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Natural Capital: Quantifying Existing Stocks and Future Potential using a Geospatial Approach

By

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In collaboration with

LUC

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Abstract

Geospatial techniques for quantifying, modelling, and mapping natural capital and ecosystem services have the potential to improve our understanding of the benefits provided by natural assets and identify changes in land use that could increase these benefits. However, questions remain around how such an approach could be implemented in practice. In this thesis, analyses are undertaken across multiple scales to explore how geospatial techniques can be applied to help solve current challenges in land management and planning.

At the local scale, a land cover and benefit transfer methodology is developed and applied for the first time to value current natural capital assets within individual farms in the UK. This work highlights how the land cover product used in the methodology can have a substantial impact on valuations, with differences of up to 58% found at the five farms studied. The magnitude of these differences varies according to the landscape structure of the farm, with higher resolution land cover products incorporating larger amounts of woodland, primarily through inclusion of smaller patches, leading to overall higher valuations.

At the national scale, the creation of new natural capital assets is explored by investigating proposed large-scale afforestation targets in the UK. In the initial part of the study, the feasibility of meeting these targets is investigated in the first national assessment of land available for afforestation, considering a range of physical, environmental, and policy constraints in three hypothetical planting scenarios. This found that while there is sufficient space to meet the afforestation targets in all three scenarios, this would require planting on a large proportion of unconstrained land, which could limit opportunities for spatially targeting woodland creation. The implications of this transformational change in British land cover, and policies that would be required to support this transition, are highlighted.

In the second part of the study, the potential to deliver ecosystem services from afforestation is investigated. Models and spatial analysis are used to quantify the provision of carbon sequestration, recreation, and flood mitigation from potential new woodland across England, identifying targeted locations where new planting could maximise the provision of these three services. The impact of planning afforestation at different spatial scales is explored by identifying priority locations nationally and within smaller planning units such as local authorities. This shows that while spatial targeting within larger spatial units results in the greatest provision of ecosystem services, targeting even within smaller units provides substantially greater benefits than random, untargeted afforestation.

Overall, the thesis develops and applies new geospatial tools for quantifying, modelling and mapping natural capital and ecosystem services. In doing so, it highlights the sensitivity of the techniques to the quality of the input data and the scale of the analysis. The outputs generate detailed insights into the distribution and potential changes in natural capital that can result from land use decisions which provides valuable evidence for directing future policy and practice.

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I gratefully acknowledge the funding received towards this project from my enterprise partner, LUC, and the Centre for Global Eco-Innovation at Lancaster University, part-funded by the European Regional Development Fund, which made this PhD possible. I am grateful also to the multitude of individuals and organisations who have assisted through the provision of data.

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Declaration

I declare that, other than where the contribution of others is specified, this thesis is entirely my own work and has not been submitted for the award of any other degree at this or any other university.

Thomas Burke

Statement of Authorship for Multi-Authored Chapters

Chapter 3: The influence of land cover data on farm-scale valuations of natural capital

Burke, T., Whyatt, J.D., Rowland, C., Blackburn, G.A., Abbatt, J. (2020). The influence of land cover data on farm-scale valuations of natural capital. Ecosystem Services, 42, 101065.

The concept and methodology for this chapter was decided upon through discussions among the authors. I was responsible for obtaining the data used and carrying out the analysis. Having obtained the results of this analysis, I produced the draft text of the manuscript, including tables and figures. I then revised this draft in response to comments and suggestions from co-authors, and following submission of the manuscript, two anonymous reviewers.

Chapter 4: Achieving national scale targets for carbon sequestration through afforestation: Geospatial assessment of feasibility and policy implications

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Chapter 5: Spatial distribution of national-scale afforestation targets for new tree planting: a multiple ecosystem service approach

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The concept and methodology for this chapter was decided upon through discussions among the authors. I was responsible for obtaining the data used and carrying out the analysis. Having obtained the results of this analysis, I produced the draft text of the manuscript, including tables and figures. I then revised this draft in response to comments and suggestions from co-authors.

I confirm that the above information on the authorship of these chapters and the contribution of Thomas Burke is correct.

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1. Introduction

1.1 Background context

Natural capital is defined as the world's stock of natural assets, such as soil, plants and water (Mace et al., 2015; Natural Capital Committee, 2017; Spake et al., 2019). Natural capital provides flows of benefits to humanity, known as ecosystem services (Burkhard and Maes, 2017; Costanza et al., 1997). Examples of these benefits include the provision of fuel, building materials and medicine, climate change regulation through carbon sequestration and storage, and the pollination of crops by insects (Millennium Ecosystem Assessment, 2003). Natural capital, and the ecosystem services it generates, is vital for human welfare, providing essential goods and underpinning the world's economic prosperity (Burkhard and Maes, 2017; Constanza and Daly, 1992). There have been numerous commitments to conserve and enhance natural capital in recent years (Com/2011/0244, 2011; Decision 1386/2013/EU, 2013; Defra, 2018a), but practical steps must be taken to meet these aspirational goals.

Quantification, modelling, and mapping of natural capital and ecosystem services provides a framework within which to explore improvements in land management (Pandeya et al., 2016). Such an approach could both improve our understanding of the benefits provided by natural assets, and identify changes in land use that could affect them. To date, however, work in this area has been largely theoretical. This thesis therefore explores how the quantification and modelling of natural capital and ecosystem services could be applied in practice to improve the environment and benefits that flow to society.

This industry-led PhD was undertaken in collaboration with LUC (Land Use Consultants), an environmental consultancy providing planning, impact assessment, landscape design, ecology, and geospatial services to the public and private sector (https://landuse.co.uk/). This gave the project a strong practical focus and a responsiveness to external developments, and proposed developments of policy, that more purely academic theses are unlikely to have. The project was initiated in 2017, a time of transition for the United Kingdom and its agricultural sector, with the country's impending withdrawal from the European Union and therefore the Common Agricultural Policy (CAP).

The CAP has long been criticised for doing little to improve environmental outcomes (Defra, 2018b; European Court of Auditors, 2017; Matthews, 2013). The development of new agri-environment schemes that reward the provision of ecosystem services and conservation of natural capital assets have therefore been proposed by researchers and organisations including Defra and the UK's Climate Change Committee (Committee on Climate Change, 2018; Defra, 2018b; Hodge, 2017). While the exact form of these schemes remains the subject of debate, the development and implementation of such schemes will inevitably be based on the ability to quantify the natural capital and ecosystem services provided within individual farms (Maes et al., 2017). Studies quantifying natural capital at this scale are, however, rare. Reviewing the literature, Malinga et al. 2015 found that the majority of studies were conducted within 'intermediate extents', ranging from 'municipality' scale (approximately 50 km^2 – 10,000 km^2), to 'national' scale (50,000 km^2 – >1,000,000 km^2), well beyond the average UK farm size of 1 km^2 (Defra et al., 2018). Similarly, Burkhard et al. (2009) notes that much work mapping ecosystem services has focused on global assessments, which are not directly applicable to local decision making. The initial motivation for this project therefore was to address this knowledge gap, using geospatial techniques to quantify natural capital within individual farms and estates.

While this initial work explored how natural capital could be quantified at the farm scale, the thesis also sought to address how natural capital and associated ecosystem services could be improved. In 2018, the UK's Climate Change Committee released a document identifying the need to create some 2.5 million hectares of new woodland by 2100 for the purposes of carbon storage and sequestration (Committee on Climate Change, 2018). This was just one, of a growing number of calls for large-scale afforestation by organisations including learned societies, charities, and public bodies. The afforestation targets offered a timely opportunity to explore how ecosystem service provision could be optimised through changes in land use, particularly through woodland creation. This raises two crucial questions: First, is there sufficient suitable space in the UK to meet planting targets? Secondly, given that woodland can provide a range of ecosystem services, how could new woodland planting be targeted to optimise service provision?

While numerous targets for woodland creation have been proposed, there has been comparatively little discussion about where these trees could be planted. Seeming at first a simple question, in reality there exist numerous constraints on woodland creation in the UK, including physically constraints to tree planting, where planting is restricted due to policy, and where it may be undesirable for environmental reasons.

This thesis therefore presents the first high resolution and comprehensive map of constraints on woodland planting for the whole of the UK. These are explored through three hypothetical planting scenarios to determine whether meeting proposed planting targets is feasible, and what the implications of this large-scale planting could be.

In addition to carbon sequestration, woodlands are known to be an important source of many other ecosystem services (Lake et al., 2020). The degree to which these services are provided depends both on the characteristics of the woodland, which provides them, and its location. The final part of the thesis therefore explores how this proposed new woodland could be sited to optimise not just for carbon sequestration, but also recreation and flood mitigation, resulting in the creation of woodland that could provide multiple benefits to society.

1.2 Aims and objectives

The overall aims of this thesis are to explore:

- The quantification of current natural capital assets, and flows of ecosystem services from them.
- The estimation and optimisation of future ecosystem service flows from the creation of new natural capital assets.

Through these two aims, the thesis explores how these approaches could be applied to help solve current challenges in land management and planning. This is achieved by answering the following key research questions:

- How do we map current natural capital stocks, and value the ecosystem services arising from them?
- Do existing datasets provide an appropriate evidence-base to inform natural capital-based environmental management?
- What are the constraints on increasing ecosystem service provision through the creation of new natural capital assets, such as woodland?
- How does scale influence natural capital mapping and ecosystem service estimation?

1.3 Thesis Organisation and Structure

This thesis comprises of six chapters, plus appendices. Chapter 1 (this chapter) provides a brief introduction to the thesis and an overview of its contents. Chapter 2 introduces the concepts of natural capital and ecosystem services in greater detail, before reviewing the techniques that have been used to quantify, value, and map them. This is followed by a review of current and future opportunities to use natural capital and ecosystem services in land management policy and decision-making, as well as factors that need to be considered when planning for this, including those issues that require further research. Following this review, three analytical chapters are presented, the first two of which have been published (see statement of authorship above for full references), and a third which has been prepared in a style appropriate for submission to a scientific journal. References for each of these have been collated into an overall reference list which is provided at the end of the thesis.

Chapter 3 explores the quantification of natural capital at the local scale. Carried out in collaboration with five land owners and managers from across the UK, this chapter presents the first assessment of natural capital within individual farms through the application of a land cover and benefit transfer approach. This approach has to date largely been confined to global and regional scale studies. The chapter emphasises how the spatial and thematic resolutions of the land cover data used can have a significant impact on valuations, and how the magnitude of this impact varies depending on the characteristics of the landscape being valued.

While chapter 3 focuses on estimating natural capital at the local scale, chapters 4 and 5 explore how this could be enhanced through woodland creation at the national scale. Chapter 4 presents the first high-resolution national assessment of space available for woodland creation in the UK, accounting for a range of physical, environmental and policy constraints in three hypothetical planting scenarios. The spatial distribution of woodland creation that would occur under each scenario, and the implications of such large-scale transformation in land cover, are also discussed, as are changes in land use policy that would be required to support this transition.

Chapter 5 takes this a step further by identifying not just where woodland **could** be created, but where it **should** be created, in order to optimise for the provision of multiple ecosystem services, specifically: carbon sequestration, recreation, and flood mitigation. Models and spatial analysis are used to quantify levels of service provision from potential new woodland across England. Areas where new planting could maximise the overall provision of these three services are then identified. The impact of planning within different spatial units is explored by identifying these locations nationally, and within boundaries such as local authority districts. This illustrates how optimising for woodland creation at different scales, as is the case in the UK, could result in different decisions and hence different levels of service provision.

Finally, chapter 6 highlights the key results, research contributions and implications from each analytical chapter. It synthesises the results of both the national and local scale case studies, presenting a holistic analysis of the role of natural capital in land use planning. It concludes by outlining recommendations for future work to further explore issues surrounding natural capital and ecosystem services in land management, and how policies could be implemented to ensure we can continue to enjoy the benefits provided by nature.

1.4 Declaration

This industry-led PhD was undertaken in collaboration with the environmental consultancy LUC (Land Use Consultants). The initial design and broad research aims of the project were developed by LUC, prior to the PhD commencing. The company initially sought to develop tools and methodologies to quantify natural capital within individual farms. These approaches could then be used to provide baseline data and evidence for the potential development of agri-environment policies, and to work with clients such as landowners to design farm-scale land management plans.

As the project progressed, LUC subsequently sought to develop expertise around the creation of new natural capital assets through afforestation, driven by interest from potential clients looking to plan for the creation of new woodlands within their boundaries, the release of national scale afforestation targets by the UK's Climate

Change Committee, and initial results from the project highlighting the relative importance of woodland as a natural capital asset in UK rural landscapes.

Within these research areas, specific aims, research objectives, and approaches were developed through collaboration between the academic supervision team, LUC, and myself.

2. Literature Review

2.1 Introduction: what are natural capital and ecosystem services?

Natural capital can be defined as the world's stock of natural assets, which yield flows of benefits to humanity (Mace et al., 2015; Natural Capital Committee, 2017; Spake et al., 2019). Generally, capital can be considered as a stock of material or information that exists at a point in time. This can take different forms, both physical, such as machines and buildings, and more intangible, such as the information stored in computers (Costanza et al., 1997). Natural capital is a physical form of capital, and as described above includes natural assets such as soil, plants, water, and the atmosphere. Capital generates either autonomously, or in conjunction with services from other capital stocks, services that enhance the welfare of humans through the transformation of materials (Costanza et al., 1997). In the case of natural capital, these flows of material, energy and information are known as ecosystem services (Burkhard and Maes, 2017; Costanza et al., 1997). Ecosystem services include amongst other things the provision of materials for fuel, building materials and medicine, climate change regulation through the sequestration of carbon by plants and storage by peatlands, and the pollination of crops by insects (Millennium Ecosystem Assessment, 2003).

Ecosystem services may also be defined as the benefits people obtain from ecosystems (United Nations, 2012; Seppelt et al., 2011). An ecosystem is a dynamic complex of plant, animal and microorganism communities and their environment, functioning as a unit (Millennium Ecosystem Assessment, 2003), and can be thought of as a natural capital asset. Ecosystems, ecosystem services and natural capital are therefore strongly and inseparably linked, with ecosystems and natural capital being the basis of ecosystem service flows from nature to society (Burkhard and Maes, 2017).

Natural capital may be non-renewable, such as North Sea oil and gas, where there is only a limited amount, or renewable, where, given appropriate management, nature continues to provide the asset with a potentially infinite yield (Helm, 2015). Natural capital may also be biotic (living, or once living) or abiotic, but in all cases generates ecosystem services. In some cases, these ecosystem services, and the pathways linking them to natural capital stocks, may be very visible and straightforward, while in other cases, the pathways linking natural capital assets and ecosystem services, or the ecosystem services themselves, may be more complex, less visible and poorly understood. For example, forests provide timber, which can be marketed directly. However, forests also hold soils and moisture, and create microclimates, all of which contribute to human welfare in complex, and generally non-marketed ways (Costanza et al., 1997).

Ecosystem services are often categorised according to their function and characteristics. The Millennium Ecosystem Assessment (2005) recognises four categories of ecosystem service: provisioning, regulating, cultural, and supporting (Table 2.1). Provisioning services are the products provided by ecosystems, such as food, fuel, and genetic resources. Regulating services are the benefits obtained from the regulation of environmental processes by these ecosystems, such as climate regulation, or erosion control. Cultural services describe the provision of nonmaterial benefits, such as spiritual enrichment, recreation, or aesthetics. Finally, supporting services are those required for the production of all other ecosystem services, such as the formation of soil and atmospheric oxygen.

Table 2.1: The four categories of ecosystem services as defined by the Millennium Ecosystem Assessment, with examples for each. Adapted from Millennium Ecosystem Assessment (2005).

<table>
<tr><th colspan="2">Categorising Ecosystem Services</th></tr>
<tr><td rowspan="3">Supporting
• Nutrient cycling
• Soil formation
• Primary production
• …</td><td>Provisioning
• Food
• Fresh water
• Wood and fibre
• Fuel
• …</td></tr>
<tr><td>Regulating
• Climate regulation
• Flood regulation
• Disease regulation
• Water purification
• …</td></tr>
<tr><td>Cultural
• Aesthetic
• Spiritual
• Educational
• Recreational
• …</td></tr>
</table>

2.2 Quantifying natural capital and ecosystem services

Natural capital assets, and the flows of ecosystem services they produce, can be quantified both in physical and monetary terms. The process of calculating these stocks and flows for a given ecosystem or region is known as natural capital accounting (Philips, 2017), and is typically carried out in distinct stages (Figure 2.1).

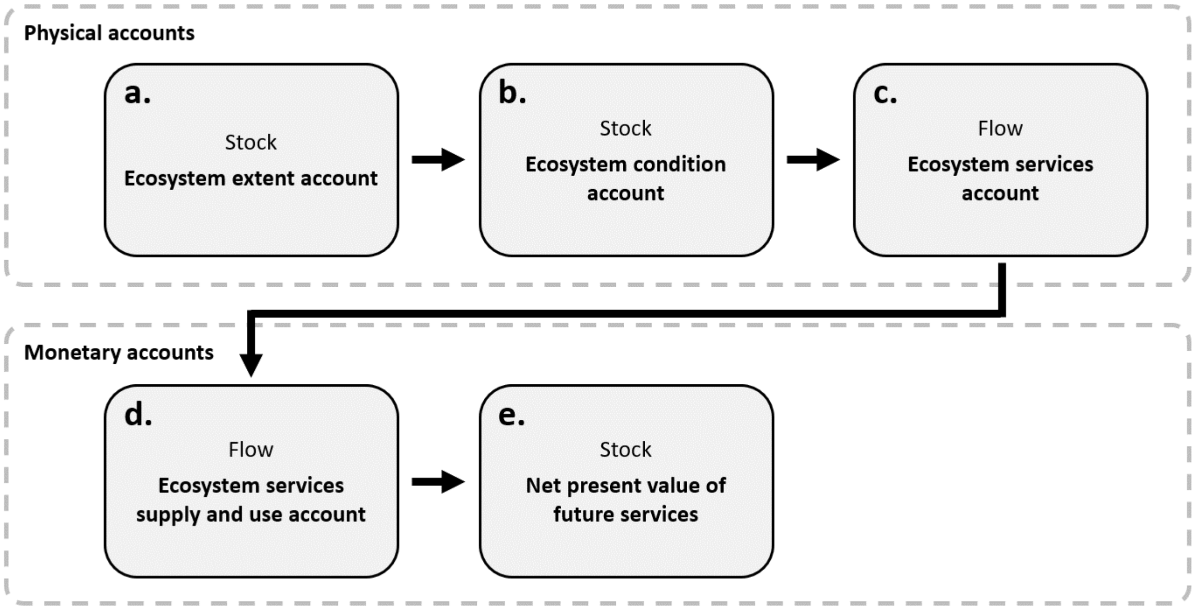


Figure 2.1: Sequence of natural capital accounting, adapted from Philips (2017). Each stage of the accounting process has been labelled a to e.

The first stage of natural capital accounting is the creation of physical stock accounts. These measure the extent or volume of a natural capital asset (Figure 2.1a), such as woodland, and its condition or quality (Figure 2.1b). Next, physical flow accounts record the ecosystem services provided by these natural capital assets, such as tonnes of timber harvested, or the number of recreational visits (Figure 2.1c). Monetary flow accounts then assign a monetary value to these services (Figure 2.1d). Finally, a monetary value can then be assigned to the natural capital asset itself by projecting the supply and use of these services into the future, and discounting them to a present value (Figure 2.1e).

While comprehensive natural capital accounts may seek to complete each of these five stages, others may focus on a specific aspect, for example measuring a single ecosystem service in physical terms only. The results of such work may be presented in numerical terms, similar to a traditional set of accounts. Table 2.2 for example shows the results of an ecosystem services assessment carried out for the Marches region of west England. Here, each of the stages in Figure 2.1 have been applied. The monetary value of several provisioning, cultural, and regulating ecosystem services is presented in a table. This table summarises the results of the assessment, although the values of individual natural capital assets, such as woodlands or meadows, are not given. Instead, they are aggregated together, and the total value of services provided in three geographical areas: Herefordshire, Shropshire, and Telford and Wrekin, is presented.

