greenhouse gas emissions arising from afforestation in the UK, Matthews (2020) concludes that most forests *on mineral soils* are a small sink for methane (CH_4), while forests *on wet organic soils* can be significant sources of CH_4, particularly deep peats with poor drainage. Nitrous oxide (N_2O) emissions from forests are usually low, and much lower than from most agricultural land uses, although they may rise briefly during and after harvesting and thinning operations.

Figure 5 brings together all the above determinants of the net carbon benefits of planting a new woodland. As noted above, this does rely on some assumptions, especially regarding the reduction in net emissions arising from the substitution of wood for other, carbon-intensive, products and fuels. Nevertheless, this shows the substantial benefits which woodlands can deliver in this respect.

An issue often omitted from assessment of climate change mitigation via afforestation is that of its albedo. One of the factors which determine global climate is the degree to which the surface of the world reflects sunlight, and hence heat, back into space. This surface reflectivity, or albedo, varies by global location, being lower near the equator than at the poles where the acute angle to the sun means that sunlight more readily bounces off the land surface, especially where it is snow covered. This phenomenon also varies by land use with the albedo of a forested landscape being generally lower than that of cultivated land (Betts, 2000). Woodlands are generally darker than bare or agricultural land and so absorb relatively more solar

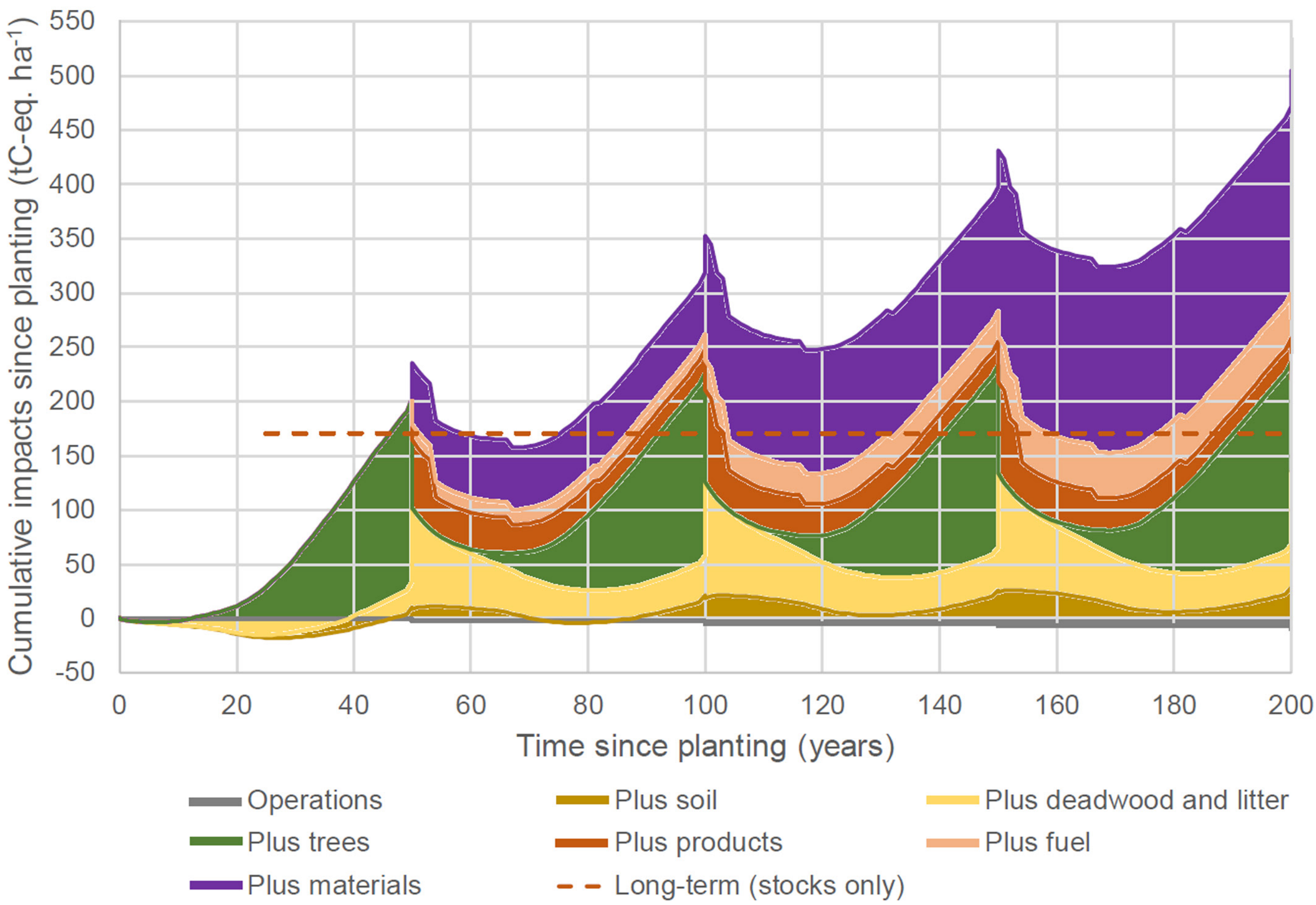

FIGURE 5 Cumulative net greenhouse gas effects of planting new Sitka spruce woodlands.
Source: Adapted from Matthews (2020). The figure assumes planting a stand of Sitka spruce with a moderate growth rate managed for timber production on a 50-year rotation without thinning. Contributions from different sources are added each year (i.e. stacked). 'Materials' refers to the reduction in net emissions arising from the substitution of wood for other, carbon-intensive, products such as bricks or steel. 'Operations' include planting and felling but produce only relatively minor emissions as shown. The dashed horizontal line shows the long-term average net greenhouse gas removal. All measures are in tC_e/ha

TABLE 1 Typical albedo values at UK or other temperate locations for different vegetation types (lower values imply lower solar reflectance and hence greater localised warming)

Land cover	Evergreen conifer forest	Deciduous broadleaf forest	Arable crops	Grassland	Moorland[d]/ heathland
Typical albedo	8%–12%	14%–18%	20%–25%	15%–20%	12%–18%
In situ measure[a]	9.6%	13.4%	16.9%	20.2%	n.a.
Satellite measure[b]	9.9%	13.6%	15.4%	17.2%	n.a.
Satellite measure[c]	10.1%	15.2%	n.a.	n.a.	n.a.

Sources: Based on Matthews (2020) and pers. comm. James Morison. (a) mean values for midsummer, midday, clear sky conditions at the 'FluxNet' network of sites (Cescatti et al., 2012); (b) MODIS satellite product derived values for the same locations as in (a) (Cescatti et al., 2012); (c) MODIS satellite product derived, white sky, summer (July–August), large-scale median value for each forest type (Leonardi et al., 2014); (d) few values are published and they vary depending on vegetation, for example, bracken compared to heather. n.a., not available.

radiation, an effect which will reduce the climate change mitigation benefits of afforesting agricultural land (Thompson et al., 2009). This effect is greatest for boreal, coniferous forests (Betts, 2000; Matthies & Valsta, 2016; see Table 1. Compared to broadleaves, conifers are darker and so reflect less sunlight. Planting conifers in high latitudes will therefore cause a relatively high reduction in solar reflectance and greater heat gain. Planting conifers on snow covered land will exacerbate this effect.

As Table 1 suggests, the magnitude of the albedo effect of establishing forests on previously unplanted land is generally significant (Thompson et al., 2009) and substantial at the extremes. Betts (2000) notes that 'high-latitude forestation would exert a positive radiative forcing through reduced albedo that in many places could outweigh the negative forcing through carbon sequestration' (Betts, 2000, p. 190). Similarly, Thompson et al. (2009) argue that ignoring albedo effects can significantly overestimate the climatic benefit of afforestation and that in the worst case a new forest planted on land which previously had high solar reflectance (notably land typically covered in snow) might actually contribute to warming.

In conclusion, while forests play an important role in promoting local climate stability and protecting against weather extremes in all locations, albedo effects mitigate against the carbon storage benefits of forests in more northerly locations (Lawrence et al., 2022). Considering the net effects on climate, forest loss from the tropics to between 30 and 40 degrees north very clearly contribute to global warming. However, 'Beyond 50_N large scale deforestation leads to a net global cooling due to the dominance of biophysical processes (particularly increased albedo) over warming from CO_2 released' (ibid., p. 1). Within the UK, for example, where there is little snow, and snowfall is declining (Brown, 2019; Morison & Matthews, 2016), research suggests that the impact on albedo is unlikely to negate the cooling effect of carbon sequestration (Jones et al., 2015; Mykleby et al., 2017).

Relative to agricultural land use, other effects of afforestation on climate in temperate areas (such as the UK) include increased emissions of volatile organic compounds (VOC) (Ashworth et al., 2012; Rosenkranz et al., 2015; Sharkey et al., 2008) and increased evapotranspiration (Nisbet, 2005). Both phenomena are associated with cloud formation, which increases albedo (Scott et al., 2014; Spracklen et al., 2008). In fact, the elevation in evapotranspiration caused by afforestation tends to warm boreal zones (through decreased albedo as described above) but cool tropical areas (through increased cloud cover and therefore albedo) (Betts, 1999; Duveiller et al., 2020). These effects are not well quantified but are thought to be relatively small in temperate zones (Matthews, 2020).

A further, potentially substantial, consequence of afforestation concerns the net change in emissions caused by any activities displaced from the planted site. The level of afforestation envisaged to attain the UKs 2050 net zero goal is substantial and likely to involve planting on considerable areas of farmland. This may result in changes in emissions elsewhere ('carbon leakage') either due to displaced domestic production or because of increased agricultural imports from countries with different agricultural emission profiles. If UK tree planting lowers domestic food production and this results in greater imports from higher carbon food producers (e.g. if domestic beef is replaced by imports produced through destruction of rainforests), then the resultant carbon leakage must be set against the gains generated by UK tree planting. However, carbon leakage relationships remain poorly quantified, vary with ongoing changes in diet and are a focus of ongoing research (Faccioli et al., 2022; Golub et al., 2013; Peña-Lévano et al., 2019; Pfaff & Robalino, 2017).

3.2.3 | Biodiversity

Global studies of biodiversity paint a consistent picture of declines associated with human activity, reporting extinction rates 100–1000 times above natural levels (Proença & Pereira, 2017) with remaining populations becoming less diverse (Newbold et al., 2015; Thomas, 2013) and more than halving in size over the last 50 years (McRae et al., 2017). Numerous international initiatives have failed to address the urgent need to 'bend the curve' on biodiversity loss (Lawton et al., 2010; Tittensor et al., 2014; Mace et al., 2018). As the world's first industrialised country and an early adopter of intensive agricultural techniques, some commentators argue that the 'UK has "led the world" in destroying the natural environment' (Davis, 2020). Certainly, biodiversity loss is a major challenge for the UK with over 40% of species in decline since the 1970s and 26% of mammals now 'at a very real risk of becoming extinct' (Hayhow et al., 2019). This loss applies even within UK woodland with declines across multiple species groups including woodland plants, birds and butterflies (Fox et al., 2015; Hewson & Noble, 2009; Kirby et al., 2005; Smart et al., 2014).

Biodiversity loss is clearly of existential importance to the species concerned, and for many people the non-use values associated with this loss are sufficient reason to prevent further loss. Yet, biodiversity provides more than just existence values. Stretching right across the panoply of values from highly visible foci of (or enhancements to) recreational experiences to inapparent yet potentially vast sources of genetic diversity, biodiversity contributes to the provision of many of the key ecosystem services which support human well-being, including the primary production of food, ecosystem functioning and climate regulation (FAO, 2019; Guo et al., 2010; Isbell et al., 2017; O'Connor & Crowe, 2005; Tilman et al., 2014). As such, at extreme tipping points, biodiversity loss would become a sustainability issue even when viewed from a purely anthropocentric perspective. The challenge here is to detect those tipping points in advance of breeching them (Dudney & Suding, 2020) and then use these to set thresholds for the no-loss or net-gain rules discussed previously. The extreme levels of biodiversity loss in the UK (House of Commons Environmental Audit Committee, 2021) suggest a pragmatic rule of thumb and Government policy is not only to prevent further loss but '*be the first generation to leave the environment in a better* state than we inherited it' (HM Government, 2018, p. 2).