Alternatively, spatially explicit ecosystem service maps are often used, illustrating how natural capital assets and ecosystem service values vary across space (Syrbe et al., 2017). Figure 2.2 for example illustrates the spatial distribution of ecosystem services in the Yahara watershed, Wisconsin, USA. Unlike Table 2.2, ecosystem service provision in these maps is not presented in monetary terms, but in physical terms, such as tonnes of carbon stored, and as unitless scores, such as potential for forest recreation. This corresponds to stages a to c in Figure 2.1.

In the following sections, typical approaches for carrying out each of the natural capital accounting steps introduced in Figure 2.1 are summarised and critically reviewed.

Table 2.2: An example of an ecosystem service assessment carried out for the Marches region of west England (Hölzinger, 2016). In this example, the five stages shown in Figure 2.1 have been applied, resulting in an estimate of the net present value of future services provided in the study area.

Assessment Area / Assessed Habitat Area		Herefordshire 110,192 ha			Shropshire 171,878 ha			Telford and Wrekin 8,423 ha			TOTAL 290,494 ha		
Ecosystem Service		High	**Central**	Low	High	**Central**	Low	High	**Central**	Low	High	**Central**	Low
Provisioning Services	Wild Food	£106	**£31**	£8	£117	**£34**	£10	£12	**£3**	£1	£234	**£69**	£19
	Ornamental Resources & Non-food Products	£190	**£39**	£11	£208	**£42**	£12	£20	**£4**	£1	£419	**£85**	£24
	Water Supply	£0	**£0**	£0	£1	**£0**	£0	£0	**£0**	£0	£1	**£0**	£0
Cultural Services	Wild Species Diversity	£1,851	**£404**	£164	£3,241	**£647**	£262	£299	**£34**	£14	£5,391	**£1,085**	£440
	Recreation & Aesthetic Values	£464	**£259**	£111	£1,050	**£544**	£240	£495	**£282**	£129	£2,010	**£1,086**	£479
	Health	£1,364	**£852**	£451	£2,329	**£1,536**	£903	£1,074	**£700**	£403	£4,767	**£3,088**	£1,757
	Productivity	£182	**£118**	£67	£366	**£237**	£134	£156	**£101**	£57	£704	**£456**	£259
Regu-lating	Flood Regulation	£1,326	**£656**	£160	£1,849	**£915**	£223	£121	**£60**	£15	£3,296	**£1,631**	£397
	Water Quality Regulation	£4	**£2**	£1	£45	**£25**	£10	£5	**£3**	£1	£54	**£30**	£12
TOTAL		£5,488	**£2,362**	£972	£9,206	**£3,981**	£1,794	£2,182	**£1,188**	£620	£16,876	**£7,531**	£3,387

Notes:
All values are stated in million pounds (£m); 2015 prices.
The capitalised value represents the present value of ecosystem services provided over a time period of 25 years.
Where monetary values have been calculated this may only cover a proportion of the full value of the ecosystem service.

Legend:
Central Central estimate
High Higher threshold of the sensitivity analysis (even if the real value could still exceed this threshold)
Low Lower threshold of the sensitivity analysis

For valuation methods, underlying assumptions and limitations see the relevant sections of the report.

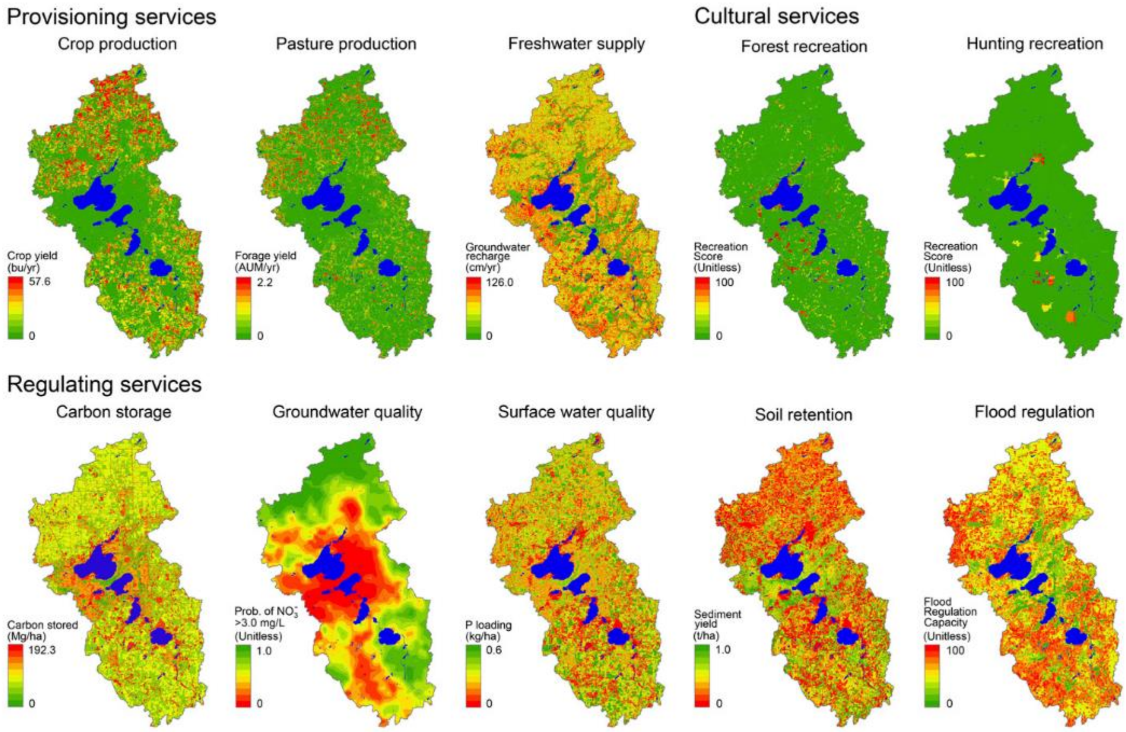


Figure 2.2: Examples of ecosystem service maps from Qiu and Turner (2013), showing the spatial distributions of 10 ecosystem services in the Yahara Watershed, Wisconsin, USA. In these examples, ecosystem service supply is measured both in physical terms, such as tonnes of carbon stored, and as unitless scores. This corresponds to stages a to c described in Figure 2.1.

2.2.1 Physical natural capital stock accounts: extent and condition

This section explores the first two steps of natural capital accounting, reviewing how the extent (Figure 2.1a) and condition (Figure 2.1b) of natural capital are accounted for.

Typically, knowledge of the extent and characteristics of natural capital assets is derived from land use and land cover mapping. Land cover can be defined as the observable biophysical material on the surface of the Earth. This can include biotic features such as vegetation, abiotic features such as water, and anthropogenic features such as tarmac (Herold et al., 2006). Land cover is related to, but distinct from, land use, and describes how these assets are used by humanity (Cihlar and Jansen, 2001). For example, in an area the land cover may be grass, while the land use may be pasture. Together, these are often referred to as Land Use / Land Cover (LULC).

LULC maps can have a wide range of characteristics with varying geographical extents, spatial and temporal resolutions, classification systems and means of production. Two LULC maps commonly used in natural capital quantification are the UK Land Cover Map (LCM) series (Rowland et al., 2017), used for example in Dales et al. (2014), Finch et al. (2021), Jones et al. (2017), Jones et al. (2019), and White (2015), and the European Union CORINE Land Cover (CLC) series (EEA, 1995), used for example in Fürst et al. (2013), Jonsson et al. (2014), and Kopmann and Rehdanz (2013).

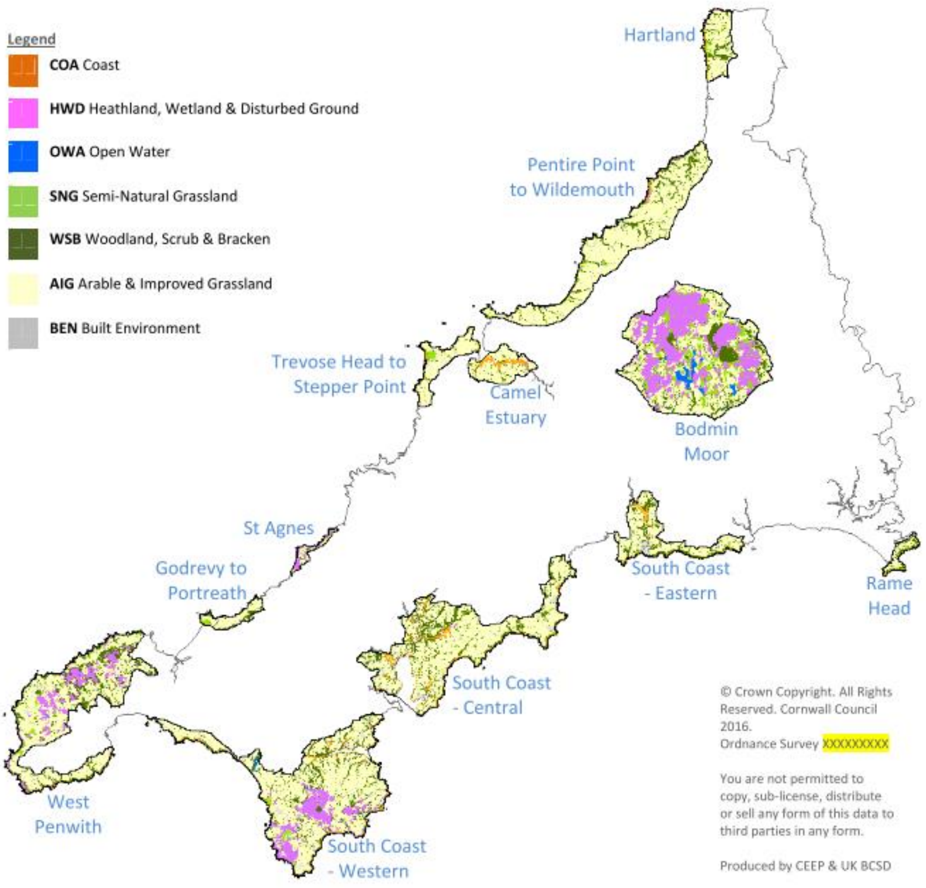
While making use of existing LULC products is common, other researchers produce bespoke classifications of satellite imagery to produce customised land cover maps. Burkhard et al. (2015) for example carried out supervised classification of SPOT5 imagery, while Hou et al. (2018) and Kreuter et al. (2001) classified Landsat imagery. This allows specific characteristics of LULC to be mapped in cases where pre-existing data may not be suitable or available. For example, a time series of imagery may be classified to monitor change over time (Kreuter et al., 2001), or high resolution imagery used to obtain more precise spatial information than is offered by existing products (Troy and Wilson, 2006).

Where information on specific assets is sought, more specialised mapping may be used, rather than general LULC maps which seek to assign a class to all features in an area. In the UK for example, the National Forest Inventory (NFI) woodland map records the location and characteristics of all woodlands and forests over half a hectare in size, using information from the interpretation of satellite imagery, aerial photography, and planting records (Forest Research, 2019a). In a study mapping the potential for timber provision, Dales et al. (2014) notes that using data from the NFI could provide a more accurate picture of existing woodland stocks than the UK LCM.

While land cover mapping is the most common source of data, some studies also integrate field measurements into their work, especially where information on the condition of assets is required (Norton et al., 2018).

These data sets may be used individually, or in combination, to identify natural capital stocks. For example, Troy and Wilson (2006) started with a 30 m resolution land cover map derived from classified Landsat imagery when carrying out an ecosystem services assessment for Maury Island, Washington, USA. This was augmented with higher 1 m resolution data from the IKONOS satellite to identify urban and barren areas, along with digitised aerial photography to delineate beaches, and existing spatial data to locate streams and wetlands. Similarly, in an ecosystem services assessment for the state of California, a vector land cover base map was augmented with updated and higher quality data for specific classes including wetlands, estuaries, streams, and woodlands to improve the precision of results (Troy and Wilson, 2006).

An example of a natural capital asset map is presented in Figure 2.3. This shows natural capital assets in the 12 management areas of the Cornwall Area of Outstanding Natural Beauty. The map was produced using existing spatial datasets, and identifies seven categories of natural capital asset. Produced as part of a natural capital assessment for the site (Hölzinger and Laughlin, 2016), the accompanying report estimates flows of services from these assets, and how they have changed over time.



*Source: **Based on GIS data provided by Cornwall Council and ERCCIS***

Figure 2.3: Example of a natural capital map for the Cornwall Area of Outstanding Natural Beauty (Hölzinger and Laughlin, 2016). The typology used here defines seven categories of natural capital asset.

2.2.2 Physical ecosystem service flow accounts

Approaches to producing physical ecosystem service flow accounts (Figure 2.1c) can typically be placed in one of two broad categories: the use of ecosystem service modelling tools, and the analysis of spatial data. These are considered in the following sections.

2.2.2.1 Ecosystem service modelling tools

A variety of models have been developed to quantify ecosystem service flows in a spatially explicit manner. These tools vary in terms of complexity, the ecosystem services considered, and the quantification approach, with some providing only measures of physical flows, and some producing monetary valuations (Bullock and Ding, 2018; Jackson et al., 2017). The more complex and comprehensive tools allow for the quantification and mapping of multiple ecosystem services at different scales, but also require higher technical skills and many inputs (Jackson et al., 2017). This may make these more complex modelling tools unsuitable if assessments need to be performed regularly, or across many sites. In these cases, the simpler benefit transfer approach (Section 2.2.4) may be more appropriate.

Two widely used examples are InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) and ARIES (Artificial Intelligence for Environment and Sustainability).

InVEST (Sharp et al., 2020) is a suite of spatially explicit Python models that considers both the supply and demand of ecosystem services, and can quantify flows in both physical and monetary terms. It includes models for quantifying 17 different ecosystem services, such as recreation and tourism, offshore wind energy production and water yield, each of which can be run independently due to the modular nature of the tool. Each model requires specific input data and parameters. While the data requirements of some models are modest, others require substantially more effort to set up, making their usage time consuming, especially where the required inputs are not readily available. Calculating annual water yield for example requires the spatial distribution of precipitation, evapotranspiration, root restricting layer depth, plant available water content, LULC, vegetation properties, watersheds and sub-watersheds at a minimum, with additional data required to produce a valuation in monetary terms. InVEST has been used in a wide range of studies. Some of these use only a single model within the toolset. Onaindia et al. (2018) for example used the crop pollination module, and Wu et al. (2013) the biodiversity module, with different techniques used to quantify other services. Others make more extensive use of the tool, using it to quantify multiple services within a study area (Bai et al., 2011; Finch et al., 2021; Hou et al., 2018; Naidoo et al., 2009; Underwood et al., 2018).

ARIES (Villa et al., 2014) is another commonly used ecosystem services model. It is accessed through a web-based platform that uses artificial intelligence to select and integrate ecosystem service models automatically. It places an emphasis on quantifying the actual demand and use of services by society, rather than only provision, but unlike InVEST focuses on physical, rather than monetary valuations. ARIES has been widely used within the research community. For example, Bagstad et al. (2017), used ARIES to quantify carbon sequestration and storage, water yield, sediment regulation, and aesthetic viewsheds in physical terms in order to identify ecosystem service hotspots in Colorado and Wyoming, USA, and Capriolo et al. (2020), used the tool to map crop pollination, outdoor recreation, flood regulation and water provision in a national assessment of ecosystem services in Italy.

In a review of comprehensive ecosystem service modelling tools (LUCI, ARIES, and InVEST), Jackson et al. (2017) found that while each modelling tool provides broadly comparable outputs, there were differences in results produced by each, and their performance against observed data for a selection of ecosystem services (water yield, carbon storage and sequestration, and nutrient retention) was variable.

While InVEST and ARIES provide comprehensive, general purpose modelling tools, other tools exist for valuing specific ecosystem services. For example, ORVal (Outdoor Recreation Valuation Tool) (Day and Smith, 2016) is a statistical recreational demand model that allows visitation and monetary welfare values to be generated for existing and hypothetical new greenspaces (Day and Smith, 2017). It has been used both to estimate the recreation value of existing sites (Clark, 2017; Day et al., 2018; Petersen, 2018; Petersen, 2021), and to predict the value of new sites in potential land use scenarios (Davis et al., 2019; Finch et al., 2021). The Cool Farm Tool (Hillier et al., 2011; Cool Farm Alliance, 2016) calculates greenhouse gas emissions from agricultural land, accounting for factors such as crop management, fertilizer use and farm machinery. It has been used in the UK National Ecosystem Assessment (Bateman et al., 2014), and in various case studies assessing emissions from different agricultural systems (Ortiz-Gonzalo et al., 2017; van Rikxoort et al., 2014; Vetter et al., 2018).

It is also possible to use more general models to estimate important components of ecosystem service assessments. The Forest Research Ecological Site Classification (Bathgate, 2011) for example provides a spatially explicit estimation of tree yield class, which can be used with emissions factors to estimate annual sequestration from an area of woodland (Finch et al., 2021). Compared with more comprehensive ecosystem service models, the more focused scope of these tools can make them more accessible. ORVal and the Ecological Site Classification models for example are available as user-friendly web apps, requiring minimal user inputs to set up and run.

As these examples illustrate, modelling tools are available for a range of ecosystem services. Some however are still poorly characterised. Jackson et al. (2017) for example notes that within commonly used tools (InVEST, ARIES, LUCI), there are a paucity of tools for mapping or quantifying cultural ecosystem services. Many modelling tools are also to a large extent black boxes, with a limited ability to modify behaviour or parameters outside of predefined settings. In some cases it may therefore be preferable to perform more bespoke analysis of spatial data.

2.2.2.2 Spatial data approaches

Information on flows of ecosystem services can also be obtained from the analysis of spatial data. This can range from the use of single layers as simple proxies for service provision, to the integration and analysis of multiple datasets to form complex biophysical models. This flexibility allows for a wide range of services to be modelled in a way that meets the requirements of specific studies. However, it also relies on a suitable methodology, and data that is of good quality, timely, and of the appropriate spatial resolution, which is not always available (Section 2.4). Perhaps the most basic approach is to simply utilise existing statistics, which describe flows of a service. Remme et al. (2014) for example used a spatially explicit dataset from a government agency recording the weight of hunted deer and boar in Limburg province, Netherlands, to quantify hunting recreation opportunities.

A similar approach is the use of existing statistics as a proxy or indicator for ecosystem service supply or use. This requires knowledge of the relationships between a measurable environmental variable, and the supply of ecosystem services (Martínez-Harms and Balvanera, 2012). These proxies may be very simple and directly linked to ecosystem service flows, such as using the number of maple syrup taps per square kilometre, obtained from agricultural census data, as a measure of maple syrup supply (Raudsepp-Hearne et al., 2010). This does not quantify an actual flow of benefits in physical or monetary terms, and so cannot be directly compared with other ecosystem services in absolute terms. It does, however, give an indication of service use, and can be used to explore spatial and temporal patterns in relative terms.

Links between proxies and ecosystem services may also be more tenuous, providing just a partial indicator of possible provision. Examples include using a database of animal observations, recorded by members of the public, as an indicator for forest recreation in Catalonia, Spain, on the basis that bird watching is an important recreational activity (Roces-Díaz et al., 2018), or using population per square kilometre as a measure of population support in north-eastern China (Wu et al., 2013), although it is questionable whether this can actually be considered an ecosystem service.

These approaches using existing statistics can be used to map current ecosystem service supply, but unlike modelling tools (Section 2.2.2.1) or more in-depth spatial analysis, cannot predict how these could change under differing management scenarios, or how supply of these services could be maximised through changes in land use.

Expert opinion may also be used to provide an indication of ecosystem service flows. Koschke et al. (2012) for example asked experts to assign values from 0 to 100 for different LULC categories, indicating their contribution to the provision of a range of ecosystem services in Saxony, Germany. Similarly, Burkhard et al. (2015) asked local experts to score the capacity of different land cover types to support or supply ecosystem services in rice cropping regions of Vietnam and the Philippines. Egoh et al. (2008) integrated expert opinion with more complex modelling of ecosystem service provision. In calculating soil retention for example, vegetation types were mapped, and ranked by their ability to curb erosion using expert knowledge. Areas with high erosion potential, and the presence of vegetation curbing this, were then identified as being important for this service.

While these approaches can give an approximate qualitative indication of ecosystem service provision, they cannot measure this in physical or monetary terms. They also rely on the knowledge of experts which may be imperfect, especially as the links between natural capital assets and ecosystem services may be complex and poorly characterised (Costanza et al., 1997).

In other cases, the spatial extent (area) of a land cover type is used as a proxy for provision. This very simple approach does not account for characteristics such as location, condition or configuration, but rather assumes that the greater the area of an asset, the more of a service it supplies. Conceptually, this could be considered similar to the benefit transfer approach (Section 2.2.4), although it makes no attempt to quantify ecosystem service flows in physical or monetary terms. For example, Raudsepp-Hearne et al. (2010) used the proportion of forest cover in an area, identified from land cover mapping, as an indicator for forest recreation, and the proportion of land used for crop production, identified from agricultural census data, as a measure of crop provision. These are in essence natural capital extent accounts (Section 2.2.1), rather than an attempt to account for service flows.

Land cover data may also be combined with existing data in a lookup table approach. Carbon storage for example may be estimated by using land cover mapping to identify carbon pools such as woodland, with data estimating the carbon content of different types of vegetation and soil (Qiu and Turner, 2013; Spanò et al., 2017). Similarly, the area of woodland could be combined with lookup tables to estimate rates of carbon sequestration by an area of trees (Remme et al., 2014; Roces-Díaz et al., 2018).

These are all comparatively simple approaches. Other studies have demonstrated how more in-depth analysis of spatial data can be used to quantify ecosystem service flows, or derive proxies for these. For example, Lautenbach et al. (2011) calculated an indicator for water quality based on the area of arable land uphill of buffer strips around water courses, and the erosion potential within these. Roces-Díaz et al. (2018) calculated a measure of flood regulation by measuring the percentage area of buffer strips along rivers covered by forest. Many studies have quantified recreation supply and demand potential using a combination of recreation opportunities such as natural forest area, population statistics, and their abilities to access these sites using transport networks (Gimona and Van Der Horst, 2007; Lautenbach et al., 2011; Qiu and Turner, 2013; Spanò et al., 2017). By identifying relationships between biophysical and spatial characteristics, and the provision of ecosystem services, these approaches have the potential to model future ecosystem service supply under differing land use and management scenarios, rather than only describing current provision.

Finally, as well as providing data for physical natural capital stock accounts through the creation of LULC mapping data (Section 2.2.1), various indicators of ecosystem service provision, and absolute measures of flows, can be obtained from remote sensing data (Chauvenet et al., 2015; Feng et al., 2010). Underwood et al. (2018) for example used the Enhanced Vegetation Index (EVI), generated from Landsat 8 imagery, to estimate above-ground live biomass, using this as a proxy for carbon stored in a landscape. Similarly, Raudsepp-Hearne et al. (2010) used estimates of net primary productivity calculated from the MODIS sensor onboard the Terra satellite to estimate above-ground carbon sequestration.

2.2.3 Monetary accounts

Once the flow of ecosystem services has been established, a range of techniques can be used to value these in monetary terms (Figure 2.1d). These can be broadly categorised as market-based methods, revealed preference methods, cost-based methods, and stated preference methods (Horlings et al., 2020; Philips, 2017).

For ecosystem services that are directly marketed, such as timber or crops, market prices may be used, once adjusted for the relevant taxes and subsidies. The UK Office for National Statistics for example suggests using market prices for valuing peat provision when producing natural capital accounts (Philips, 2017). Resource rent is another common approach, especially when valuing provisioning services. This can be described as the difference between the cost of extraction and processing, and the amount the product is sold for (Philips, 2017). Remme et al. (2015) for example used an agricultural statistics database to value crop production in the Netherlands using a resource rent approach, while Capriolo et al. (2020) used it to value water provisioning in creating natural capital accounts for Italy. Related to this, production functions attempt to value the contribution of an ecosystem service (typically a regulating service) to a market price or output through its contribution to the production process, such as the contribution of pollinators to fruit tree production (Philips, 2017).

Revealed preference techniques involve inferring the price placed on goods by consumers by examining their behaviour in a related market (Treasury, 2018). Hedonic pricing aims to extract values for environmental services from market-based transactions (Philips, 2017). For example, the relationship between house prices and peace and quiet may be analysed to assign a monetary value to the environmental benefit (Treasury, 2018). A related approach is avertive behaviour, where the consumer reveals their value for non-market environmental quality by buying substitute products, such as air filters, when the environmental quality is damaged in some way, such as the presence of air pollution (Philips, 2017). The final revealed preference technique is the travel cost method, a method used to value services such as recreation, which identifies a complementary relationship between market goods (expenditure on travel) and environmental goods (especially nature based recreational visits) (Philips, 2017). This is the approach used in the ORVal valuation tool (Section 2.2.2.1) to estimate the welfare values of new and existing greenspaces (Day and Smith, 2017).

Cost-based methods assess the costs avoided as the result of the presence of an ecosystem service (Philips, 2017). This can be the costs of damages avoided because of the service, such as using data on air pollution related health care costs that would be reduced by the removal of particulate matter by trees to value air quality regulation (Remme et al., 2015). Similarly, the replacement costs of an alternative if the ecosystem service were lost can be used. Remme et al. (2015) for example also valued groundwater extraction for drinking water, using differences in production cost between ground water extraction, and abstraction from surface sources that would have to take place were this not possible.

Finally, stated preference methods can be used to identify willingness-to-pay for ecosystem services (Horlings et al., 2020). These survey based approaches ask individuals directly what value they attach to specified environmental changes (Horlings et al., 2020), such as recreation in forests (Fitzpatrick Associates, 2005).

2.2.4 Benefit transfer

The approaches discussed in Sections 2.2.2 and 2.2.3 describe techniques for estimating physical flows of ecosystem services, and assigning these monetary values.

Alternatively, studies valuing natural capital and ecosystem services within an area may use existing values through the benefit transfer (also known as value transfer) method. This is the process by which existing valuation data is applied to a different context or situation, and is the most common method of mapping ecosystem service values in monetary terms (Schägner et al., 2013).

In practical terms, the benefit transfer approach involves the mapping of natural capital assets, which are assumed to provide a set of services. The monetary value of these services is obtained from existing studies (primary valuations) in a value per unit area format. As discussed previously (Section 2.2.1), the extent of natural capital assets can be obtained through a variety of means. Benefit transfer studies have used land cover maps (Burkhard et al., 2009b; Dales et al., 2014; Sutton and Costanza, 2002; Troy and Wilson, 2006), classified satellite imagery (Burkhard et al., 2015; Kreuter et al., 2001; Troy and Wilson, 2006) and a combination of both (Brenner et al., 2010; Troy and Wilson, 2006).

Studies utilising a benefit transfer approach have typically been carried out at municipality or national scales. An early and influential application of the benefit transfer approach to ecosystem service valuation was presented by Costanza et al. (1998) who estimated the value of ecosystem services on a global scale. The areas of 16 biomes were combined with the per unit values of up to 17 ecosystem services, averaged from over 100 primary studies. The results provided an innovative insight into the role ecosystem services play in contributing to human welfare, albeit with huge uncertainties involved. This directly contributed to further studies, with the same per unit area values subsequently being used in Sutton and Costanza (2002) with finer

1 km^2 resolution land cover data, and at a regional scale in Seidl and Moraes (2000) and Kreuter et al. (2001). While these large-scale studies provide valuable information and can inform broad policy objectives, they are not directly applicable to local scale decision making (Burkhard et al., 2009).

Troy and Wilson (2006) argue that primary valuation research will always be the best strategy for valuing ecosystem services, but that value transfer is a meaningful "second best" strategy, and a useful starting point. Primary valuation can be both time consuming and expensive. The benefit transfer approach therefore reduces both the time and resources needed to develop estimates of ecosystem service value (United States Environmental Protection Agency, 2000), allowing quantification to be carried out repeatedly at relatively low costs (Jacobs et al., 2017).

This is however at the expense of accuracy. Studies quantifying the error associated with benefit transfer based approaches to valuation are rare, however, Eigenbrod et al. (2010) found that land cover-based proxies provided a poor fit to primary data in their UK assessment, and that correlations between services changed depending on whether primary data or proxies were used for the analysis. Nevertheless, they conclude that land cover may be suitable for identifying broad-scale trends in ecosystem service provision.

Similarly, while as described previously the spatial data used to identify natural capital assets in a benefit transfer approach can come from a variety of sources, in most studies the impacts of the spatial and thematic characteristics of these valuations is discussed only in a limited fashion, although they have the potential to be significant (Kandziora et al., 2013). Indeed, many spatially explicit ecosystem service assessments do not even state the resolution at which values are mapped (Schägner et al., 2013).

2.2.5 Synthesising multiple ecosystem service assessments

Various methods can be used to synthesise the results of multiple ecosystem service assessments (Cortinovis et al., 2021), and identify 'hotspots' where provision is highest, primarily for the purposes of conservation (Cimon-Morin et al., 2013). Hotspots can be defined both as key areas that provide more than one ecosystem service (Gimona and Van Der Horst, 2007; Gos and Lavorel, 2012; Hou et al., 2018; Wu et al., 2013), or a large proportion of a single particular service (Anderson et al., 2009; Bai et al., 2011; Egoh et al., 2008; Hou et al., 2018; José V. Roces-Díaz et al., 2018; Wu et al., 2013).

A range of methods can be used to identify these hotspots (Cortinovis et al., 2021; Schröter and Remme, 2016). Typically these approaches first involve mapping the provision of individual ecosystem services to a grid of equally sized cells, as either a raster or a vector fishnet. Less commonly, spatial units such as administrative boundaries may also be used. Once mapped, the highest valued cells or spatial units are identified, often through a cell ranking approach. For each service, cells are ranked from high to low according to the ecosystem service value they contain, such as the amount of carbon storage within the cell. The top-ranking cells are then selected to form a single-service hotspot (Schröter and Remme, 2016). These top-ranking cells may be selected by a range of measures, including a top percentage or quantile (Anderson et al., 2009; Bagstad et al., 2017; Bai et al., 2011; Davids et al., 2016; Hou et al., 2018; Roces-Díaz et al., 2018; Wu et al., 2013), a top Jenks natural breaks class (Onaindia et al., 2013; Peña et al., 2018), or less commonly, those above an expert-defined threshold (Egoh et al., 2008).

Where multi-service hotspots are desired, individual ecosystem service maps are typically first combined to determine the mean ecosystem service value for each cell. Where these values are expressed in different units (such as tonnes of carbon sequestered and annual recreational visits), they must first be normalised to a common range. Each service may be assigned the same weight, or different weights to reflect difference in importance. Once the intensity is calculated, top-ranking cells can then be selected as with single-service hotspots (Schröter and Remme, 2016), including top quantiles (Gimona and Van Der Horst, 2007), or those above the average (Queiroz et al., 2015). An alternative approach involves first classifying individual service maps by quantiles, then overlaying these to identify cells that contain upper quantiles of multiple services (Qiu and Turner, 2013; Spanò et al., 2017).

A richness, or diversity approach (Davids et al., 2016; Hou et al., 2018; Wu et al., 2013), counts the number of ecosystem services provided in an area. This requires the individual services to be expressed in binary terms, i.e. presence or absence. Where they are initially expressed as a continuous variable, areas of presence are first defined using a cell ranking approach described previously. Hotspots are then defined as areas where a defined number of services are provided (Schröter and Remme, 2016).

Another approach is spatial clustering, typically using the Getis-Ord G^*_I statistic to identify clusters of high ecosystem service provision (Bagstad et al., 2017; Roces-

Díaz et al., 2018). As with a cell ranking approach, this can be used to identify single-service hotspots, or multi-service hotspots by first combining the individual layers (Schröter and Remme, 2016).

Less commonly, software such as MARXAN (Game et al., 2008) has been used to identify areas for conservation through the use of heuristic optimisation. This identifies sites to protect based on the proportion of conservation features within them. Traditionally, MARXAN has been used to identify protected areas to conserve species, however in recent years it has also been adapted to identify ecosystem service hotspots (Schröter and Remme, 2016).

The use of different approaches can result in hotspots being identified in different locations, with differing spatial distributions. Figure 2.4 for example shows multi-service hotspots in Telemark, Norway, created using three different approaches: intensity, richness, and MARXAN. While the same input data is used in each, the resulting hotspots are very different.

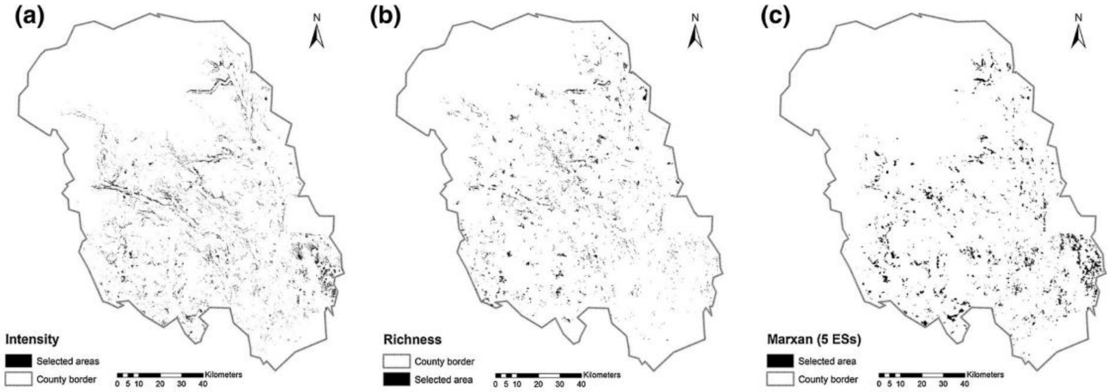


Figure 2.4: Examples of ecosystem service hotspots in Telemark, Norway, delineated using a.) the intensity approach, b.) richness, and c.) MARXAN (Schröter and Remme, 2016).

2.3 Natural capital and ecosystem services in environmental management

Natural capital, and the ecosystem services it generates is vital for human welfare, providing essential goods and underpinning the world's economic prosperity (Burkhard and Maes, 2017). Costanza et al. (1997) uses the example of artificial biospheres to highlight the value of ecosystem services, noting that experience with manned space missions indicates that replicating ecosystem services is an exceedingly complex and expensive proposition. Indeed, natural capital can be thought of as the source of all other types of capital (Natural Capital Committee, 2017), with natural capital being required for the construction of both manufactured and human capital (Constanza and Daly, 1992).

This importance has been increasingly recognised in recent years. The natural capital concept has been gaining traction at both the national and international scale, with commitments both to conserve and enhance natural capital assets, and flows of ecosystem services, and to account for them in planning and decision making. Organisations from across the world continue to join the Natural Capital Coalition

(Natural Capital Coalition, 2018). The first priority of the European Union's Environment Action Plan (Decision 1386/2013/EU, 2013) was to protect, enhance and conserve natural capital, while its biodiversity strategy aimed to halt the degradation of ecosystem services in the EU by 2020 (Com/2011/0244, 2011). In the United States, the White House Council on Environmental Quality has issued a memorandum directing government agencies to incorporate ecosystem services into federal planning and decision making (Executive Office of the President of the United States, 2015). In the UK, the Natural Capital Committee has been established to advise the government on the sustainable use of natural capital (Natural Capital Committee, 2016), and commitments to protecting and growing natural capital and using it as a tool in decision making have been made in the government's 25 year environment plan (Defra, 2018a).

The following sections explore in further detail how the quantification, valuation, and mapping of ecosystem services can be used to inform environmental management, helping us to understand, and ultimately improve, the benefits gained from nature. These sections review both how the approaches are currently applied, and how they could be used to address current environmental challenges.

2.3.1 Natural capital accounting and valuation

The quantification of natural capital and ecosystem services could provide an innovative framework for the improved management of land and resources (Pandeya et al., 2016). Specifically, the valuation of natural capital in monetary terms is being increasingly used as a tool to better understand the benefits provided by nature. Some have argued that natural capital should not be valued, rather that nature and natural capital has an infinite, intrinsic value. However, as Costanza et al. (1997) notes, values, acknowledged or not, are regularly placed on ‘intangible’ things such as environmental aesthetics and long-term ecological benefit. Even human life is valued. When construction standards are set, for example for bridges, human life is valued, because spending more money on construction would save lives; this is also the case with nature. Helm (2015) argues that by making a choice, a price is being put on nature, because conservation budgets are limited, and decisions must be made as to which assets most and where conservation efforts should be focused. If all of nature was equally priceless, these choices could not be made. Similarly, Costanza et al. (1997) and Herrera Environmental Consultants et al. (2004) argue that decisions made about ecosystems imply valuations, although not necessarily in monetary terms. The choice therefore is not whether to value nature or not, but rather should these valuations, and the uncertainties involved in them, be made explicit.

Natural capital accounts, which estimate the current monetary value of natural capital assets in an area, can clearly showcase and raise awareness of the benefits provided by natural assets. By using monetary value as a common unit, these benefits can be compared and monitored in a consistent way. This also allows environmental data to be integrated with economic data, for example by including natural capital in addition to produced capital, such as infrastructure, when calculating indicators of living standards (Connors and Philips, 2017). In addition to providing a snapshot of a single point in time, well developed accounts also provide a baseline from which changes in natural capital and ecosystem services can be tracked (Philips, 2017), for example to assess the effectiveness of conservation policies (Lü et al., 2012).

At the largest scale, global estimations of the value of natural capital and ecosystem services highlight their vast importance to humanity (Costanza et al., 1997), while national or regional accounts can raise awareness of the economic significance of assets in these areas. In the UK, natural capital accounts are being developed by the Office for National Statistics, which quantify the extent, condition, and value of natural capital assets at a national scale, and track changes over time (Connors and Philips, 2017). At a smaller scale, regional accounts have been produced for areas such as the Marches (Hölzinger, 2016) and the Black Country (Hölzinger, 2011), providing valuable evidence for organisations such as local authorities.

The valuation of natural capital therefore has the potential to be a valuable tool for improving our understanding of the natural environment and the benefits it provides to society, and monitoring how these change over time. To date, studies producing monetary natural capital accounts have typically been comparatively broad and exploratory. However, the approach could also prove effective in addressing more specific environmental challenges.

For example, with the UK's withdrawal from the European Union, and therefore Common Agricultural Policy (CAP), the development of new agri-environment policies that reflect and reward the conservation and enhancement of natural capital assets, and provision of ecosystem services, has been proposed by some (Committee on Climate Change, 2018; Defra, 2018b; Hodge, 2017).

Agricultural ecosystems are the largest ecosystems in the Anthropocene (Willemen et al., 2017), with nearly half of the EU's total land area being managed by farms

(Statistical Office of the European Communities, 2018). If well managed, agroecosystems can be a source of ecosystem services, such as the regulation of soil and water quality, carbon sequestration, support for biodiversity, and cultural services (Power, 2010). However, the CAP has been criticised for doing little to encourage these good environmental outcomes (Defra, 2018b; Matthews, 2013), and supporting food production without rewarding the provision of ecosystem services (Committee on Climate Change, 2018).

The development of such replacement agri-environment policies would fundamentally require methods to quantify and value natural capital assets, and flows of ecosystem services from these, in order to be implemented. Firstly, knowledge of the value of ecosystem services provides the scientific basis from which subsidies, or compensation for their provision and conservation, can be calculated. Secondly, monitoring and quantifying changes in service provision is required both for these payments to be made, and to assess the effectiveness and impacts of interventions (Maes et al., 2017).

2.3.2 Ecosystem service mapping and planning

In addition to the general valuation of natural capital and ecosystem services, the production and analysis of spatially explicit maps of ecosystem service provision also has the potential to be a valuable tool for informing environmental management.

Efforts to map the current distribution of ecosystem services have grown exponentially in recent years (Schägner et al., 2013), with a majority of studies having a spatial component and outputs. The production of such ecosystem service maps can support decision making, and raise public awareness and understanding (Burkhard et al., 2013; Hou et al., 2018). Examples of this include using ecosystem service maps to visualise the scales at which different services operate, identify locations of ecosystem service provision, facilitate negotiations amongst stakeholders, and target interventions to improve ecosystem service supply (Willemen et al., 2017). In addition to studying single ecosystem services, the integration of multiple ecosystem service maps, to study interactions between services and identify hotspots (Section 2.2.5), can also be used to identify areas for conservation, investment, or management (Bagstad et al., 2017; Balvanera et al., 2009; Egoh et al., 2008; Philips, 2017).