Woodland clearly has the potential to contribute towards bending the curve on biodiversity loss. Globally forests support about 70% of terrestrial biodiversity (IUCN, 2017) while forest loss and degradation are a major cause of biodiversity decline (Betts et al., 2017). Given the 750,000 ha expansion of woodland planned for the UK to 2050 the potential clearly exists to deliver not only the carbon storage which provides the immediate impetus for that land use change, but also biodiversity gains; indeed, the substantial conservation potential of these new forests (Brockerhoff et al., 2017) will have to be realised if the UK government is to deliver on its environmental improvement commitments.

As ever the interacting issues of species, space and time, along with a further factor, management, will be crucial here. A sizeable literature exists examining the speed at which biodiversity responds to changes in land use and woodland creation (see review by Burton et al., 2018), but broadly changes tend towards some new equilibrium over time with the speed of transition varying by location and characteristics of the woodland and the generalist or specialist nature of the wildlife species under consideration. For example, in a study of biodiversity response to reforestation within the Scottish Highlands, Warner et al. (2021) report that plant, beetles and bird assemblages transition towards those found in established habitats within the first 30 years of reforestation with native species.

Substantial environmental variation across the UK results in significant locational effects in the biodiversity response to woodland creation. For example, looking at bird species response to broadleaf planting across the country, Bateman et al. (2014) show this varying from strongly positive to insignificant and even negative relative to a current baseline depending on location, a result which reflects both the differing character of areas and their current levels of biodiversity.

The influence of species (including mixtures of conifers and broadleaves) and management interacts strongly in determining the biodiversity response to woodland creation. The literature generally supports the planting of native species as the best approach to boosting biodiversity (Betts et al., 2021; Calviño-Cancela et al., 2012; Cossalter & Pye-Smith, 2003; Di Sacco et al., 2021; Warner et al., 2021). For the UK, this would generally imply a preference for native broadleaves such as oak, over non-native conifers, such as Sitka spruce. In a comparison of conifer, broadleaf and mixed woodlands (from Sweden), Felton et al. (2010, 2021) report significantly higher bird species richness, evenness and abundance in stands with a higher proportion of broadleaved trees. However, this does not imply that conifer woodland, even the large majority, which is planted with non-native species, is of no importance for biodiversity (Confor, 2020; Lindenmayer & Franklin, 2002). In a comparison of non-native conifer (Sitka and Norway spruce) with native conifer (Scots pine) and broadleaf (oak) species conducted across northern and southern Britain, Quine and Humphrey (2010) compared species richness of a wide range of different taxonomic groups (lichens, bryophytes, fungi, vascular plants, invertebrates and songbirds). While several individual differences were observed, no significant difference was found in the overall species richness between the non-native and native stands.

One of the problems dogging the fraught literature on tree species and biodiversity is a tendency to draw somewhat extreme comparisons between idealised and demonised states, for example between monoculture plantations of exotic conifers and well-established native broadleaf woodlands. Instead, Brockerhoff et al. (2008) stress the importance of clarity over the choice of comparison. The conversion of intensively farmed agricultural land to native broadleaf woodland almost always leads to increases in biodiversity, while the replacement of broadleaved (and especially old growth) woodland with (typically non-native) conifers leads to biodiversity losses. However, conversions from intensive agricultural land to even non-native conifer can contribute to biodiversity gains; gains which can be enhanced using mixtures of species. A key factor in realising any gain, however, concerns the way in which those woodlands are managed.

Management is a crucial factor in the realisation of biodiversity gains from woodland (Barlow et al., 2007; Betts et al., 2021; Brockerhoff et al., 2008; Swanson et al., 2011). Intensively managed forests, regardless of species composition, may have relatively low conservation value. The forest manager has a wide range of levers with which to improve conservation quality. In addition to a move away from homogeneous age monocultures (especially of non-native species; Felton et al., 2010), positive management initiatives include longer rotation lengths with mixed aged stands retaining old growth features, especially standing and fallen deadwood, encouraging heterogeneity in canopy layer structure at scales relevant to wildlife, using thinning regimes to increase light penetration, and avoiding the clearance of understory along with dead wood and green tree retention (Calladine et al., 2017; Felton et al., 2016; Franklin et al., 2018; Franklin & Johnson, 2012; Lindenmayer et al., 2012; MacLean et al., 2009; Pommerening & Murphy, 2004; Puettmann et al., 2015). Together, these strategies aim to recreate the natural disturbance vegetative structure of 'ecological forestry' (Betts et al., 2021). Such approaches can contribute significantly to the enhancement of biodiversity. However, the added complexity of such management and their impact upon yields imply costs to forest operators compared to homogeneous plantations (Newton & Cole, 2015). As a partial response to this, Betts et al. (2021) advocate a land sharing–sparing or 'Triad' zoning approach 'where the landscape is divided into three sorts of management (reserve, ecological/extensive management and intensive plantation)'. This takes advantage of the biodiversity advantages of dedicated 'spared' areas devoid of forest production operations, with commercial support provided by intensive timber production in other areas.

Alongside a focus on the ways in which afforestation might contribute to biodiversity, there is a growing realisation that trees themselves need to be a focus of conservation concern. The Global Tree Assessment (BGCI, 2021) compiled risk information on the nearly 60,000 tree species worldwide noting that 30% of these species are threatened with extinction. Some 142 species have already vanished from the wild, while 442 are highly endangered with fewer than 50 individual trees remaining. The principal threats to trees globally are forest clearance for crops (impacting 29% of at-risk species), logging (27%), clearance for livestock grazing or farming (14%), clearance for development (13%) and fire (13%) with both the spread of invasive pests and diseases and climate change being increasing threats (BGCI, 2021; Briggs, 2021).

While countries such as Madagascar, with nearly 3000 endemic tree species, hold the largest number of at-risk species (over 1800), even in the UK the threat to tree biodiversity is highly significant and a focus of public concern (Briggs, 2021; The Guardian, 2021). Of the 86 tree species present in the UK, 34 are endemic and 35 (41%) are considered as threatened with extinction (BGCI, 2021) indicating that trees need to be considered as part of the biodiversity crisis and worthy of inclusion within the objectives of afforestation planning (Messier et al., 2021).