While much existing work has focused on measuring and mapping current ecosystem service flows, estimating how these could change under different land use and land management scenarios can also provide valuable information for environmental management. Finch et al. (2021) for example used empirical data and predictive models to explore whether land-sharing or land-sparing scenarios would result in the greatest ecosystem service provision at two sites in lowland England. Other examples include identifying optimal coastal realignment scenarios (Davis et al., 2019), modelling the impact of water supply scenarios (Crossman et al., 2015), and predicting the impact of stakeholder-defined land use scenarios (Nelson et al., 2009). Consideration of ecosystem service provision could therefore be beneficial whenever decisions about land use are to be made.

In the UK for example, numerous targets for woodland creation have been proposed recently by bodies including learned societies, charities, and government departments. While carbon storage and sequestration is the primary goal of these schemes, appropriate planning could also ensure other ecosystem service benefits are gained from these large-scale afforestation efforts. As well as carbon sequestration (Cannell and Milne, 1995; Dewar, 1990), woodlands are known to provide a range of important ecosystem services such as recreation (Bell and Ward Thompson, 2014; Goodenough and Waite, 2019) and flood mitigation (Dadson et al., 2017). The degree to which these ecosystem services are provided depends on the location, extent, configuration, and condition of the woodland from which they originate (Section 2.2). The spatial distribution of ecosystem service provision is therefore heterogeneous, and woodland in one location will not necessarily provide the same benefits as another, even if it has similar characteristics (Gimona and Van Der Horst, 2007). A woodland located near a population centre for example may provide a greater recreation benefit than one in a remote, hard to access site. A predictive ecosystem services modelling approach therefore offers an opportunity to target woodland creation where potential benefits are greatest.

2.4 Summary and research gaps

Natural capital is defined as the world's stock of natural assets, which yield flows of benefits to humanity, known as ecosystem services. Natural capital and the flows of ecosystem services it produces can be quantified in both physical and monetary terms. Physical natural capital stock accounts measure the extent of a natural capital asset, and its condition or quality. Physical flow accounts record the ecosystem services provided by an asset, such as tonnes of timber harvested, or the number of recreational visits. Monetary flow accounts then measure the value of these services in monetary terms. Finally, a monetary value can then be assigned to the natural capital asset itself.

Numerous methods can be used to quantify these flows of ecosystem services and assign them a monetary value, including the use of modelling tools and the analysis of spatial data. These methods vary considerably in terms of complexity, data requirements, and the accuracy and format of outputs produced.

The quantification of natural capital and ecosystem services is being increasingly recognised as a potentially valuable framework for the improved management of land. The creation of natural capital accounts can highlight and track changes in the benefits provided by natural assets, enabling them to be incorporated into decision making. Predictive modelling can estimate how changes in land use could impact ecosystem service provision, and what the optimal changes may be to maximise the provision of these services. A number of factors must however be considered when planning for the practical implementation of these approaches.

In particular, data and methods must be appropriate for the scale of the analysis. To date, many studies have focused on mapping and quantifying ecosystem services at a global scale. These studies, which cover large areas, often at coarse spatial resolutions, provide valuable information and can inform broad policy objectives, but are not directly applicable to local decision making (Burkhard et al., 2009).

For example, while agri-environment schemes may be devised and administered at a national or multi-national level, they are ultimately implemented from the bottom up by individual farms and estates. For a farmer to consider natural capital when planning potential changes in land use, or for subsidy payments to be made based on the provision of ecosystem services from within their land, accurate valuations at farm-scale are required. Existing studies mapping and quantifying ecosystem services at this scale are however rare (Chan et al., 2006), with many instead being carried out within extents well beyond the average UK farm size of 0.81 km^2 (Defra et al., 2018).

The benefit transfer approach (Section 2.2.4) is the most common technique for mapping ecosystem service values in monetary terms (Schägner et al., 2013). The approach has the potential to produce valuations rapidly and at low cost through the use of existing data (Troy and Wilson, 2006), especially when compared to more complex modelling tools (2.2.2.1) and spatial analysis (2.2.2.2), which can have more substantial requirements in terms of input data, time and expertise. It therefore has the potential to be a valuable tool for producing valuations for the many thousands of farms in the UK, and ensuring these are kept up-to-date. However, to date, its use to produce valuations within individual farms has yet to be explored.

Amongst the issues, be considered is the choice of land cover data used in the benefit transfer process. As discussed in Section 2.2.1, the spatial data used to identify the extent of natural capital assets may be obtained from a variety of sources including land cover maps and classified satellite imagery, each of which can have very different spatial and thematic characteristics. These characteristics are known to impact and bias valuations produced using the benefit transfer approach at a national scale (Konarkska et al., 2002), although in many studies the potential impacts of this are discussed only in a limited fashion. Indeed, many spatially explicit ecosystem services assessments do not mention the resolutions at which values are mapped (Schägner et al., 2013). Spatial resolution can also impact the identification of hotspots and perceived relationships between the supply of different ecosystem services, with interactions between services being dependent on the scale at which they are mapped (Hou et al., 2018; Roces-Díaz et al., 2018).

In addition to the characteristics of the data used, the spatial boundaries within which analysis takes place can also impact results and therefore needs to be considered. Planning for land use change can take place within a wide range of administrative boundaries. The UK for example has seen afforestation schemes take place nationally with the establishment of Forestry Commission plantations, regionally with initiatives such as the Northern Forest, within individual counties by local authorities with their own local afforestation targets, and within individual farms and estates through programmes such as Countryside Stewardship.

Studies have shown that the spatial distribution of ecosystem service hotspots can be impacted by the size of the spatial unit within which they are identified (Blumstein and Thompson, 2015). This then raises the more general question of which scale, if any, is most appropriate for natural capital informed land management decisions to be made at.

Therefore while these approaches have much potential, there is a need to develop and assess them at a number of scales to ensure that assessments of current and predicted future natural capital stocks and ecosystem service flows are robust. This will enable ecosystem service-based approaches to inform policy and decision making in agriculture, forestry, and land management more generally at a wide range of scales.

3. The influence of land cover data on farm-scale valuations of natural capital

Thomas Burke, J. Duncan Whyatt, Clare Rowland, G. Alan Blackburn & Jon Abbatt

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Abstract

The valuation of natural capital within individual farms could inform environmentally beneficial land use change and form the basis of agricultural subsidy schemes based on the provision of ecosystem services. Land cover extents can be used in a benefit transfer approach to produce monetary valuations of natural capital rapidly and at low cost. However, the methodology has not before been used within individual farms, and the impact of land cover data characteristics on the accuracy of valuations is uncertain. Here, we apply the approach to five UK farms of contrasting size, configuration and farming style, using three widely available land cover products. Results show that the land cover product used has a substantial impact on valuations, with differences of up to 58%, and the magnitude of this effect varies considerably according to the landscape structure of the farm. At most sites, valuation differences are driven by the extent of woodland recorded in the landscape, with higher resolution

land cover products incorporating larger amounts of woodland through inclusion of smaller patches, leading to higher overall valuations. Integrating more accurate land cover data and accounting for the condition, configuration and location of natural capital has potential to improve the accuracy of valuations.

3.1 Introduction

Recognition of the vital importance of natural capital and the ecosystem services it provides has increased in recent years. Natural capital is defined as the elements of nature that provide benefits to humanity, including ecosystems, species, freshwater, land and minerals (Spake et al., 2019; Mace et al., 2015), while ecosystem services can be defined as the contribution of ecosystem structure and function to human wellbeing (Burkhard and Maes, 2017). Efforts to map the spatial distribution of these have grown exponentially since Costanza et al. (1997) presented their seminal study valuing ecosystem services globally (Schägner et al., 2013), and the natural capital concept is now being integrated into planning and policy development. The European Union aims to halt the degradation of ecosystem services in the EU by 2020 (European Commission, 2011), while organisations from across the world continue to join the Natural Capital Coalition (Natural Capital Coalition, 2018). The UK has established the Natural Capital Committee to advise the government on the sustainable use of natural capital (Natural Capital Committee, 2016), and commitments to protecting and growing natural capital and using it as a tool in decision making have been made in the government's 25 year environment plan (Defra, 2018a).

The conservation and enhancement of natural capital assets will necessitate work in agricultural areas. Nearly half of the EU's total land area is managed by farms, including arable land and grassland, and other features such as woodland and water that can be found within farms and estates (Statistical Office of the European Communities, 2018). Agricultural landscapes therefore encompass large areas of natural capital assets, which provide vital ecosystem services such as carbon sequestration and pollution removal. Within the EU there has been growing concern that agricultural subsidies provided through the Common Agricultural Policy have done little to preserve these assets and the services they provide. It has been argued that these area-based payments do not improve environmental outcomes (Defra, 2018a), and support food production without rewarding the provision of ecosystem services (Committee on Climate Change, 2018). Reforms in 2014 saw the introduction of ‘greening’, where farmers must implement certain environmentally focused measures or lose up to 30% of their basic payments. However, it has been argued that these measures, which were diluted from initial proposals, are unlikely to lead to major environmental improvements (Matthews, 2013) and unlikely to benefit biodiversity (Pe’er et al., 2014). The European Court of Auditors (2017) have since concluded that whilst greening adds complexity to the payments system, it is unlikely to provide significant benefits to the environment and climate, and has led to very limited changes in farming practices. The development of new land use policies that reflect and reward the provision of ecosystem services and the conservation and enhancement of natural capital assets have therefore been proposed (Hodge, 2016; Committee on Climate Change, 2017; Defra, 2018b). Importantly, such policies rely on the ability to accurately and objectively quantify the natural capital and value the ecosystem services provided on individual farms and monitor changes over time.

Schägner et al. (2013) reviewed current approaches to mapping ecosystem service values in monetary terms. Most common is the use of land cover data to map the extent of natural capital assets, which are assumed to supply a set of services. The value of these services is then obtained from existing studies (primary valuations) in a value per unit area format. For example, land cover data may be used to identify 1,044 ha of woodland at a site. Existing valuation data may indicate that a hectare of similar woodland provides $1,826 worth of benefits to humanity annually through the provision of services such as carbon sequestration and pollution removal, leading to the provision of $1,906,410 of services each year (Troy and Wilson, 2006). This process is repeated for other assets identified from land cover data to value the total ecosystem service provision within an area. We refer to this methodology as the land cover and benefit transfer technique.

These primary valuations are themselves obtained through a number of methods. For example, timber production, a provisioning service, may be measured in physical terms as the volume in cubic metres of timber harvested. Stumpage prices (the price paid to harvest a given volume of timber) may then be used as a measure of its monetary value (United Nations, 2012). For other services, the value can be less obvious. For example, the travel cost method may be used where the costs incurred by travelling to a site is used as a proxy for the sites recreational value (Philips, 2017). This however can be time consuming and costly to carry out. By using existing data, the land cover and benefit transfer technique has the potential to allow for the valuation of ecosystem service provision within an area rapidly and at a low cost, and provides an alternative when primary research is not possible or feasible (Troy and Wilson, 2006).

The spatial data that is typically used to support this approach includes land cover maps (Sutton and Costanza, 2002; Dales et al., 2014; Troy and Wilson, 2006;

Burkhard et al., 2009), classified satellite imagery (Kreuter et al., 2001; Troy and Wilson, 2006; Burkhard et al., 2015) and a combination of multiple layers (Brenner et al., 2010; Troy and Wilson, 2006). However, in most cases, the impacts of the spatial and thematic characteristics of these data on valuations is discussed only in a limited fashion, although they have the potential to be significant (Kandziora et al., 2013). Indeed, many spatially explicit ecosystem service assessments do not even state the resolution at which values are mapped (Schägner et al., 2013).

To date, studies focused on the measurement and valuation of natural capital in agricultural areas and at a scale appropriate for management and decision-making on individual farms have been limited. The land cover and benefit transfer approach has largely been used to produce valuations across large areas and at coarse spatial resolutions. Costanza et al. (1997) estimated the value of ecosystem services on a global scale. For each of 16 biomes, their areas were combined with the per unit values of up to 17 ecosystem services, averaged from over 100 primary studies. The global nature of this study means these biomes are extremely broad, aggregating together for example African rangeland and British pastures, while the 1 degree spatial resolution of the land cover data used (Matthews, 1982) means whole farms and estates would be assigned a single land cover. These per unit values were subsequently used in Sutton and Costanza (2002) with finer 1 km2 resolution land cover data, although this is still too coarse to map natural capital assets at a local scale. These broad global valuations compiled by Costanza *et al*. have also been used in regional scale studies, including Seidl and Moraes (2000) and Kreuter et al. (2001). Alternatively, Brenner et al. (2010) compiled a new database of primary valuations.

While the land cover data used in these studies potentially has a high enough spatial resolution to detect farm scale variations, none are focused on agricultural areas. The regions studied are also far beyond the size of a typical British or European farm. Troy and Wilson (2006) mapped ecosystem service values at five locations including Maury Island, Washington, a site covering 2,495 ha. Although comparable in size to farms studied in this paper, the island nature of the site meant most land cover classes used were coastal in nature.

Where valuations have been carried out in a primarily agricultural context, this has involved the use of land cover data, but not benefit transfer, and the valuation of a small selection of services. This includes the use of expert opinions to rate the ability of different land covers to supply ecosystem services in rice cropping regions of southeast Asia (Burkhard et al., 2015), and the use of statistical data such as crop composition and yield to quantify food provision in the Halla-Leipzig region of Germany (Burkhard et al., 2009). In the UK, national natural capital accounts provide valuations for ecosystem services provided by assets including farmland, freshwater and woodland (Connors and Philips, 2017). However, these accounts are not spatially explicit, and do not describe provision in individual farms and holdings. Dales et al. (2014) produced maps of 10 ecosystem services using data from the UK National Ecosystem Assessment (UK National Ecosystem Assessment, 2011). However, rather than providing a monetary valuation, this assessed the importance of eight broad habitats for delivering 16 ecosystem services, with each being assigned a category from “High” to “Low” or not applicable.

Therefore, there is a pressing need for a methodology that is capable of quantifying the monetary value of the provision of ecosystem services within individual farms in a spatially explicit manner. While large scale national or regional studies can reveal general trends and inform broad policy objectives, local, farm scale data is required to implement these. For a farmer to consider natural capital when planning potential changes in land use, or for subsidy payments to be made based on the provision of ecosystem services from within their land, valuations known to be accurate at a local scale are required. The land cover and benefit transfer approach described here is well established and has the potential to produce these valuations in a quick and cost effective way. However, its use within individual farms, and the impact of land cover data on the accuracy of valuations, have yet to be adequately explored.

In this paper, we use the land cover and benefit transfer approach to produce monetary valuations of ecosystem service provision within individual farms. Using three commonly used land cover datasets as inputs in the valuation process, we explore how their differing characteristics impact the valuations produced. Through the use of five farms with contrasting landscape characteristics as case studies, ranging from small to large landholdings and covering livestock and arable farming, we explore how the interactions between land cover data and landscape characteristics can influence valuations in different environments. Finally, we explore how the approach could be developed further in order to provide more accurate valuations of ecosystem services.

3.2 Materials and methods

3.2.1 Study sites

The UK was chosen as the study area for this work as it contains farms that vary over a wide range of sizes, landscape configurations and farming styles, while land cover data at a range of thematic and spatial resolutions is available for the country (Section 3.2.2). National natural capital accounts are currently being developed for the UK, which incorporate most of the land cover types found in the country (Section 3.2.5). Furthermore, the UK government has recently proposed an overarching framework for sustaining agriculture and protecting the environment which is based on a natural capital approach (Defra, 2018b); appropriate valuation mechanisms are now required in order to implement this approach at the individual farm scale.

Five farms were chosen as case studies in order to test the applicability of the land cover and benefit transfer approach. Table 3.1 shows that the five farms cover a range of sizes and types, which are typical of the UK, while Figure 3.1 shows their distribution and boundaries, which span a range of different landscape characteristics (this is demonstrated further in the results section below).

Table 3.1: Summary of the key characteristics of the farms studied.

Site	Location	Size (ha)	Type
Site 1	Leven, Fife	652	Arable, pasture, forestry
Site 2	Cheviot Hills, Northumberland	4,897	Upland sheep farming
Site 3	Penrith, Cumbria	4,150	Sheep farming, pasture, some arable
Site 4	Ashbourne, Derbyshire	315	Dairy
Site 5	Farnham, Surrey	900	Traditional mixed agriculture

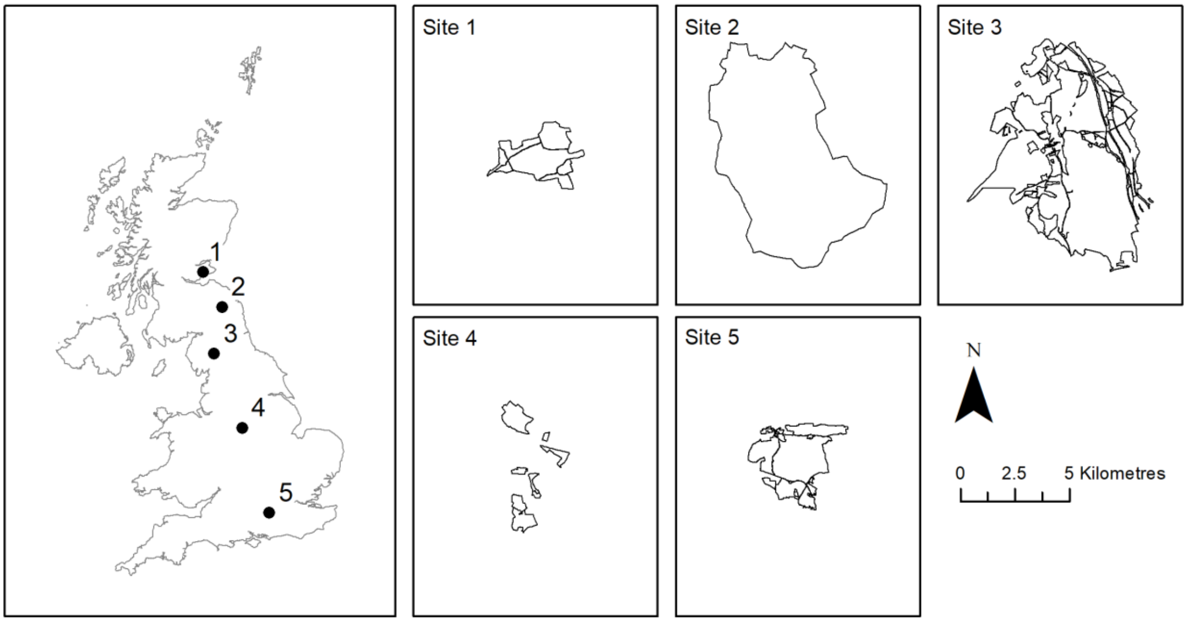

Figure 3.1: Locations and boundaries of farms used in the study.

3.2.2 Land use / land cover data

For all sites, three land cover data sources were used: the Centre for Ecology & Hydrology Land Cover Map 2015 (LCM) (Rowland et al., 2017), CORINE Land Cover 2012 (CLC) (Copernicus Programme, 2019) and the European Space Agency Climate Change Initiative Land Cover map 2015 (CCI-LC) (UCL Geomatics, 2017). The characteristics of the three datasets are summarised in Table 3.2. For each source, the most recent release available was used.

Table 3.2: Comparison of the characteristics of the three land cover datasets used.

	LCM	**CLC**	**CCI-LC**
Spatial Resolution	Minimum mappable unit 0.5 ha, minimum feature width 20 m	Minimum mappable unit 25 ha, minimum feature width 100 m	300 m pixels (9 ha)
Spatial Extent	United Kingdom (UK)	Much of Europe	Global
Attribute Resolution	21 classes, based on UK Biodiversity Action Plan Broad Habitats (Jackson, 2000)	44 classes in a three-level hierarchy	22 classes, some further divided with regional information
Classification Methodology	Random forest classifier, simplified Ordnance Survey cartography as spatial framework	Computer aided manual interpretation	Pixel classifier. Annual maps produced by back / up-dating a baseline map
Format	Vector, parcel based	Vector, parcel based	Raster, pixel based
Access	Requires license for vector version	Freely available	Freely available
Notes	n/a	n/a	Urban areas (Pesaresi et al., 2013; Pesaresi et al., 2016) and water bodies (UCL Geomatics, 2017) largely identified using external datasets.

3.2.3 Land cover classification system harmonisation

Each of the three land cover datasets used in this study employs a different classification scheme, with different numbers of output classes that represent different types of land cover. To enable comparisons between datasets, a common classification system was developed. Each of the three land cover maps were reclassified, where necessary by renaming or combing the original classes, to produce a land cover map that had eight 'harmonised' output classes. Table 3.3 demonstrates how the original classes from the three land cover maps correspond with the harmonised classes.

It has to be recognised that due to the disparate nature of the classification schemes used in each dataset, there are some uncertainties in the correspondence between classes. For example, the CLC *Sport and leisure facilities* class is part of the *Artificial non-agricultural vegetated areas* category in the three-level hierarchical CLC classification scheme. This is a land use, rather than land cover class, and includes buildings, infrastructure, or green spaces that are used for sport and leisure. In this study, this class was assigned to the *built-up areas* harmonised class for comparison purposes, but it may include land covers that could be more appropriately assigned to another class, such as grassland. Similarly, the CLC *sparsely vegetated areas* class was assigned to *bare-areas.* The CLC nomenclature guidelines note that this class represents areas where vegetation covers 10 – 50% of the surface, therefore much of the land surface will be bare earth. However, it is noted that by doing this, the extent of vegetation present will be underestimated. Due to their broad nature, CCI-LC classes were harmonised using their correspondence with IPCC land categories (UCL Geomatics, 2017).

Table 3.3: Land cover classes present in the three original land cover maps, and the harmonised class they were assigned to.

Harmonised class	**LCM**	**CLC**	**CCI-LC**
Grassland	Acid grassland Calcareous grassland Improved grassland	Natural grasslands Pastures	Grassland Mosaic herbaceous cover (>50%) / tree and shrub (<50%)
Arable and horticulture	Arable and horticulture	Non-irrigated arable land	Cropland, rainfed - Herbaceous cover Mosaic cropland (>50%) / natural vegetation (tree, shrub, herbaceous cover) (<50%) Mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%)
Bog	Bog	Peat bogs	Shrub or herbaceous cover, flooded, fresh/saline/brakish water
Woodland	Broadleaf woodland Coniferous woodland	Broad-leaved forest Coniferous forest Mixed forest Transitional woodland-shrub	Tree cover, broadleaved, deciduous, closed to open (>15%) Tree cover, needleleaved, evergreen, closed to open (>15%) Mosaic tree and shrub (>50%) / herbaceous cover (<50%)
Freshwater	Freshwater	No equivalent mapped in the five farm areas	No equivalent mapped in the five farm areas
Heather	Heather Heather grassland	Moors and heathland	No equivalent mapped in the five farm areas

Bare areas	Inland rock	Mineral extraction sites Sparsely vegetated areas	Bare areas
Built-up areas	Suburban Urban	Discontinuous urban fabric Sport and leisure facilities	Urban areas

3.2.4 Accuracy assessment

An accuracy assessment of the three land cover maps was carried out for each of the five farm sites. Reference data pertaining to the eight harmonised land cover classes was collected by a single researcher through manual visual interpretation of high resolution aerial photography (Esri World Imagery layer, 2009-2016). The reference data were collected at point locations using a stratified random sampling approach. As the product with the highest spatial resolution, LCM was used to stratify the reference points according to land cover class. For each class, reference points were created at random locations within the boundaries of that class at each site, with the number of points being equal to one point per ten hectares of that class. A minimum spacing of 25 m was used, and a minimum of three points were sampled for each class, at each study site. Points that lay on the boundary between two land cover classes, or where the land cover could not be accurately distinguished were excluded (Table 3.4). Confusion matrices were then constructed using the reference data and three land cover maps at each site, and the overall accuracy (%) and kappa coefficient was calculated.

Table 3.4: Number of reference points classified and excluded at each site.

Site	Reference points used for validation	Number of reference points excluded
Site 1	72	1
Site 2	482	15
Site 3	412	7
Site 4	33	1
Site 5	85	12

3.2.5 Ecosystem service valuations

Ecosystem service valuations (Table 3.5) were primarily calculated from UK natural capital ecosystem service accounts (Connors and Philips, 2017). These accounts present the total monetary value of ecosystem services provided by farmland (arable and horticultural land, improved grassland and rough grazing), freshwater (wetlands and open waters) and woodland (coniferous and deciduous) for the whole of the UK. The general methodologies used to obtain these valuations are outlined in Philips (2017), and vary depending on the ecosystem service and natural capital asset from which it originates. For example, for timber provision, the volume of removals is sourced from the Forestry Commission, and their value from the Forestry Commission Coniferous Standing Sales Price Index. For recreational visits to freshwaters, the number of visits and amount spent during trips were obtained from Natural England's

Monitoring Engagement in the Natural Environment survey. Admission fees and travel expenditure are then used as an estimate of willingness to pay for access to the site through the travel cost method. The values of each service provided by an asset are then summed to obtain an overall valuation. For example, woodland is assigned a high valuation primarily due to the significant value of carbon sequestration and pollution removal.

For each land cover, we divided the total value of services provided by its total area in the UK to derive a per unit area value in the format £/ha/yr. For some years valuations within the accounts were incomplete, with some services not being valued, and so here the most recent complete valuation was used. Values for heather and bog, or comparable land covers, were not available from Connors and Philips (2017), and so were sourced from the literature. Similar to the UK natural capital accounts, this study calculated first the physical (Remme et al., 2014) and then monetary (Remme et al., 2015) flows of services using various methods. Monetary valuations were then divided by the area of landcover to produce a per hectare valuation. Values for built-up and bare areas were assumed to be zero, although this is likely to be an underestimate.

The resulting valuations are therefore based on best available data but are limited by the broad nature of the classification systems used in the primary studies. For example, it is recognised that service provision from grassland and arable land will differ. However, at this time they are treated as a single unit within the UK natural capital ecosystem accounts and disaggregation is not currently possible. Similarly, not all ecosystems are valued, and different services are valued for the different land covers. Other factors that will affect ecosystem service flow such as the condition, configuration and location of natural capital assets are also not accounted for here, but are identified as an important topic for future research (Section 3.4.1).

Table 3.5: Details of ecosystem service valuations used, showing how the original valuations map onto the harmonised land cover classes used in this study.

Valuation class	Value (£/ha/yr)	Source	Services Valued	Notes	Harmonised Class
Farmland	105	Connors and Philips (2017)	Crops and grazed biomass Water abstraction Pollution removed Time spent at habitat Education visits	For year 2014. Includes arable and horticulture, improved grassland and rough grazing	Arable and horticulture Grassland
Freshwater	569	Connors and Philips (2017)	Water abstraction Peat extraction Fish capture Pollution removed Time spent at habitat	For year 2013	Freshwater
Woodland	738	Connors and Philips (2017)	Total timber removals Carbon sequestration	For year 2015	Woodland

			Pollution removal Time spent at habitat		
Heath	384	Remme et al. (2015), Remme et al. (2014)	Hunting Drinking water extraction Air quality regulation Carbon sequestration Nature tourism	Average value used. Converted from €426/ha/yr	Heather
Peatland	412	Remme et al. (2015), Remme et al. (2014)	Hunting Air quality regulation Carbon sequestration Nature tourism	Average value used. Converted from €457/ha/yr	Bog
Built-up areas	0	Assumed value	n/a	n/a	Built-up areas
Bare areas	0	Assumed value	n/a	n/a	Bare areas

3.2.6 Valuation process

The total extent of each land cover type as recorded by the three land cover datasets was calculated for each farm. These were then multiplied by the value per unit area for each land cover type to obtain a total annual monetary value of ecosystem services for each farm. The total value was then divided by the total area of each farm to calculate an average value of ecosystem services per hectare, for comparison with current government subsidy values which are expressed on a per hectare basis and for comparison with valuations performed in previous research.

3.3 Results

3.3.1 Comparison of land cover datasets

3.3.1.1 Accuracy

The accuracy assessment indicates that LCM is the most accurate of the three products assessed (Table 3.6), with the highest overall accuracy, 19 – 35% greater than CCI-LC or CLC at all locations barring Site 4 (the small dairy farm), and the highest kappa coefficient at all sites. The overall accuracy for LCM ranges from 78% at Site 2, to 89% at Site 5. CLC and CCI-LC display similar overall accuracies, with at most a 3.6% difference between them (Site 3).

Table 3.6: Overall accuracy and kappa coefficient for the three land cover maps at each site.

Site	LCM		CLC		CCI-LC	
	Accuracy (%)	Kappa	Accuracy (%)	Kappa	Accuracy (%)	Kappa
Site 1	86	0.80	53	0.23	51	0.20
Site 2	78	0.59	58	0.34	59	0.12
Site 3	83	0.67	67	0.36	63	0.12
Site 4	85	0.49	88	0.40	88	0
Site 5	89	0.84	69	0.53	71	0.52

3.3.1.2 Spatial and thematic resolution

Visual inspection of the harmonised maps produced for each site indicates that while all datasets show broadly similar patterns of land cover, there are significant differences. LCM, having the highest spatial resolution, records smaller patches of land cover. This is especially apparent at Site 3 (Figure 3.2) where LCM records many small patches of trees scattered across the landscape, while CLC shows only the larger patches at the southern and northern ends, and CCI-LC only woodland to the north. Here, nearly half of the woodland recorded by LCM is present in parcels below 25 ha (the minimum mappable unit of CLC), and 19% in parcels below 100 m (the minimum mappable width of CLC) (Table 3.7).

Table 3.7: The proportion of woodland recorded by LCM present in parcels below 25 ha in area, and below 100 m in width, the minimum mappable unit and width for CLC, respectively.

Site	% woodland below 25 ha	% woodland below 100 m
Site 1	14	19
Site 2	12	6
Site 3	43	19
Site 4	8	85
Site 5	9	5

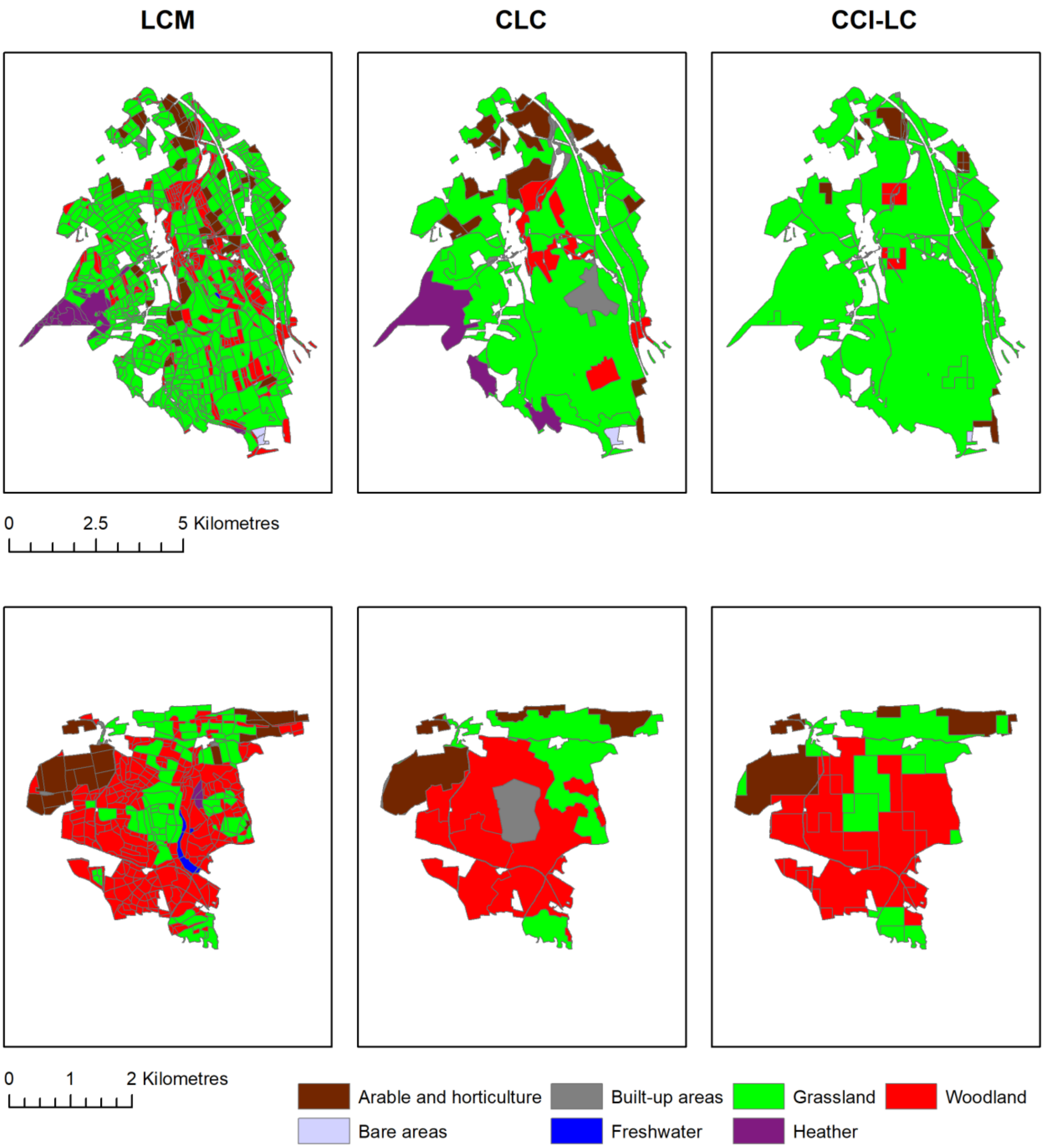

Figure 3.2: Land cover maps, using the harmonised classification scheme, for Site 3 (top) and Site 5 (bottom).

When the land cover maps are harmonised according to Table 3, LCM records the most classes and CCI-LC the least for all sites (Table 3.8). This is due to both the spatial and thematic characteristics of the products used. The high spatial resolution of LCM makes it possible to record smaller features such as the river at Site 5 (Figure 3.2) and buildings at Site 1 and Site 2, which are not included in the other datasets. CCI-LC does not include a class for heather or comparable land covers, instead including it within broad shrub and herbaceous cover categories, which when harmonised are classed as grassland and bog.

Table 3.8: Number of unique land cover classes recorded at each site by each dataset, using the harmonised classification system described in Table 3.

Site	LCM	CLC	CCI-LC
Site 1	6	3	3
Site 2	6	5	4
Site 3	7	6	5
Site 4	3	2	1
Site 5	6	5	3

The land cover classes used by LCM, based on UK Biodiversity Action Plan Broad Habitats (Jackson, 2000), were easily matched with ecosystem service valuations used here (Connors and Philips, 2017; Remme *et al*., 2015). However, the original CLC classification system includes both land use and land cover classes. This can be seen at Site 3 and Site 5 (Figure 3.2), where large areas of grassland are classed by CLC as sport and leisure facilities. As this category can include both green space and buildings, it was assigned to built-up areas in our harmonised classification scheme. However, in reality, the area recorded is simply fields used for recreational activities.

Similarly, the original CCI-LC classification scheme uses several broad mosaic classes, such as *mosaic natural vegetation (tree, shrub, herbaceous cover) (>50%) / cropland (<50%)*, aggregating together features that have significantly different ecosystem service values.

3.3.1.3 Monetary ecosystem service accounts

Total ecosystem service valuations range from £33,110 per annum at Site 4 (using CLC and CCI-LC) to £1,264,299 per annum at Site 2 (using CLC) (Table 3.9). Figure 3.3 shows the distribution of ecosystem service values from each land cover class at the five sites. The average ecosystem service value per hectare ranges from £105/ha/yr at Site 4 (using CLC and CCI-LC), to £456/ha/yr at Site 5 (using LCM) (Table 3.10).

Table 3.9: Total annual ecosystem service valuations for each site, as derived from the three different harmonised land cover maps.

	Total ecosystem service valuation (£/yr)		
	LCM	**CLC**	**CCI-LC**
Site 1	170,269	119,605	98,623
Site 2	1,254,608	1,264,299	792,526
Site 3	870,135	694,129	478,186
Site 4	35,399	33,110	33,110
Site 5	410,721	387,564	404,215

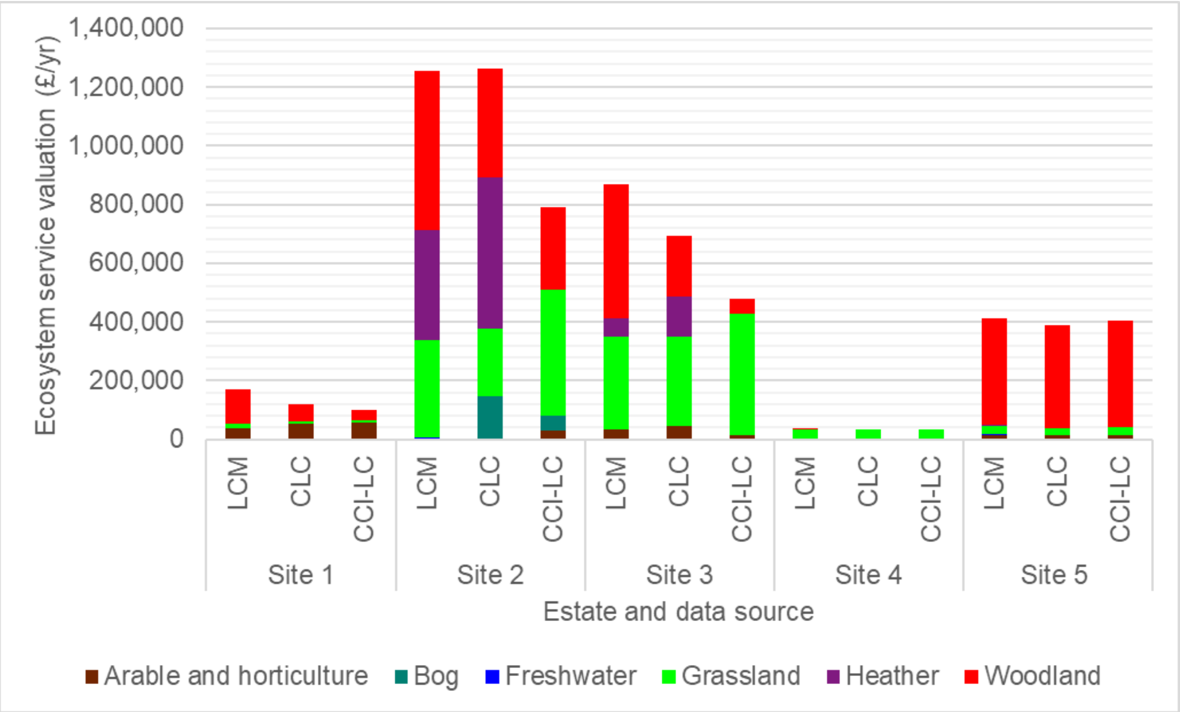

Figure 3.3: Ecosystem service valuations by land cover class for the five sites, based on the three different harmonised land cover maps.

Table 3.10: Average annual ecosystem service value per hectare for each site, as derived from the three different land cover maps.

	Average ecosystem service valuation (£/ha/yr)		
	LCM	**CLC**	**CCI-LC**
Site 1	261	183	151
Site 2	256	258	162
Site 3	210	167	115
Site 4	112	105	105
Site 5	456	431	449

It is clear that the varying spatial and thematic characteristics of the land cover data used has a significant impact on final monetary valuations. Using LCM as the input spatial data results in the highest valuations for all sites, bar Site 2, where CLC results in a slightly higher valuation. Conversely, CCI-LC leads to the lowest valuations at all sites except Site 5, where CLC is marginally lower.

LCM records the most woodland at all sites, in part due to its ability to record small and narrow parcels of trees. As woodland has the highest ecosystem service value (£738/ha/yr), this results in a higher overall valuation. The ability of LCM to detect a wider range of land cover types and smaller features has a minor impact. For example, the presence of freshwater accounts for at most 1.3% of a valuation (Site 5). Rather, at most sites valuations are primarily dominated by farmland (arable and grassland), which has a low value (£105/ha/yr) but is present in large areas, and woodland, which has the highest value and is present in moderate amounts.