Biodiversity is clearly valuable; however, the challenge of placing a monetary estimate on key aspects of that value (in particular, non-use and ecosystem function values) is extremely challenging. This situation may change and certainly the scientific knowledge upon which any robust valuation needs to rely is the subject of ongoing research. In the meantime, we advocate the definition and application of net-gain rules for biodiversity within cost–benefit analyses of decisions. Projects which deliver adequate net gains in biodiversity (judged by ecological criteria) may be facilitated by this approach (the compensation being physical replacement of biodiversity lost). The costs of providing such compensation provide evidence to allow the estimation of a marginal compensation cost curve for biodiversity (similar to the marginal abatement cost curves estimated for greenhouse gas emissions; Moran et al., 2011; Kesicki & Ekins, 2012). Those that are unable to deliver the level of biodiversity net gain necessary to satisfy adequate compensation rules are prevented from proceeding. In this manner, the issue of biodiversity can be brought within the remit of economic decision making (Bateman & Mace, 2020), although this raises several challenges in the judgement of adequate biodiversity compensation. For example, does the planting of a greater number of trees now compensate for the destruction of ancient woodland, especially given the considerable time lags involved?

3.2.4 | Recreation, physical and mental health

Alongside carbon storage and the provision of habitats, open access recreation ranks as one of the major public benefits of woodland. While the lockdown constraints of the Covid-19 pandemic impinged significantly upon forest recreation, prior to this over 600 million recreational visits were made to public woodlands in the UK annually (Forestry Commission, 2018) which, at roughly 9 visits per person per year, make these one of the top recreational attractions in the country. Superb data resources such as the Monitor of Engagement with the Natural Environment (MENE) survey (Natural England, 2017) have permitted the development of advanced online decision support tools such as the Outdoor Recreation Valuation (ORVal) tool (https://www.leep.exeter.ac.uk/orval; Day & Smith, 2018). This allows decision makers to interrogate data, understand the factors determining recreation patterns and explore the effect of altering the distribution of woodland recreational sites. ORVal also employs revealed preference methods to estimate economic valuations of recreation behaviour (Freeman III et al., 2014). This examines the range of recreational options available to a population (including visits to woodland, alternative attractions or not making visits), the characteristics and qualities of those sites (ranging from physical attributes such as the extent to which they are wooded, landscape features, the presence of wild species), how they vary across locations (in terms of the expenditure, travel time and other disutility which each option entails) and how that population itself varies (e.g. in terms of available income, transport options). By considering these multiple real-world determinants together the method reveals the trade-off between the costs of trips and the number of trips taken, thereby valuing the latter while taking full account of all the factors mentioned.

Results from the application of revealed preference tools such as ORVal highlight the magnitude of outdoor recreation values. For example, even during the restrictions imposed by the Covid-19 pandemic, Day et al. (2020) reports total outdoor recreation values of £5.4 billion during the first 3-month lockdown of 2020. The patterns of recreational values are also revealing. As illustrated in the left-hand side (LHS) panel of Figure 6, values are highest in and around population centres rather than in remote albeit often beautiful locations. This is hardly surprising, while individuals might be prepared to pay high costs for occasional trips to the latter locations, the places

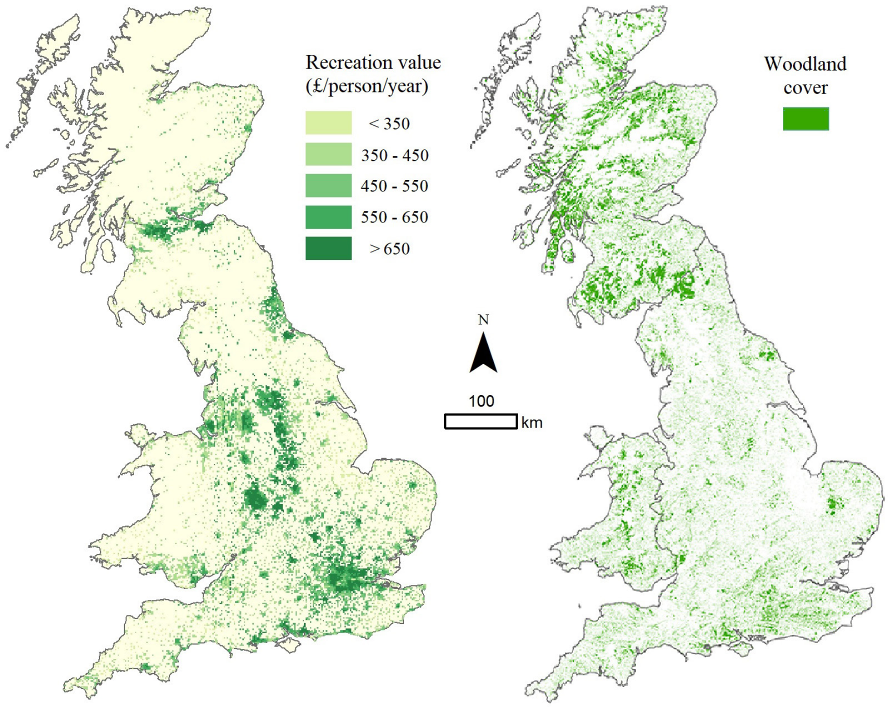

FIGURE 6 Contrasting (LHS) the geographical distribution of annual welfare benefits from access to outdoor recreation opportunities with (RHS) woodland cover in Great Britain.
Source: LHS: Bateman et al. (2014). RHS: ONS (2020b). Contains Forestry Commission information licensed under the Open Government Licence v3.0. Contains OS data Crown copyright 2020. RHS map graphic created by ONS Geography

they visit most frequently and which, across the year, deliver the highest welfare gains, are typically closer to hand where travel and time costs are lower. The contrast with the current availability of woodlands, shown in the right-hand side (RHS) panel of the figure is stark; the current distribution of woodlands and hence supply of their recreational experiences, is almost the opposite of the pattern of recreational demand. Forests have been banished to remote uplands, inaccessible to most of the population. This becomes particularly obvious when we consider the geographical scale of these maps which reveals relatively few major woodlands within easy access of major urban centres.

The simple comparison provided by Figure 6 yields a clear policy guide for the siting of future woodland recreation opportunities: woodlands sited near to or within population centres will yield the highest recreational values. In the UK, this has been reflected in various initiatives (Defra and Forestry Commission England, 2013) but this is yet to be reflected on the ground.