The characteristics of the sites themselves also influence valuations and the suitability of different datasets for producing them. The consistency of valuations produced using different land cover data vary considerably between sites. At certain farms, such as Site 3, the landscape is highly fragmented, with small patches of land cover, especially trees, scattered throughout the site. Here, annual valuations range from £478,186 using CCI-LC to £870,135 using LCM, a difference of 58%, with LCM recording 620 ha of woodland, and CCI-LC just 69 ha. Conversely, at sites such as Site 5, the landscape, and especially areas of woodland, are more continuous. Here valuations are the most homogenous, ranging from £387,564 using CLC to £410,721 based on LCM, a difference of just 6%. This indicates that at certain sites with large continuous areas of land cover, coarser resolution datasets such as CLC and CCI-LC may be suitable for producing valuations, while at others the ability to distinguish small patches of land cover, and therefore a high resolution product, is required.

While most sites examined in this study are dominated by woodland and farmland, at Site 2 heather also makes a significant contribution to valuations. The upland heather moorland environment of this site results in perhaps the most uncertain of valuations produced. Here, 17% of the land area is classified differently in each dataset. LCM has its lowest accuracy of all sites, while for CCI and CORINE it is their second lowest behind Site 1. This may be due to the difficulties involved in classifying spectrally similar land covers, or differences in their exact definition. For example, while LCM requires an area to have a layer of peat 50 cm or higher to be classified as bog (Centre for Ecology & Hydrology, 2017), for CLC the requirement is 30 cm

(Kosztra et al., 2017). The landscape here is also very much a mosaic, with it being difficult to determine when one land cover ends, and another begins.

3.4 Discussion

The use of secondary data within a land cover and benefit transfer methodology allowed for farm scale valuations to be produced rapidly and at little cost, with the most time-consuming aspect being the identification of ecosystem service valuations per unit area for the different land cover types.

The average ecosystem service value per hectare ranges from £105/ha/yr to £456/ha/yr, which is comparable to current agricultural subsidies in the UK provided through the Basic Payments Scheme. However, these payments, which range from £63/ha/yr to £232/ha/yr in England (Rural Payments Agency, 2018) and £12/ha/yr to £218/ha/yr in Scotland (Rural Payments & Services, 2016), are provided only for grassland (including heather suitable for grazing) and arable land (Rural Payments Agency, 2018; Rural Payments & Services, 2017), which were found to have the lowest ecosystem service valuations. Features such as woodland, which have the highest valuations, are excluded. Additional funding can be sought through schemes such as the Rural Development Program for England which provides payments not only for agricultural land, but also includes multi-year grants for the creation and management of woodland (Rural Payments Agency et al., 2019). It is also important to note that the valuations produced in this study should be interpreted as partial or minimal as a number of ecosystem services, such as pollination, are not included (Connors and Philips, 2017). This complexity makes meaningful comparison between the measured ecosystem service valuation for a site, the true value of services provided, and the total amount of subsidies and funding available difficult.

In a test study, eftec (2018) found that without agricultural subsidies, both an environmentally focused organic estate and a more typical intensive farm would make a loss financially. However, while the environmentally focused site produced net benefits from natural capital such as soil carbon sequestration, the more typical site led to a degradation of public goods. Introducing a natural capital approach to agricultural policy development would allow for the impact of farming practices on the environment, both positive and negative, to be demonstrated, and ensure that funding supports both beneficial farming practices, as well as food production.

It is difficult to compare valuations presented here with past studies using the land cover and benefit transfer approach, as these have been carried out in significantly different environments and at different scales. Sutton and Costanza (2002) determined a total terrestrial ecosystem service value of $49 billion for the whole of the UK, equal to an average of £1593/ha/yr[1], which is significantly higher than estimates produced for the agricultural sites in this study. This may in part be due to higher valuations for certain land cover classes, as well as differences in the distribution of land cover in agricultural areas versus the country as a whole. Troy and Wilson (2006) derived valuations for a number of sites of varying spatial scales in the USA. Again, ecosystem service valuations are generally higher than those used here, being inflated

[1] Using an exchange rate of 1 USD = 0.79 GBP, obtained 20 June 2019

by highly valued coastal and wetland classes. For example, ‘Freshwater wetland’ at Maury Island is valued at £57,502/ha/yr[1], and ‘Fresh water bodies / coastal embayments’ at Massachusetts £30,165/ha/yr[1]. This suggests that valuations in the present study are lower than those seen in previous studies due to the inland locations of the sites used and lack of inland water bodies. Kreuter et al. (2001) calculated an average value of £118/ha/yr[1] to £126/ha/yr[1] for a 141,67 ha area of San Antonio, Texas, using the same ecosystem service values per land cover type as Sutton and Costanza (2002). This is significantly lower than valuations in other studies, and comparable to those generated by the present study. This may be due to the land cover classes recorded in the study area: ‘Rangeland’, ‘Woodland’, ‘Bare soil’, ‘Residential’ and ‘Commercial and Transportation’, with no highly valued coastal or wetland classes, which are comparable to the land covers at the five sites in this study. These studies also value different services. For example, Sutton and Costanza (2002) and Kreuter et al. (2001) include valuations for a number of services not considered here (Table 3.5) including biological control, genetic resources and soil formation. Troy and Wilson (2006) similarly account for soil retention and formation. Conversely, educational visits to farmland are accounted for in our study, but not in these previous studies.

3.4.1 Uncertainties and future work

In the land cover and benefit transfer approach, land cover is used as a proxy for ecosystem service supply. However, there are uncertainties associated with this. Eigenbrod et al. (2010) compared land cover-based proxies to primary data, finding that while proxies may be suitable for identifying broad-scale trends in ecosystem services, there was a poor fit of proxies to the primary data. Nevertheless, as highlighted earlier, it is impractical and financially prohibitive to collect primary field survey data on ecosystem services across the broad spatial scales covered in this study. The spatially continuous nature of land cover maps offers a more comprehensive method for quantifying ecosystem services.

As acknowledged in Section 3.2.5, the valuation categories used here are broad, aggregating together for example farmland and grassland, limiting the accuracy of valuations produced. It can be expected that in time, as the number of primary valuations increases, the use of more fine grained classification schemes will become more viable. Due to the simple nature of the valuation approach used here, it would be straightforward for valuations per unit area to be updated to reflect improvements in knowledge of ecosystem service provision.

In this work we use a simple benefit transfer technique, with unadjusted unit values obtained from existing studies. However, more sophisticated approaches exist, such as value function transfer which predicts ecosystem service values as functions of the characteristics of the assets, the beneficiaries, and the context within which they will be provided (Ready and Navrud, 2005; Brouwer, 2000). Schägner *et al.* (2013) reviewed methodologies for mapping ecosystem service values. By using a single value for each type of asset, there is an assumption that ecosystem service supply and value is uniform across a given land cover. However, this is a gross simplification. Other techniques include the use of adjusted unit values, value functions, and validated and non-validated models. Through the use of these more sophisticated techniques, a range of attributes can and should be considered when estimating the value of ecosystem services provided. In the UK natural capital accounts methodology (Philips, 2017), natural assets are identified as stocks, which give rise to flows of services. Three main characteristics are described which can influence the capacity of these stocks of assets to deliver ecosystem services: extent, condition and spatial configuration. Included in measures of condition is proximity to areas of population. We suggest that this could be considered as part of a wider assessment of the location of the asset, that is, its position in relation to other assets. Based on this, we propose a four step framework for the assessment of natural capital stocks in order to accurately assess flows of services in physical and then monetary terms (Figure 3.4).

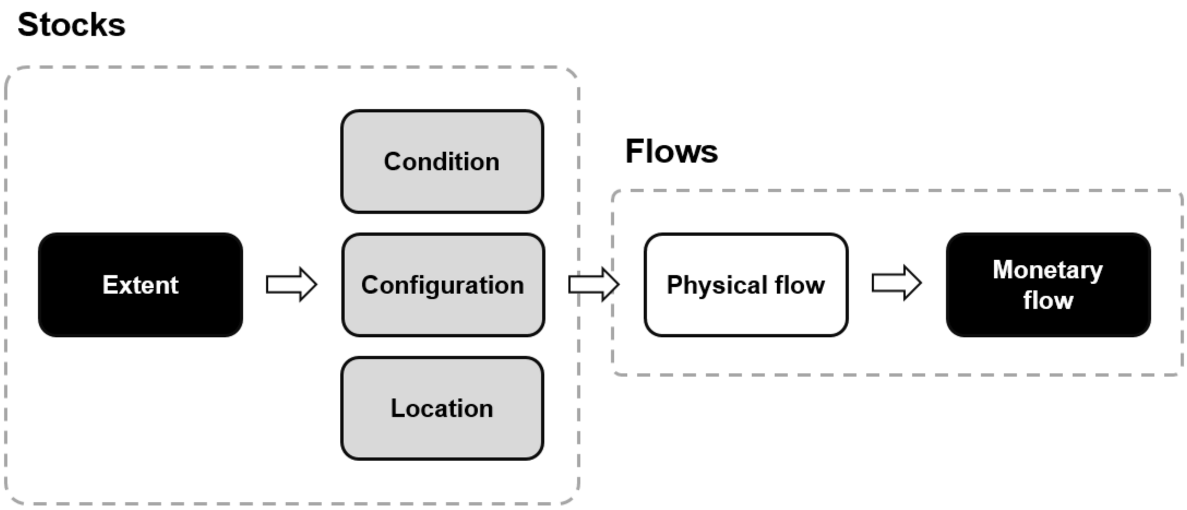

Figure 3.4: The proposed sequence for natural capital valuations, adapted from Philips (2017).

This work demonstrates that farm scale monetary valuations produced using a land cover and benefit transfer approach can be highly sensitive to the characteristics of the land cover data used, due to variations in the extent of different land covers recorded (black boxes, Figure 3.4). Uncertainty is therefore introduced immediately in the first step of this framework. Future work should consider the condition, configuration and location of stocks (grey boxes, Figure 3.4) to more accurately value flows of services in physical and monetary terms, and assess the uncertainties involved in their measurement.

Condition, otherwise referred to as quality, includes the attributes of an asset, such as water quality and soil carbon content. Here it is assumed that all assets are functioning equally and provide the same services, using a simple single value per unit area approach. For valuations derived from the UK natural capital accounts, benefits from one scale (national) are applied to a very different scale (individual farms), meaning assets are given a value averaged across all assets across the UK. For valuations derived from the literature, assets are assumed to be functioning the same as those in the study area that values were derived from. These valuations will include services that are not applicable to all sites. As an example, valuations for farmland include values for educational visits and recreational time spent at the habitat, however not all sites used in this study allow these. Troy and Wilson (2006) give the example of the recreational value of coniferous forest, which may yield different results if the forest is on public or private land due to differences in access. Where services are provided by an asset, the magnitude of this provision will be affected by a range of factors. A simple example would be how the value of woodland as a wildlife habitat changes depending on its age, tree species composition and the health of the trees. These ecosystem service indicators are likely to be complex. In this example, different wildlife will prefer different conditions, and different species of wildlife could be considered more or less valuable in different locations. Management practices, such as the distinction between organic and conventional farming, will also impact the condition of assets. Studies have shown that organic farming has a positive effect on biodiversity when compared with conventional techniques for example (Bengtsson et al., 2005; Winqvist et al., 2011; Tuck et al., 2014), and on the provision of ecosystem services including soil carbon storage (Drinkwater et al., 1995) and in certain landscapes biological control (Winqvist et al., 2011). Farm management practices may also result in negative externalities, or ecosystem disservices, such as eutrophication caused by fertilizer usage. These are not considered in this work, which focuses on the valuation of ecosystem services, but would need to form part of a more comprehensive assessment of the environmental impacts of agriculture.

The configuration of an asset will also have an impact, with factors such as fragmentation and connectivity being known to affect the value of a habitat for biodiversity. Similarly, the recreational value of a tree will vary depending on whether it stands alone or is part of a wider woodland. Measures of configuration are not currently included in the UK natural capital accounts as they are noted to be challenging to compile (Philips, 2017). While configuration can be considered as the position of an asset in relation to itself, location refers to the position of an asset in relation to other assets or features. Dales et al. (2014) note how knowledge of the locations of an asset and beneficiaries of ecosystem services would be advantageous. For example, a habitat supporting pollinators may be more valuable when located near to certain agricultural crops. Similarly, a woodland may have more value for recreation when located near areas of population (Philips, 2017).

Not all ecosystem services have been accounted for in this study, and it is unlikely that we will ever be able to accurately quantify and appreciate the full extent of all the benefits provided by nature. Any valuations produced should therefore be seen as partial, or a lower bound, only. Some natural capital assets and the services they provide are also not accounted for due to the spatial and thematic resolution of the data used. The urban classes in both LCM and CLC include green space such as gardens and parks, as well as artificial surfaces. As the ecosystem service value of urban areas was assumed to be zero, the value of urban green space has not been accounted for, although it is known to be significant (Anderson, 2018; Willis and Petrokofsky, 2017). This being said, as the farms examined in this study are predominantly rural and agricultural in nature with limited (if any) urban areas, the impact of this would be expected to be small.

Augmenting the land cover maps used in this study with additional more accurate or detailed layers would improve the accuracy of valuations. Of the three datasets tested, LCM was found to be the most accurate, with an overall accuracy ranging from 78% to 89%, indicating there is room for improvement at all sites. Woodland, as a significant contributor to valuations at most sites, and land covers such as bog and heather which were classified poorly in existing datasets could be valuable targets for future work. For example, the CEH Woody Linear Features Framework (Scholefield et al., 2016) could be used to include hedges and narrow lines of trees, which are not included in any of the land cover datasets used in this study. Alternatively, land cover data could be optimised before use. Dales et al. (2014) suggest that local practitioners could clean the data before analysis is carried out. Other data sources, such as aerial photography or topographic mapping could also be used to check and update land cover data, assuming data were available for the appropriate date. Both options however would require additional time and effort to be devoted to the creation of valuations. Finally, as well as improving the accuracy of surface land cover data, valuations could also be improved by accounting for features in the subsurface. As an example, only surface water, and not ground water is valued here, but incorporating this component would require the use of additional data.

3.5 Conclusion

This work demonstrates a land cover and benefit transfer-based approach to natural capital and ecosystem service valuation on individual UK farms. The suitability of three widely available land cover products was assessed. It was found that the varying spatial and thematic characteristics of these products can have a significant impact on final valuations. LCM was found to be the most accurate at most sites, and its use as the input spatial data also results in the highest valuations at the majority of farms. This is partly due to the ability of LCM to detect small patches of land cover, especially trees, scattered through the landscape, inflating valuations. The impact of this is greatest at sites where the landscape is fragmented, and less where it is more homogenous. The presence of bog and heather also make a notable contribution to valuations at some sites, especially the upland hill farm studied. These land covers appear to be mapped less accurately, with disagreement between the datasets examined. As well as spatial resolution, thematic resolution is also important, as it is difficult to assign the land *use* classes of CLC and mosaic classes of CCI-LC to suitable values from primary studies.

Using a land cover and benefit-transfer approach allows ecosystem service valuations to be produced rapidly, and at little or in some cases no cost. However, significant uncertainties are acknowledged, especially regarding the benefit transfer process. We describe a framework for future work that also accounts for the condition, configuration and location of natural capital assets to improve the accuracy of valuations produced, while the integration of additional, more accurate and detailed land cover data has the potential to reduce errors associated with the extent of assets.

3.6 Acknowledgements

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3.7 Addendum to section 3.2.1: selection of the five study areas

When identifying study areas for inclusion in this work, every effort was made to ensure that these had a range of geographical locations, farming styles, and sizes.

The managers of Sites 1 and 3 volunteered to participate in the study as these estates had an existing relationship with LUC through their consulting work. Similarly, Site 2 had previously collaborated with Lancaster University on past projects.

For the remaining study areas, a call for participants was made through the Country Land and Business Association (CLA) to find sites that met criteria not already covered by Sites 1, 2 or 3. These were:

- Arable in the East of England
- Pasture / Dairy farm in the Midlands
- Mixed use - pasture, arable, forestry in the South or South West

Following this call for participants, the managers of two estates volunteered to take part in the study. These were Site 4 (dairy, midlands), and Site 5 (mixed use, south).

4. Achieving national scale targets for carbon sequestration through afforestation: Geospatial assessment of feasibility and policy implications

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This chapter is a replication of a constituent paper of this research that was published in Environmental Science and Policy.

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Abstract

To explore the feasibility of meeting recently proposed large-scale tree planting targets, a UK wide assessment of land available for afforestation was carried out, considering a range of physical, environmental and policy constraints in three hypothetical planting scenarios. Results show there is sufficient space to meet these targets in all three scenarios, even if planting is prevented on good to moderate quality agricultural land and within protected areas. However, this would require planting on a large proportion of unconstrained land, especially for the more ambitious targets, which is unevenly distributed across the UK. This would limit opportunities for spatially targeting woodland creation, which may restrict the provision of additional ecosystem services such as air pollution control and recreation, and induce widespread

negative impacts on landscapes and communities. In order to overcome these limitations, relaxing constraints, such as permitting afforestation of higher quality agricultural land, will need to be considered. Meeting many of the proposed afforestation targets would result in a transformational change in British land cover, which could replace or significantly impact the business models of tens of thousands of farms, and see the replacement of hundreds of thousands to millions of hectares of grassland, arable and horticultural land and other land covers. This would require rates of planting that far exceed those seen historically. Policies and mechanisms that could be used to encourage this planting, both by the state and private sectors, are discussed.

4.1 Introduction

Greenhouse gas emissions from human activity are giving rise to what is now being described as a 'climate emergency' (Ripple et al., 2020). These emissions are estimated to have caused warming of approximately 1° C above pre-industrial levels, and this is likely to reach 1.5° C, described as 'dangerous climate change' (Lewis, 2016), between 2030 and 2052 if they continue at the current rate (IPCC, 2018). Climate-related risks to health, livelihoods, food security, water supply, human security and economic growth are projected to increase with global warming of 1.5° C, and increase further with warming of 2° C (IPCC, 2018). It has been suggested that Earth is approaching thresholds that if crossed could cause continued warming and a 'Hothouse Earth' scenario, with consequences for ecosystems, economies and society (Steffen et al., 2018).

Stabilization of the Earth system to avoid risks related to global climate change will require rapid decarbonisation through both cuts to greenhouse gas emissions, and the protection and enhancement of biosphere carbon sinks (Rockström et al., 2017). At a global scale, afforestation and reforestation has the potential to be our single largest natural climate solution (Griscom et al., 2017; Bastin et al., 2019), and recent years have seen numerous international policies and agreements with the aim of protecting and extending the world's forests. Article 5 of the Paris Agreement encourages parties to conserve and enhance carbon sinks, including forests (United Nations, 2015), while the Bonn Challenge (bonnchallenge.org), initiated by IUCN and the Government of Germany in 2011, has gathered 62 commitments by governments and other organisations to restore over 170 million hectares of woodland by 2030 to provide carbon sequestration and other benefits. Carbon credits, introduced by the Kyoto protocol, allow for carbon sequestration to offset emission elsewhere, providing funding for the developing of forestry projects (Kula, 2010).

Within Europe, the European Union aims to cut emissions by 40% by 2030 (European Commission, 2014), including emissions and removals by the land use, land use change and forestry sector (Regulation (EU) 2018/841 of the European Parliament and of the Council (2018)), and to be climate neutral by 2050 (European Union, 2020), and has proposed that this be made law (European Commission, 2017). Although the EU does not have a common forestry policy, the EU Forest Strategy recognizes the ecosystem services provided by forests, including climate change mitigation (European Commission, 2013), and funding for forestry and afforestation is provided through the Common Agricultural Policy. Individual nations also have their own targets for afforestation with varying levels of ambition (Department of Communications Climate Action and Environment, 2019; Palaghianu, 2015; Andrasevits et al., 2005).

The UK is the first major economy to pass net zero emission laws, which require it to bring all greenhouse gas emissions to net zero by 2050 (Climate Change Act 2008 (c. 27) (as amended)), as recommended by the Committee on Climate Change, the UK's independent climate change advisory body. To achieve this, the Committee recommends planting 30,000 ha of woodland per year, along with an increase in woodland management, to increase the net forestry sink to 22 $MtCO_2e$ per year by 2050 (Committee on Climate Change, 2019). Numerous other targets for woodland creation in the UK have also been proposed in recent years by bodies including learned societies, charities and government departments. These targets, which largely aim to deliver carbon sequestration and storage to various degrees and in various timeframes, advocate for the establishment of hundreds of thousands to millions of hectares of new woodland within the next 30 to 80 years. Zero Carbon Britain for example suggests planting 3 million ha of woodland to achieve carbon neutrality by 2030 (Centre for Alternative Technology, 2013), while The Royal Society and Royal Academy of Engineering propose planting 1.2 Mha of land by 2050 (Royal Society and Royal Academy of Engineering, 2018). Despite this prevalence of targets, less has been said of where these trees could or should be planted, the mechanisms for doing so, and the potential environmental and societal impacts this transformational change in British land cover could have.

Woodland cover in the UK is low by European standards (FAO, 2015). It currently stands at 3.19 million hectares, or 13% of total land area (Forestry Commission, 2019a), up from a low of approximately 5% at the start of the 20th century (Aldhous, 1997). Inappropriate siting of forests has the potential to cause environmental and ecological damage (Warren, 2000; Stroud et al., 2015; Sloan et al., 2018). Farmers are also often reluctant to convert productive land to forestry (Lawrence and Dandy, 2014), while a shortage of agricultural land in the UK is projected by 2030 (CISL, 2014). Therefore, further expansion of woodland area is constrained by a number of policy and environmental considerations, as well as by the availability of physically suitable land for trees to grow upon.

To date there has been no UK-focused assessment of space available for tree planting. Several studies have focused on identifying land available for afforestation at a global scale. Nilsson and Schopfhauser (1995) aggregated estimates of suitable and available land for plantations in the world's regions, finding 345 Mha available for the purpose of sequestering carbon, although this did not take spatial issues into consideration. Benítez et al. (2007) used a spatial approach, assuming planting was possible on certain types of land cover (e.g. shrublands), but removing areas subject to various constraints, such as highly productive agricultural land, to identify land available for afforestation and reforestation at a spatial resolution of 0.5°. A similar process was used by Zomer et al. (2008) at a higher spatial resolution of 500 m; however, they only considered developing countries. More recently, Bastin et al. (2019) used measurements of tree cover in protected areas with machine learning algorithms to map the tree cover that could potentially exist globally with minimal human activity, accounting for climatic and environmental conditions, with a spatial resolution of 30 arc seconds.

While these studies provide dramatic examples of the potential for afforestation at global and continental scales, they are of limited value for implementing it at the national scale. This is due to the coarse resolution of the inputs and constraints considered, which do not adequately reflect those of relevance to individual countries. In Europe, some analysis has been carried out at a national and regional scale. For example, Farrelly and Gallagher (2015) found 4.65 million hectares of land in the Republic of Ireland to be potentially suitable for forestry, accounting for a range of physical and environmental constraints. Within the UK, similar assessments have been carried out for Scotland (Sing et al., 2013) and Snowdonia National Park in Wales (Gkaraveli et al., 2004); however, the lack of a detailed and comprehensive assessment of space available for afforestation means the feasibility of meeting UK wide planting targets is uncertain.

Not only does the availability of space available for these targets need to be considered, but also where planting could take place and what impacts this would have. While afforestation can bring public benefits, the level of which vary spatially (Bateman et al., 2014; Gimona and Van Der Horst, 2007; Bailey et al., 2006), inappropriate siting of woodland has the potential to negatively impact local communities and landscapes (van der Horst, 2006; Ní Dhubháin et al., 2009). Meeting these targets will also require rates of planting far beyond those that have been achieved in recent years (Forestry Commission, 2019a), and appropriate mechanisms and policies for this will need to be identified.

The aim of this paper is to assess the feasibility of achieving large-scale afforestation targets in the UK. To do this we address the following objectives:

- Provide the first comprehensive collation of targets for woodland creation in UK.
- Present the first high resolution UK-wide assessment of space available for woodland creation, accounting for a variety of physical, environmental and policy constraints in three hypothetical planting scenarios.
- Using these scenarios, explore spatial patterns of afforestation that could occur under different land use policies, and the potential environmental and societal impacts this could have.

- With reference to current and historic rates of planting, explore mechanisms that could be used to enable large scale afforestation, and the challenges associated with this.

4.2 Materials and methods

4.2.1 Identifying afforestation targets

Recent targets for woodland creation in the UK were identified from the literature. Where the target was given as an area, the annual rate of planting required to achieve this was calculated from the given start date and end date. If no start date was given, this was assumed to be the year the target was published. Where this target was given relative to the current woodland area, such as a doubling of this, it is assumed the current area of woodland in the UK is 13% (3.19 million hectares) (Forestry Commission, 2019a) unless given otherwise. Conversely, where the target was given as an annual rate of planting, the area of woodland this would result in was calculated from the given start date and end date.

4.2.2 Identifying constraints on woodland creation

To identify locations where woodland planting would be possible, a number of constraints were considered (Table 4.1). These constraints were assigned to three broad categories. The first category (‘physical constraints’) includes land already covered in woodland and land where physical factors would make large-scale woodland planting impossible or prohibitively difficult, such as water bodies. Here we define the natural treeline as 600 m, its approximate location in England (Backshall, 2001), although the exact elevation will depend on local variations in temperature, shelter and humidity (Ratcliffe and Thompson, 1988; Pearsall, 1989).

Table 4.1: Constraints and data sources used.

Constraint	Devolved Administration	Source	Product
Base			
Land area	England, Scotland, Wales	Ordnance Survey (OS)	Boundary-Line
	Northern Ireland	Ordnance Survey of Northern Ireland (OSNI)	Open Data Largescale Boundaries – NI Outline
Physical constraints			
Existing woodland	England, Scotland, Wales	Forest Research	National Forest Inventory (NFI)
	Northern Ireland	UK Centre for Ecology & Hydrology (UKCEH)	Land Cover Map 2015 (LCM2015)
Water, rock and coastal sediment	England, Scotland, Wales, Northern Ireland	UK Centre for Ecology & Hydrology	Land Cover Map 2015
Climatic treeline (600 m)	England, Scotland, Wales	Ordnance Survey	OS Terrain 50
	Northern Ireland	Ordnance Survey of Northern Ireland	Open Data 50m Digital Terrain Model
Urban and suburban areas	England, Scotland, Wales, Northern Ireland	UK Centre for Ecology & Hydrology	Land Cover Map 2015
Environmental Constraints			
Peat	England, Scotland, Wales, Northern Ireland	British Geological Survey (BGS)	Geology 625k
Bog	England, Scotland, Wales, Northern Ireland	UK Centre for Ecology and Hydrology	Land Cover Map 2015

Policy constraints			
Protected areas	England	Natural England, Historic England	Site boundaries. See Table A2.
	Scotland	Scottish Government, Scottish Natural Heritage, Historic Environment Scotland	
	Wales	Natural Resources Wales, Cadw	
	Northern Ireland	Department of Agriculture, Environment and Rural Affairs (Daera), Department for Communities	
Agricultural land	England	Natural England	Provisional Agricultural Land Classification (ALC)
	Scotland	Hutton Institute	Land Capability for Agriculture Scotland
	Wales	Welsh Government	Predictive Agricultural Land Classification
	Northern Ireland	UK Centre for Ecology & Hydrology	Land Cover Map 2015

The second category ('environmental constraints') includes areas where planting is possible, but where doing so would cause environmental harm. This category includes both peat and bog. While some studies have shown a potential for some afforested peatland to act as a carbon sink, the dynamics of carbon sequestration from peatland afforestation are complex and hindered by a lack of data (Crane, 2020), and carbon benefits from woodland creation are generally greatest on soils with low levels of organic matter (Forestry Commission, 2017). Past planting on peatland habitats has caused significant environmental damage (Warren, 2000; Stroud et al., 2015) and the UK Forestry Standard now prohibits planting on peat exceeding 50 cm in depth, and on sites that would compromise the hydrology of bog or wetland habitats (Forestry Commission, 2017). Recent evidence suggests that rewetting and full restoration of wetlands is an effective means of generating carbon sinks (Evans et al., 2021) and, should be carried out promptly to have the most beneficial effects during predicted peak warming (Günther et al., 2020).

The third category ('policy constraints') includes areas where planting would also be possible, but may be restricted for planning or policy reasons. This category includes both protected or designated areas (Table A2) and higher quality agricultural land. Practically, planting within a protected area such as a National Park or Area of Outstanding Natural Beauty ('AONB') is difficult as it generally requires completion of an Environmental Impact Assessment (Forestry Commission, 2019b). Historically, forestry in the UK has competed with agriculture for space (Edlin, 1969). Today, farmers are often reluctant to convert productive land to forestry (Lawrence and Dandy, 2014), and a shortfall of farmland in the UK is projected by 2030 (CISL, 2014). For this reason, higher quality agricultural land is included as a constraint on woodland planting.

When collecting data to map some of the constraints, it was found that some parts of the UK were not covered by datasets that otherwise extended across the majority of the country. In these cases, alternative data sources or proxies had to be used to achieve full coverage of the constraint. Notably, there is no unified agricultural land classification for the UK. For Great Britain, agricultural land classifications for England, Scotland and Wales were harmonised (Table 4.2). An agricultural land classification map was not available for Northern Ireland, hence the Arable and Horticulture class from CEH LCM2015 (Centre for Ecology & Hydrology, 2017) was used instead, under the assumption that if land is currently being used for cropping rather than pasture, it is likely to be of good quality. Furthermore, the British Geological Survey Geology 625k dataset was used to identify peat, although this does not to extend to the western limit of Northern Ireland. The LCM2015 Bog class, which maps ericaceous, herbaceous and mossy swards in areas with a peat depth greater than 50 cm does, however, have national coverage and is also included in the environmental constraints. Issues surrounding data access and availability are discussed further in Section 4.3.4.

Table 4.2: Harmonisation of UK agricultural land classification maps, and their use in scenarios.

	Nation and data source			
	England	**Scotland**	**Wales**	**Northern Ireland**
Scenario status	Provisional Agricultural Land Classification (MAFF, 1988)	Land Capability for Agriculture, Scotland (The James Hutton Institute, 1981)	Predictive Agricultural Land Classification (Welsh Government, 2017)	CEH Land Cover Map 2015 (Centre for Ecology & Hydrology, 2017)
Planting not permitted	Grade 1 'Excellent quality'	Class 1 'Land capable of producing a very wide range of crops'	Grade 1 'Excellent quality'	Existing arable and horticultural land
	Grade 2 'Very good quality'	Class 2 'Land capable of producing a wide range of crops'	Grade 2 'Very good quality'	
Planting permitted in Agricultural Sacrifice Scenario, but not Restrictive or Protected Areas Sacrifice Scenarios	Grade 3 'Good to moderate quality'	Class 3.1 'Land capable of producing a moderate range of crops'	Grade 3a 'Good quality'	
		Class 3.2 'Land capable of producing a moderate range of crops'	Grade 3b 'Moderate quality'	
Planting permitted	Grade 4 and below	Class 4.1 and below	Grade 4 and below	Non arable and horticultural land

4.2.3 Construction of scenarios

The physical, environmental and policy constraints were used in combination to form three hypothetical planting scenarios (Table 4.3). In the first, referred to as the *Restrictive Scenario*, all constraints are used, and planting is not permitted on ALC grade 3 ('good to moderate quality') or above (England and Wales), Land Capability for Agriculture (LCA) class 3.2 or above (Scotland), or existing Arable and Horticultural land identified by the CEH Land Cover Map (Northern Ireland). The second scenario, referred to as the *Agricultural Sacrifice Scenario,* takes a more permissive approach to planting on agricultural land, using the same constraints as the Restrictive Scenario but also allowing for planting on ALC grade 3 (England and Wales) or LCA class 3.1 and 3.2 (Scotland) land that is not ruled out by other constraints. The final scenario, referred to as the *Protected Areas Sacrifice Scenario*, uses the same constraints as the Restrictive Scenario, but also allows planting in any protected or designated areas that are not ruled out by the other constraints. Note: in all scenarios, planting is not permitted on current arable and horticultural land within Northern Ireland. The three scenarios were used to represent contrasting and diverse land use strategies spanning a range of possible approaches and demonstrating their implications, as a basis for informing future policy development which may favour one of these scenarios or a mixture of approaches.

Table 4.3: Differences in constraints used in the three scenarios. Note that all other constraints not included in the table were applied in all scenarios.

	Restrictive Scenario	Agricultural Sacrifice Scenario	Protected Areas Sacrifice Scenario
Protected areas	Planting **not** permitted	Planting **not** permitted	Planting permitted
Agricultural land	Planting **not** permitted on:	Planting **not** permitted on:	Planting **not** permitted on:
England	ALC grade 3 or above	ALC grade 2 or above	ALC grade 3 or above
Scotland	LCA class 3.2 or above	LCA class 2 or above	LCA class 3.2 or above
Wales	ALC grade 3b or above	ALC grade 2 or above	ALC grade 3b or above
Northern Ireland	Existing arable and horticultural land	Existing arable and horticultural land	Existing arable and horticultural land

4.2.4 Analysis

All data input layers were converted to a 10 m resolution raster grid for use in the analysis. These layers primarily defined constraints, where planting is either possible, or permitted under a scenario, or not. The approach used was a subtractive, binary overlay methodology. The UK land area was used as the initial base layer, defining the spatial extent over which planting may potentially occur. Constraints, which define areas where planting cannot or should not take place, were then overlaid on the base layer, with unavailable land removed accordingly. The result is a map identifying land covered by one or more category of constraint described in Section 4.2.2 and therefore unavailable for planting, and remaining land, which is not covered by a constraint, and therefore available for afforestation. The CEH LCM2015 25 metre resolution raster product was used to identify current land cover in the UK that could be lost if afforested.

4.3 Results and discussion

4.3.1 Targets for woodland creation in the UK

Twelve targets for woodland creation were identified from six groups and organisations (Table 4.4). These range from 265,000 ha by 2050 (Committee on Climate Change Low Ambition), to 4,000,000 ha by 2100 (Committee on Climate Change High Ambition). The planting rates required to achieve these targets range from 9,200 ha/yr until 2050 (Committee on Climate Change Low Ambition), to 176,000 ha/yr until 2030 (Zero Carbon Britain: Rethinking the Future).

Table 4.4: Selected UK wide woodland planting targets. The lowest and highest targets are highlighted.

Scheme / Report	Target (ha)	Average annual planting rate (ha/yr)	Start	End	Source
Zero Carbon Britain: Rethinking the Future	3,000,000	176,000[1]	2013[2]	2030	Centre for Alternative Technology (2013)
More Trees Please	3,190,000[3]	123,000[1]	2019[2]	2045	Friends of the Earth (2019)
Greenhouse Gas Removal. Report by the UK Royal Society and Royal Academy of Engineering	1,200,000	37,500[1]	2018[2]	2050	Royal Society and Royal Academy of Engineering (2018)
Committee on Climate Change (Low Ambition)	**265,000**[4]	9,200	2016	2050	Committee on Climate Change (2018) and Thomson et al. (2018)
	724,000[4]	9,200	2016	2100	
Committee on Climate Change (Medium Ambition)	898,000[4]	31,000	2016	2050	Committee on Climate Change (2018) and Thomson et al. (2018)
	2,448,000[4]	31,000	2016	2100	
Committee on Climate Change (High Ambition)	1,477,000[4]	50,000	2016	2050	Committee on Climate Change (2018) and Thomson et al. (2018)
	3,977,000[4]	50,000	2016	2100	
Committee on Climate Change (Net Zero)	970,000[3]	30,000	2019[2]	2050	Committee on Climate Change (2019)
Committee on Climate Change (Net Zero Speculative)	1,455,000[3]	50,000	2019[2]	2050	Committee on Climate Change (2019)

Keeping it cool: How the UK can end its contribution to climate change	1,200,000[5]	40,000	2020	2050	Vivid Economics and WWF (2018)

[1] Planting rate calculated from published target area, and start and end year.
[2] No start date given, date of publication used.
[3] Assumes current UK woodland area of 13%.
[4] Assumes current UK woodland area of 15%.
[5] Target area calculated from published planting rate, and start and end year.

4.3.2 Constraints on woodland creation in the UK

The areas covered by each of the eight constraints on woodland planting were calculated (Table 4.5, Figure 4.1). Across the whole of the UK, policy constraints cover the greatest area, with nearly 16 million ha covered by one or more policy constraint if planting is prevented on ALC grade 3 and LCA grade 3.1 and 3.2 land (Restrictive Scenario), and over 9 million if it is permitted (Agricultural Sacrifice Scenario). There is significant variability between nations. In England, the most extensive constraint is agricultural land, with ALC grades 1 to 3 covering 8.5 million hectares, 65% of its total land area. In Scotland, the extent of good quality agricultural land is far lower, with just 2% of its land having an LCA class of 2 or above, and 17% class 3.2 above. Here protected areas form the single largest constraint, with nearly a third of the country having a protected designation of some form. The majority of environmental constraints (peat and bog) also lie within Scotland, with 1.2 million hectares of these covering 15% of the country. Northern Ireland has a similarly high proportion of land covered by environmental constraints, at 14%.

Table 4.5: Area covered by each physical, policy and environment constraint. The sum of the areas of each individual constraint will exceed the area of each country, due to overlapping of constraints. The subtotal for each category records the area covered by one or more constraint, accounting for overlapping.