Alongside recreational experiences, access to woodlands provides visitors with both physical and mental health benefits. From an analytic perspective, the value of these benefits depends not only on their magnitude, but also on the extent to which they are net gains as opposed to transfers from other activities. So if the opening of a new woodland site merely shifts recreational activities from one location to another the net gain will be significantly lower than the apparent benefit value of that new site. There is also a causality issue. Visitors to woodland sites may be healthier than non-visitors, but is this a reflection of the benefits offered by the site, or of the people who visit? While the environmental health literature is large, we are unaware of any study which jointly controls for these transfer and causality issues. Nevertheless, the extant literature suggests that, within both rural and urban settings, engagement with trees, woodlands and the wider natural environment does generate significant physical and mental health benefits (Bell & Thompson, 2014; Cox et al., 2017; Karjalainen et al., 2010; O'Brien & Morris, 2014; Saraev et al., 2020; TDAG, 2021; Townsend, 2006). Evidence suggests that such environments might offer unique elements of support for better mental health (Mitchell, 2013). This reinforces the case for ensuring that sufficient weight is placed on the accessibility of woodlands to people; that weight being determined by the benefit value this will generate.

3.2.5 | The water environment

In general, forests play a positive role in the provision of higher quality water supplies as they usually have lower inputs of nutrients and pesticides than agriculture, reduce erosion and can to some extent reduce pollutants loads (Calder et al., 2007). Set against this, forests have long been associated with the acidification of waterways (Battarbee et al., 1988; Nisbet, 1990). While certain trees can raise the acidity of the soils in which they root (Neina, 2019; Turpault et al., 2007), the acidification of waterways is principally a reflection of air pollution. Trees 'scavenge' acid air pollution which would otherwise be more widely dispersed

(UKCEH, 2021). As emission control policies have reduced air pollution, the surface water acidification associated with forests has declined (Nisbet & Evans, 2014).

Water availability has a major role in the growth and flourishing of forests, and forests themselves influence the water cycle. This relationship is in major part a function of forest size (Sheil, 2018). While the higher evaporation rates of forests mean that they use more water than non-irrigated agriculture (Calder et al., 2007; Nisbet et al., 2011), major forests such as the Amazon recycle rainfall and provide agriculture with a buffer against drought (Staal et al., 2018).

Even the modest scale of British woodlands can influence local climates albeit to a relatively small extent (Norris et al., 2012). The UK's combination of high rainfall and large population density means that the relatively low surface runoff, groundwater recharge and water yield characteristics of forests have made them a focus for 'nature based solutions' to flood risk (Hartmann et al., 2019). In particular, the 'hydraulic roughness' (the combination of factors such as tree trunks, interception loss, higher water infiltration rates, soil surface roughness, forest litter and dead wood typifying the vegetative structure of the 'ecological forestry' described previously) may slow water transport speeds and desynchronise flood flows (Calder et al., 2007; Nisbet, 2020). That said, while forests can mitigate smaller, local flooding they have limited capacity to influence either extreme floods or those at the large catchment scale (ibid.).

Methods for valuing changes in water quality, quantity and flood risk changes are well established. The value of reducing flood risk is typically assessed by looking at avoided damages (Penning-Rowsell et al., 2014). Changes in water availability can be valued through a variety of approaches, for example by looking at consequent changes in economic output (OECD, 2016; UN, 2012). Within a developed country such as the UK, potable water quality is typically tightly regulated so that changes usually only apply to surface water quality. Here the ease of payment enforcement through household water bills has led to common application of stated preference techniques (Bateman et al., 2002; Day et al., 2012; Metcalfe et al., 2012).

3.2.6 | Urban benefits

Alongside their recreational, physical, and mental health, aesthetic, decarbonisation, flood risk reduction and biodiversity qualities, trees can provide a range of further benefits for urban populations (TDAG, 2021). Given that urban locations are where most of the world's population live, this has been an active area for tree research with the development of a number of useful decision support systems such as the USDA Forest Service i-Tree software freely available at https://www.itreetools.org.

In a warming world, trees generate a significant counterbalancing effect (Akbari et al., 2001) being responsible for almost 2°C of cooling on average in urban areas worldwide (Loughner et al., 2012; McDonald et al., 2016) and with the potential to generate even greater temperature reductions in cities designed with tree cooling in mind (Turner-Skoff & Cavender, 2019; Zhou et al., 2019). This cooling generates a wide range of benefits including reductions in energy use, emissions and a range of heat-related illnesses.

The effect of trees on the chemistry of the air, and hence health, is complex. Trees can absorb particulates (such as those emitted by vehicles) and inorganic airborne nitrogen molecules (such as ammonia emitted by intensive poultry units) providing a significant barrier to air pollution, though as ever the type and positioning of trees determines the benefits generated (Barwise & Kumar, 2020; Hewitt et al., 2020; Lockwood et al., 2008; UKCEH, 2021). The scale of benefit is generally thought to be substantial. For example, examining the mid-sized US city of Portland, Oregon, Rao et al. (2014) show that urban trees reduce asthma and respiratory disease incidents by tens of thousands each year. Using highly conservative methods focussing just on the immediate costs of these cases suggests a benefit value of $7 million USD annually. Repeating this across the country and extending the analysis to the wider health impacts of trees (including the benefits of recreation discussed previously) suggests that this is one of the major benefits of trees and woodland.

Certain trees, notably eucalyptus, oaks, maples, poplars and willows can emit significant levels of volatile organic compounds (VOC) which, in turn, can contribute to the formation of ozone, carbon monoxide and aerosol particles, which are harmful to breathe but climate-cooling (Geron et al., 1994; Nowak et al., 2002). However, the net effect on VOCs is complicated by the fact that removal of trees in areas where their cooling properties are important can increase VOC emission from anthropogenic sources (Cardelino & Chameides, 1990; Nowak, 2002).

Several studies have examined the acoustic properties of trees as a means of reducing urban noise (Espenido et al., 2018; Van Renterghem et al., 2015; Van Renterghem & Botteldooren, 2008). Valuation of these benefits can in principle be readily undertaken using the hedonic property price method (which reveals the value of visual amenity reflected in the price of houses; Freeman III et al., 2014) and indeed this approach has been applied to both the valuation of noise reductions (e.g. Day et al., 2007) and the proximity of trees (e.g. Netusil et al., 2010).

3.2.7 | Other cultural services

Building on the seminal work of the Millennium Ecosystem Assessment (2005) and recent extensions (Daniel et al., 2012), environmental cultural services can be pragmatically classified into (i) those services which are challenging to quantify let alone value, such as social, spiritual, heritage and education services and (ii) those for which assessment and valuation methods are more developed, including recreation, tourism and landscape aesthetics. All these cultural services are positively related to perceived environmental quality. Therefore, a straightforward guideline with respect to the former, less analytically tractable group, is to adapt the biodiversity no-loss approach and ensure that change does not alter sites in ways which would be seen as environmentally degrading. The strong support that exists for woodland expansion (Forest Research, 2021) suggests that this goal should be readily attainable provided that the type of woodland created is viewed positively.

So what type of woodland do people see as environmentally (and hence culturally) preferable? A common conception is that where native broadleaves are perceived as a significant element of woodland creation such woodlands are regarded as positive enhancements to the environment. While the research literature broadly supports this view (Lee, 2001), it also suggests that species is far from the only important issue. Multiple factors determine preferences including the extent to which forests are diverse or highly structured, the size and age of trees, the degree of management, the availability of recreational facilities and prior conceptions of naturalness which can vary between locations (Edwards et al., 2010; Häfner et al., 2018; Jensen, 1993; Kellomäki & Savolainen, 1984; Oosthoek, 2013). Decision makers perceptions of public preferences are not always accurate, for example Jensen (1993) reports that policy makers overestimated preferences for natural, less managed woodlands. Given the lack of valuation evidence regarding some cultural services, a better understanding of preferences would be advantageous.

Turning to consider group (ii) above, these more readily quantifiable benefits have been the focus of extensive research. de Groot et al. (2010) define landscape aesthetics as the 'appreciation of natural scenery' and this has been the focus of a considerable number of economic valuation studies. While early studies used survey based stated preference methods (Willis & Garrod, 1993), these have been replaced with revealed preference analyses, typically using the hedonic pricing method to examine the premiums that house purchasers are prepared to pay for amenity views (Belcher & Chisholm, 2018; Cavailhès et al., 2009; Garrod, 1994; Garrod & Willis, 1992a, 1992b; Powe et al., 1997; Sander et al., 2010; Schläpfer et al., 2015; Tyrväinen, 1997). These illustrate that woodland landscapes generate positive values although this is less true of conifer plantations, a result which is supported by stakeholder perception studies (Dhubháin et al., 2009).

3.3 | Woodland costs

As discussed previously, the context determines whether woodland creation is beneficial or costly; for example, landscape impacts may be positive or negative depending on the type of forestry and what it replaces; a woodland generally sequesters carbon but placed in the wrong location it will trigger net emissions; the biodiversity benefits of woods can be highly significant but in some areas, planting could destroy valuable habitats. However, we start this section with an item which is, in most cases a cost, the opportunity cost of foregone agricultural production.

3.3.1 | Agricultural output

The scale of greenhouse gas removal necessary to deliver net zero cannot be achieved without land use change (Climate Change Committee, 2020; Roe et al., 2019). Within the UK, this will require sustained woodland creation on a scale not seen since the early years of the Forestry Commission. It is inevitable that some of the three-quarters of a million hectares of land needed to provide the necessary greenhouse gas removal via forestry will have to come from agriculture and foregone food production constitutes a major opportunity cost of woodland creation. This trade-off cannot be avoided (although it can be very significantly reduced). However, when viewed from a purely market priced, private goods, commercial perspective, two major factors mitigate heavily in favour of farming in the comparison with woodland.

The first of these factors is discounting and the need for annual income. Commercial forestry is a protracted business. Even the most rapid rotations take around 40 years to complete. While the felling benefits produced can be very substantial, from the vantage point of the land use decision, they are long delayed. The process of discounting reflects the way in which decision makers relate future benefits and costs to present-day values and the fact is that, while ground preparation and planting are immediate costs, the revenues generated by felling are long delayed.

Once discounted the net present value of forestry is generally lower than that of agriculture in all but the most marginal areas (Bateman et al., 2014; Bradfer-Lawrence et al., 2021) and in many locations this difference is substantial. The difference in the market value of outputs can then be set against the considerable public good benefits delivered by woodlands. Unless they are incentivised by subsidies, public goods will be of secondary concern to many private sector decision makers. This problem becomes even more pressing where there is a need for annual income from the land. Even in locations where discounting does not reduce future woodland values to less than agricultural returns, this annual income requirement can still rule out forestry as a viable option. Governments have reacted to this issue by offering woodland grants and annual subsidies—however, this leads us to a second problem.

The subsidy system has always favoured agriculture over forestry, and even in the present climate crisis this remains a factor. In the 2020 Budget, the Chancellor of the Exchequer announced the creation of the UK Nature for Climate Fund 2020–2025 worth £640 m over that 5-year period (0.1% of the public investment announced in that budget). In the following year, the England Trees Action Plan announced that £500 m of that fund would be spent on tree planting to 2024 (H.M. Government, 2021). This is a substantial sum. However, remembering that woodlands must compete with agriculture for land, the £125 m p.a. this represents equates to just over 3% of the £3 billion paid in agricultural subsidies each year. This uneven distribution of public funding has been a constant challenge to forestry. The solution could be in the reforms to the subsidy system.

3.3.2 | Subsidies and other incentives

From a public cost–benefit perspective, government subsidies are merely transfers across society, a zero-sum game. However, they can induce major change in the shape of an economy and the range and value of both public and private goods they generate. It is this change which justifies their use.

The past seven decades of UK land use policy have strongly favoured farming over forestry. However, the UK's exit from the EU also meant an exit from the constraints of the Common Agricultural Policy (CAP) with its focus upon subsidising food production. The origins of this strategy lie in the second world war (WWII) when Europe suffered extreme food shortages. However, the massive over-production almost inevitably caused by early CAP policies such as price intervention turned Europe into a major exporter of food sold at artificially low prices to address mounting stocks (Matthews, 2008). Recent reforms have tempered but not eliminated the over-production caused by public subsidy of a private good (Blanco, 2018).

In the run up to formally leaving the EU, advice to the Government suggested that UK agricultural policy should take a radical departure from the food production focus of the CAP (Bateman & Balmford, 2018; Natural Capital Committee, 2017); a switch to the principle of Public Money for Public Goods (PMPG). This builds on long-standing official recognition that public spending should not focus on private good production activities which can be more efficiently provided by the market, but rather support those public goods which the market will not provide without regulation or incentive support (H.M. Treasury, 2007, 2013). Food is a private good, which in the UK is very efficiently produced and bought and sold in markets. The popular argument that agricultural subsidies secure lower food prices and social access to food is dubious. While society may well have a public interest in ensuring that the poorest have access to food, delivering this through subsidising food production, which will then be sold to the highest bidder, is massively inefficient.

Similarly, the use of public funds to promote import substitutions runs contrary to the comparative advantage argument (see Section 3.2.1). As with timber there may be a case for having some modest level of domestic production as a bulwark against supply chain problems and to ensure food security. Although abandoned now, for several decades after WWII the UK effectively guarded against such supply side problems by maintaining publicly funded stocks of certain basic foodstuffs, and indeed some countries, such as Germany, still maintain substantial public food stocks (Folkers, 2019). However, such interventions would not justify the level of subsidy under the CAP which amounted to nearly half the value of agricultural production (and more than half the EU budget) at its peak (Bateman & Balmford, 2018).

This does not imply that farm subsidies should end or even be reduced; but what is paid for needs radical alteration. Agriculture has the potential to deliver massive levels of public goods (Natural Capital Committee, 2017). The 2020 Agriculture Act (UK Parliament, 2020) formally placed the PMPG principle at the heart of UK farm policy, setting out a timetable to phase out the traditional focus on production subsidies well before the end of this decade and replacing it with a focus on the provision of public goods.

The substantial subsidies paid to agriculture provide an obvious solution to the challenge of funding and finding the land for the large-scale expansion of woodland required for the UK to satisfy its climate change commitments. The valuable benefits which farming and other land can provide through conversion to woodland will not be provided by the market of its own volition. The PMPG subsidy focus ushered in by the UK Agriculture Act is a major step forward in this respect and provides the opportunity to convert UK farming to a major supplier of both private and public goods, but it will need supporting measures. For example, for many years, a farmer who planted trees on their land would find that area removed from eligibility for agricultural subsidies. This loss of payment support compounded the substantial loss in production income associated with such land conversion. While recent policy changes are opening routes to address this challenge, such as capital and annual maintenance grants payable under the Countryside Stewardship scheme (Rural Payments Agency, 2022) and support for agroforestry schemes (Rural Payments Agency, 2020; see SM), this problem is far from completely eradicated and the memory of such disincentives will take some time to overturn. As part of this process it may be that the common requirement that tree felling should always be followed by replanting may have to be reviewed in some cases. This requirement makes conversion of land into woodland an irreversible process which both limits future decision flexibility and may reduce the capital value of farmland. Ironically, it may be that allowing for the possibility of trees being cut down, and so retaining some flexibility in future land use, might be necessary to ensure that more are planted. Furthermore, roughly one-third of UK agricultural land is farmed under tenancy agreements (Barclay, 2010) where increases in grant payments can simply translate into increases in rental costs removing the incentive properties of those subsidies.

This is far from an argument for turning all UK farmland into forestry. Some 77% of the UK is currently under agriculture (Defra, 2021) and the full extent of proposed woodland creation to 2050 would represent just 3% of the country, taking overall woodland coverage to 16%. This change would not alter the dominance of farming over the landscape but has the potential to turn land into a powerhouse of public good creation, addressing the net zero challenge and providing the ‘plus’ in net zero plus.

4 | SUMMARY AND NECESSARY IMPROVEMENTS TO DECISION MAKING

As this review shows, woodland creation has the potential to deliver a huge variety of public and private benefits and costs almost all of which vary according to the type of woodland and its location. Provided the ‘right place for the right tree’ natural capital principles set out above are followed, then ‘net zero plus’ woodlands can provide a truly exceptional range of public and private benefits including:

- Climate change mitigation through the sequestration and storage of carbon and the substitution of harvested wood for more emission-intensive products;
- Conservation and enhancement of biodiversity;

- Recreational access and improvements to physical and mental health;
- Timber and other wood products;
- Water quality improvement;
- Water quantity regulation and flood risk reduction;
- Improved soil health including reductions in erosion and the sedimentation of waterways;
- Overall improvements in air quality including reductions of particulates and ammonia concentration (although adjustment for the VOCs they can produce is required);
- Urban cooling;
- The provision of amenity views;
- Locations for privately funded carbon sequestration projects, for example, as in the Woodland Carbon Code (2021);
- Locations for meaningful net gains in biodiversity arising from offsetting the impacts of development.

Set against all the above we have to also consider the costs which woodland creation can incur, including:

- Foregone agricultural output.
- The potential for carbon leakage from induced food imports.
- Any negative albedo effect of forests, increasing global warming.
- The typical tree nursery, planting, protection, maintenance and felling costs associated with forestry.

Within each of these benefits and costs, the site specificity can be highly pronounced, changes affect multiple benefits and costs simultaneously but in ways which are often not simply correlated. Few if any of these impacts can be routinely ignored or readily simplified without risking serious errors in decision making.

This complexity leads us to three immediate conclusions:

- First, the degree of complexity means that we must bring scientific data and quantification into the decision-making process. Simple rules will not be adequate for understanding the complex system response to land use change from agriculture to woodland.
- Second, the diversity of values triggered by these changes, both private commercial revenues and expenditures and wider social benefits and costs, means that we also must bring socioeconomic data into that decision process.
- This of course leads us to a third conclusion: that any robust decision support system will have to integrate both the science and socioeconomic aspects of alternative options for action and their consequences.

The SEE principles of the natural capital approach provide the conceptual framework for this integration, but this also needs the practical development of compatible decision support tools. Figure 7 provides a visual illustration of the complexity of issues which need to be taken into consideration to robustly undertake the land use change needed to deliver net zero plus woodlands.

Starting at the upper left of Figure 7, we see (a) the Spatial, Temporal, Economic and Policy (STEP) drivers of land use change which determine and alter (b) land use, its domestic outputs and (c) consequent international imports. Taking into account a variety of important complexities (such as (d) land ownership and tenure; (a) climate change; changes in risks such as (e) pests, disease and (f) wildfire; the greenhouse gas removal (GGR) and emission attributes of (b) agriculture and (g) other land uses), the policy element of (a) the STEP drivers can be altered in a variety of ways, each producing (h) an alternative land use future which, in turn, generate (i) the diversity of benefits and costs discussed previously in this paper. All these benefits and costs are brought together in (j) the Decision Support System (DSS) which integrates all elements together and incorporates information about the uncertainty in values which affects all data and derived models.

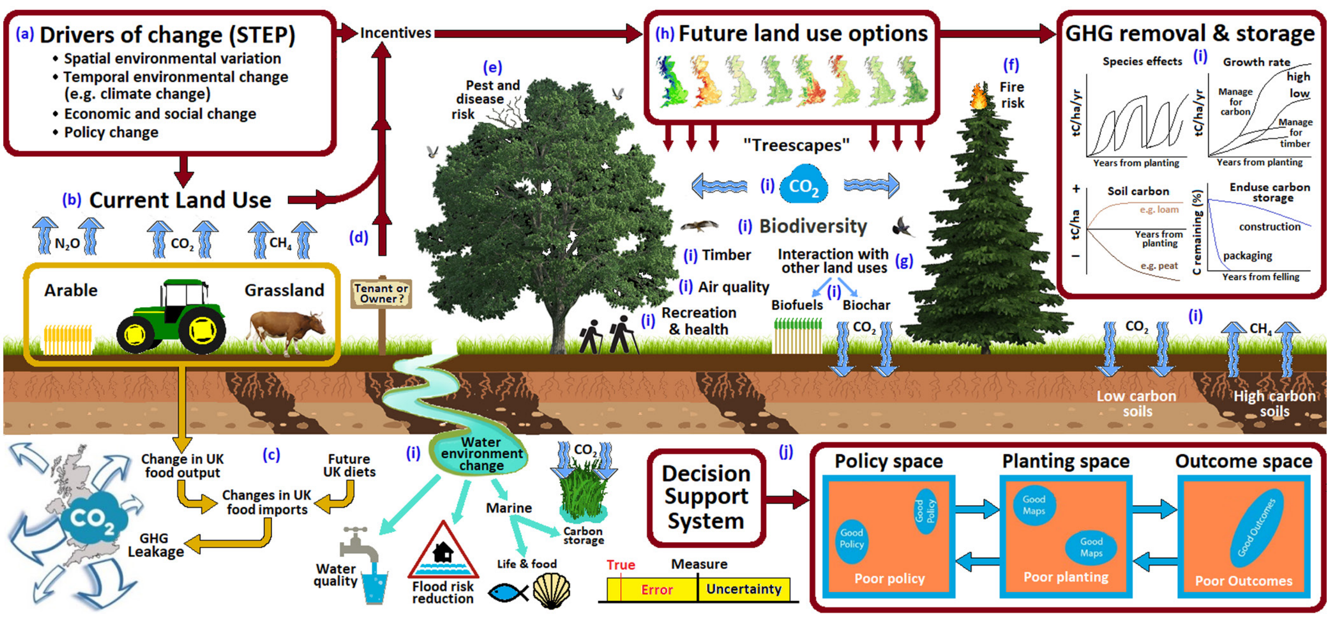

FIGURE 7 The dimensions of a robust woodland creation decision support system (DSS)

Each of the benefits and costs described in Figure 7 is determined through a set of scientific and socioeconomic relationships. The decision support system (DSS) brings these relationships, inputs and outputs, each with its own degree of uncertainty, together within a tool designed to support decision making. This allows the decision maker to examine the consequences of policy changes upon land use and woodland planting and resulting outcomes. Alternatively, the DSS can be driven in reverse. Here decision makers specify desired outcomes for which the DSS determines compatible land use and planting and the policies necessary to achieve this.

The more holistic analysis described throughout this paper and summarised in Figure 7 is within our grasp with many elements contained within existing systems and the remainder having the data necessary for their production. Tools such as the **Integrated Valuation of Ecosystem Services and Tradeoffs** (InVEST) DSS produced by the Natural Capital Project at Stanford University (https://naturalcapitalproject.stanford.edu/software/invest) provide a ready means of integrating diverse natural and social science spatial data onto a ubiquitous grid system which has been used round the world in hundreds of applications. Closer to home and specifically tailored to the UK, the NEV Modelling Suite developed by Day et al. (2020) for the Department for the Environment, Food and Rural Affairs (Defra) provides proof of concept for such tools and their capacity to provide a substantial data-driven input to decision making. Such systems need careful co-design with those who will use them. Despite the absolute necessity of bringing data and evidence into decisions through such DSSs, it is, of course, also necessary to include both stakeholder (Burton et al., 2019; Di Sacco et al., 2021) and wider public consultation.

The costs of extending existing systems to produce a fully working version of a DSS which embraces the diversity of issues described in Figure 7 are trivial compared to those which will have to be incurred to undertake the necessary woodland creation to deliver on the net zero commitment. Even those costs are massively exceeded by the environmental and social benefits of attaining that goal, benefits are which, in turn, dwarfed by the costs of inaction.

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CONFLICT OF INTEREST

Rosemary Hails is a Lead Editor for People and Nature, but was not involved in the peer review and decision-making process. The other authors have no conflicts of interest to declare.

AUTHORS' CONTRIBUTIONS

I.J.B. conceptualised and structured the paper helped by B.H.D. and D.W.; The content of the paper was written by all authors (I.J.B., K.A., A.A., C.B., R.A.B., A.B., R.E.B., F.H.T.C., R.M.C., B.H.D., C.D.-R., S.E., K.G., N.G., B.G., R.H., A.B.H., A.Har., A.Has., M.S.H., T.C.H., A.I., C.F.L., D.J.L., A R.M., M.C.M., J.I.L.M., A.M., C.P.Q., P.S., C.R.T., E.I.V., M.W., D.W. and G.X.). All authors contributed critically to the revision process which was led by I.J.B. and R.H. All authors gave final approval for publication.

DATA AVAILABILITY STATEMENT

This is a review paper and has no original data to archive.

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