		Area (ha)				
	Inset (Figure 4.1)	**UK**	**England**	**Scotland**	**Wales**	**Northern Ireland**
UK land area	**-**	24,366,167	13,046,152	7,881,022	2,078,202	1,360,791
		Physical constraints				
Existing woodland	a	3,135,900	1,294,952	1,416,162	305,140	119,646
Water, rock, coastal sediment	b	548,052	170,544	335,180	31,866	10,462
Climatic treeline	c	547,613	41,325	483,533	22,027	728
Urban and suburban	d	1,765,110	1,422,121	179,913	105,609	57,467
Total		5,817,400	2,882,553	2,287,712	458,834	188,301
		Policy constraints				
Protected areas	e	7,132,201	3,678,497	2,458,181	622,052	373,471
Agricultural land (Restrictive Scenario and Protected Areas Sacrifice Scenario)	f	10,714,376	8,486,840	1,324,618	807,199	95,719
Agricultural land (Agricultural Sacrifice Scenario)	f	2,599,117	2,202,271	178,635	122,492	95,719
Total (Restrictive Scenario and Protected Areas Sacrifice Scenario)		15,991,647	10,575,565	3,685,493	1,286,777	443,812
Total (Agricultural Sacrifice Scenario)		9,435,182	5,651,072	2,623,232	717,066	443,812
		Environmental constraints				

Peat	g	1,565,591	429,236	970,345	24,903	141,107
Bog	h	962,970	196,325	648,589	26,156	91,900
Total		1,938,537	493,275	1,203,254	47,334	194,674

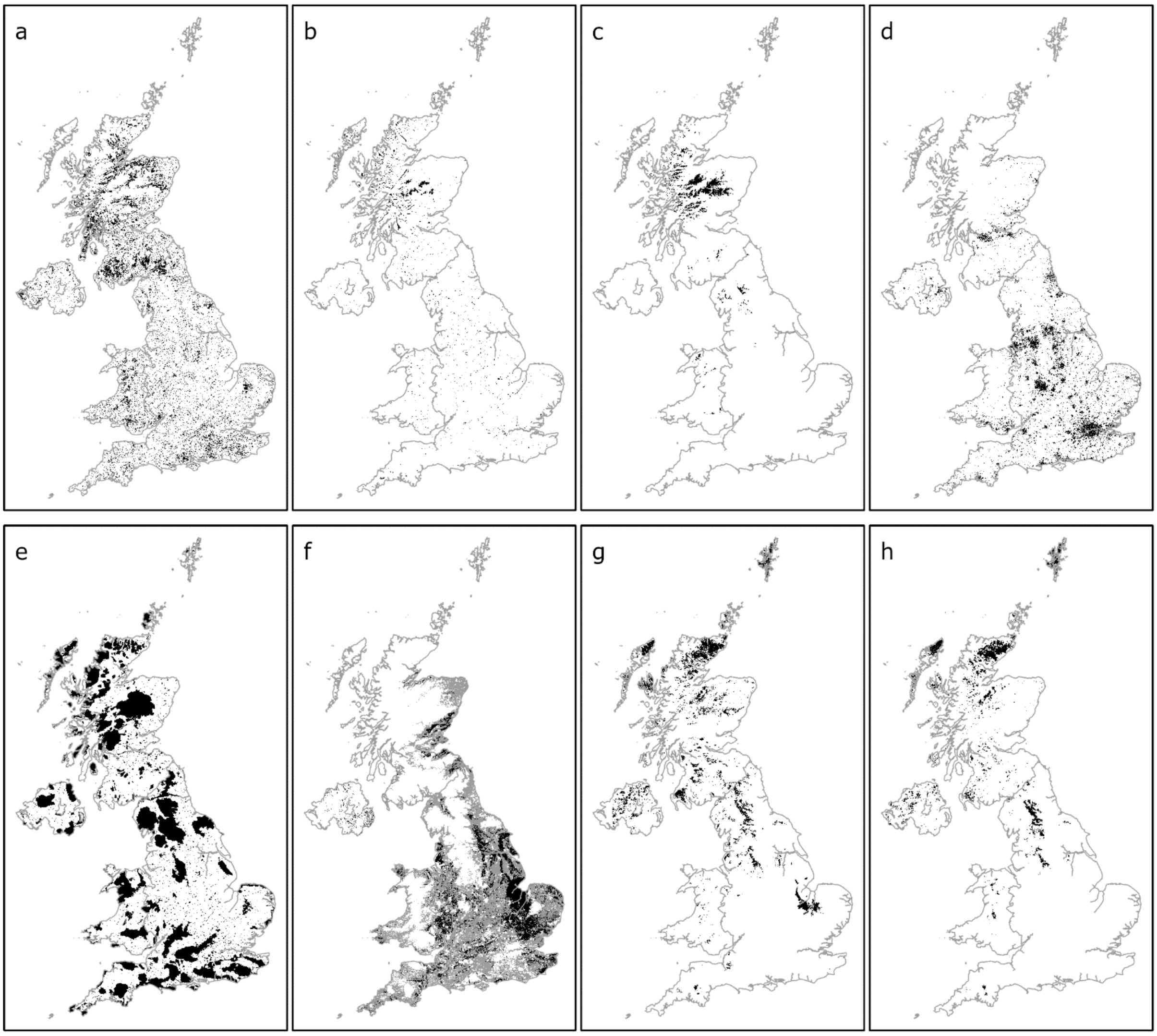

Figure 4.1: Spatial extent of the constraints identified. a.) Existing woodland. b.) Water, rock and coastal sediment. c.) Approximate climatic treeline (600 m). d.) Urban and suburban areas. e.) Protected and designated areas. f.) Agricultural land: ALC grade 1+2 (England and Wales), LCA grade 1+2 (Scotland) shown in black; ALC grade 3 (England and Wales), LCA class 3.1+3.2 (Scotland), existing arable land (Northern Ireland) shown in grey. g.) Peat. h.) Bog.

4.3.3 Land available for afforestation in the UK

Constraints on woodland planting in each of the three scenarios are shown in Figures 4.2, 4.3 and 4.4 using a colour blending style symbology, suited to showing the locations of and interactions between three classes of data (Huck et al., 2019). Here, each of the three binary categories of constraint (physical, policy and environmental) are represented by a primary colour in the subtractive colour model (cyan, magenta and yellow). The presence of a colour indicates that an area is covered by the corresponding constraint. Where an area is covered by two or more constraints, these colours overlap to produce a new colour (e.g. cyan and magenta produce blue).

Under the Restrictive Scenario, 4.7 million ha remains available for planting in the UK (Figure 4.2, Table 4.6). This is sufficient to meet even the most ambitious goal of 4 million ha of woodland by 2100; however, this would require planting nearly 85% of available land. Meeting the goal of 1 million hectares, one of a range of measures being targeted by the UK government to meet its pledge of carbon neutrality by 2050 (Department for Business Energy and Industrial Strategy, 2017), would require planting 21% of available land. With 2.3 million ha (29% of its land area) available, Scotland holds nearly half of the UK's available land. Conversely, just 9% of England – 1.2 million ha, was identified as being available under this scenario. Northern Ireland has by far the greatest proportion of land available, with just over half having the potential for woodland expansion. However, the lack of an agricultural land classification map, and the use of currently arable and horticultural land as a proxy, will likely have contributed to this outcome (Section 3.4).

The Agricultural Sacrifice Scenario (Figure 4.3, Table 4.6) is the least restrictive of those assessed, as it more than doubles the land available for planting to 10.4 million ha, more than twice the area required for the most ambitious woodland planting target, and ten times the goal of 1 million ha. The spatial distribution of available land also changes dramatically, with the greatest area of available land now being within England (5.5 million ha), rather than Scotland (3.2 million ha).

The Protected Areas Sacrifice Scenario (Figure 4.4, Table 4.6), results in more land being available for planting than in the Restrictive Scenario, but less than the Agricultural Sacrifice Scenario, with 7.8 million hectares identified. Within England, much of this is in the north, where 30% of land is suitable for afforestation, compared with 13% in the midlands and 16% in the south.

In all three scenarios tested there is sufficient space to meet even the highest woodland creation goals (Table 4.4). However, this will require the afforestation of large proportions of the land identified as being available, especially for the most ambitious targets and most restrictive scenario. This has the potential to leave little flexibility to choose where to plant, whether this is to locate new woodlands in the most suitable locations, or due to the ease or difficulty of converting land from its current use to forestry. For example, the establishment of woodlands primarily in remote upland areas, as would occur with the Restrictive Scenario, risks limiting the provision of ecosystem services such as air pollution control and recreation which vary spatially (Bateman et al., 2014; Gimona and Van Der Horst, 2007; Bailey et al., 2006), restricting the benefits other than carbon sequestration these trees could provide. Efforts to encourage farmers to afforest their land has seen little success in recent years and initiatives to promote afforestation of large proportions of their holdings will be even more problematic. Likewise, compulsory purchase of land for the establishment of state woodland is likely to prove highly controversial (discussed further in Section 4.3.6).

These findings highlight the considerations, and likely compromises, that will need to be made when planning for large scale afforestation at a national scale. While in the

Restrictive Scenario more than 80% of the UK is covered by one or more constraints, much of this is purely in the form of policy constraints – protected areas and agricultural land. Therefore, it may be preferable to allow for some planting to be undertaken in these areas, which, as the Agricultural Sacrifice and Protected Areas Sacrifice Scenarios illustrate, opens substantially more space for afforestation, potentially allowing for this planting to be better targeted spatially. These decisions will need to be considered at the full range of spatial and policy levels. Nationally, as all three scenarios show, space available for afforestation is not evenly distributed throughout the UK, and this distribution is different in each scenario. It may, for example, be considered necessary to allow for planting on agricultural land in the south of England, or protected areas in the north-west, to ensure populations in these areas have access to the benefits woodland can provide. Within the UK, nations may identify different priorities for land use. Scotland, for example, may place a higher value on protecting its comparatively low proportion of high-quality agricultural land. Finally, at a local scale afforestation can have a large impact on communities and landscapes which will need to be considered (discussed further in Section 4.3.5).

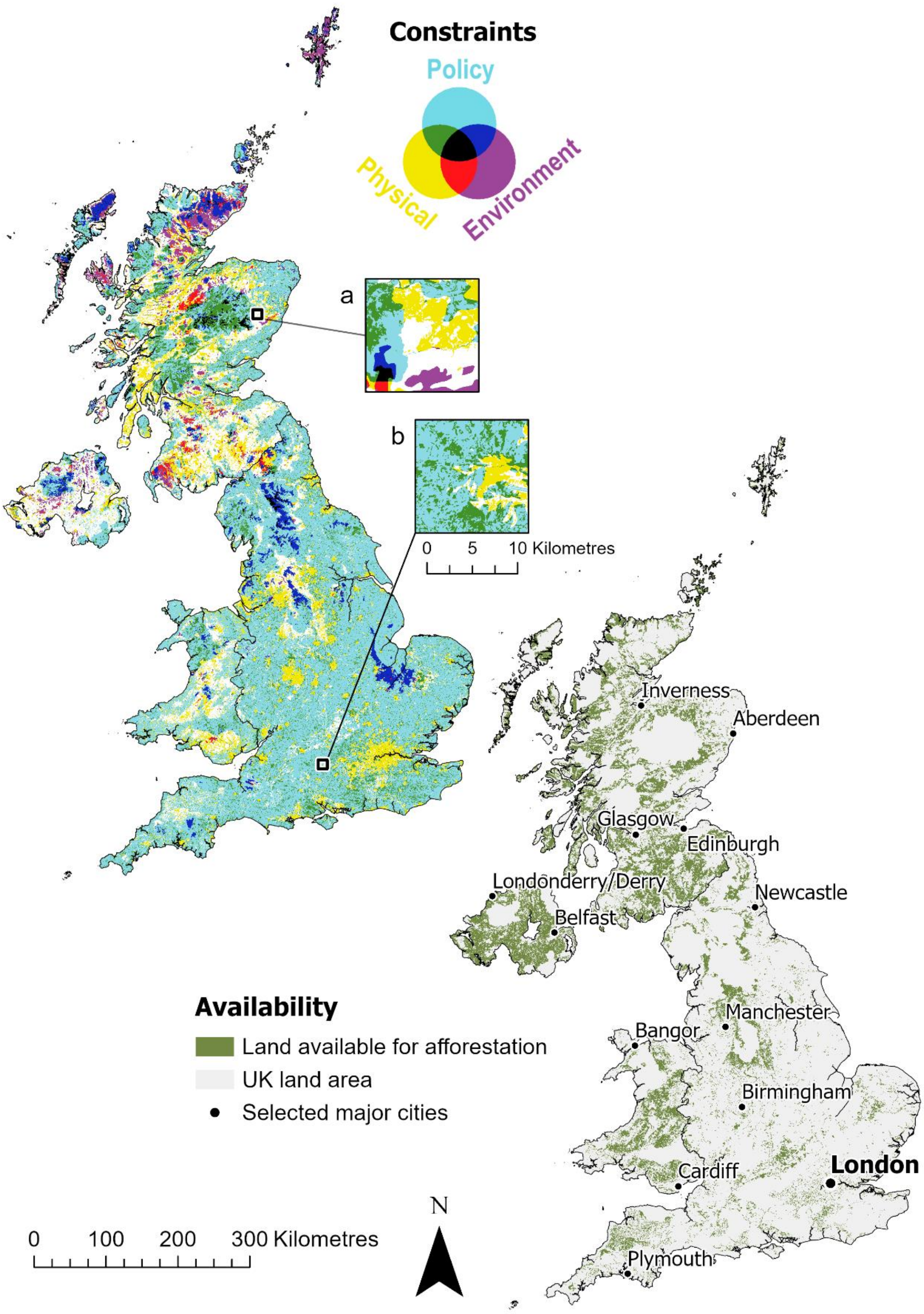

Figure 4.2: (Top left): Constraints on afforestation in the UK in the **Restrictive Scenario**. White indicates that there is no constraint, and that the land is therefore available for planting. This is displayed as green in (Bottom right): potential for afforestation. Selected major cities are included for context (© OpenStreetMap, openstreetmap.org/copyright).

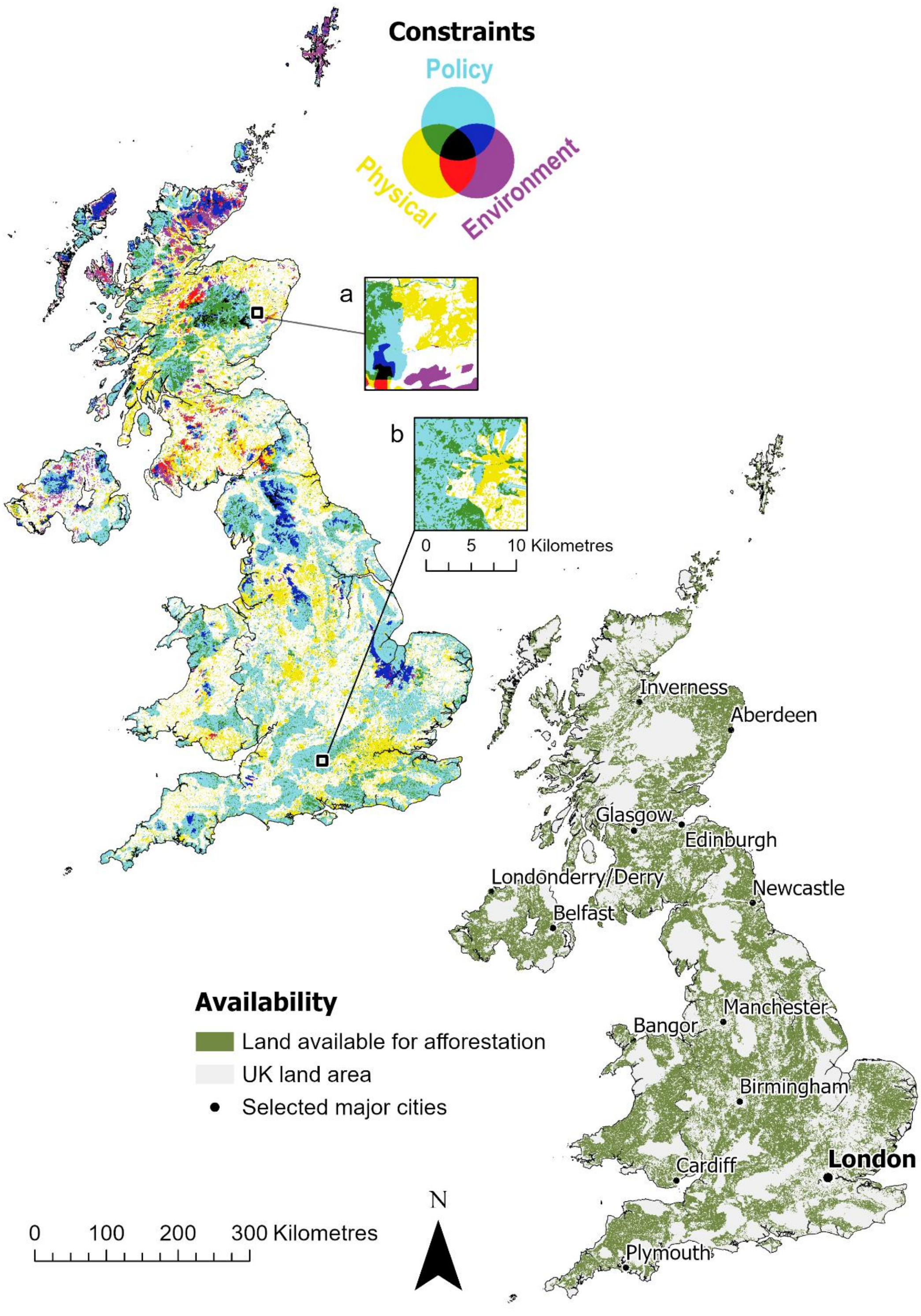

Figure 4.3: (Top left): Constraints on afforestation in the UK in the **Agricultural Sacrifice Scenario**. White indicates that there is no constraint, and that the land is therefore available for planting. This is displayed as green in (Bottom right): potential for afforestation. Selected major cities are included for context (© OpenStreetMap, openstreetmap.org/copyright).

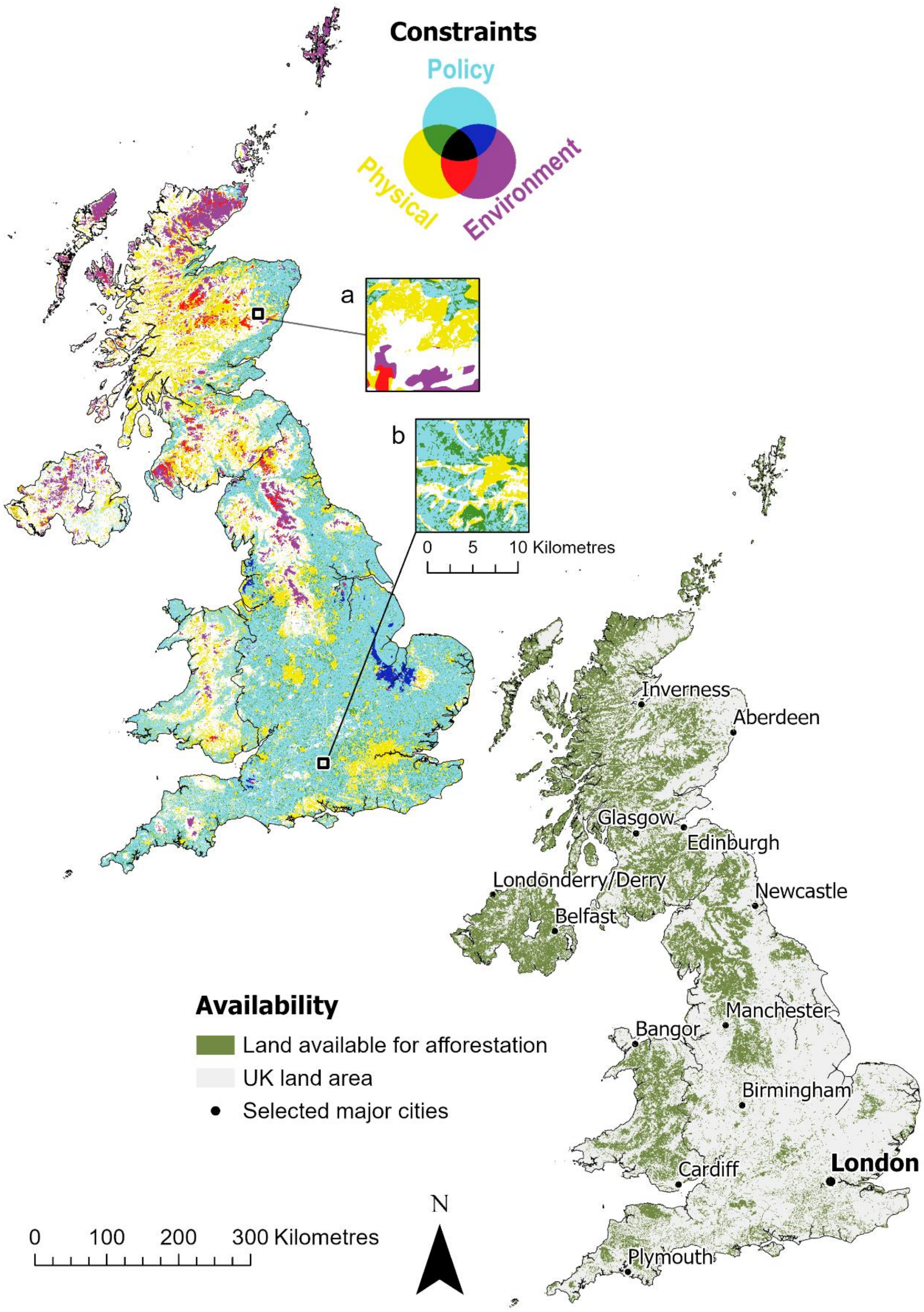

Figure 4.4: (Top left): Constraints on afforestation in the UK in the **Protected Areas Sacrifice Scenario**. White indicates that there is no constraint, and that the land is therefore available for planting. This is displayed as green in (Bottom right): potential for afforestation. Selected major cities are included for context (© OpenStreetMap, openstreetmap.org/copyright).

Table 4.6: Land available for, and constraints preventing, woodland creation in the UK under three hypothetical scenarios.

Country	Available		Physical constraint (including combined with other constraints)		Policy constraint only		Environmental constraint only		Policy & environmental constraint	
	Ha	**%**	**Ha**	**%**	**Ha**	**%**	**Ha**	**%**	**Ha**	**%**
Restrictive Scenario										
UK	4,717,880	19	5,817,400	24	12,251,353	50	694,524	3	885,014	4
England	1,230,392	9	2,882,553	22	8,504,474	65	22,776	0	405,957	3
Scotland	2,299,667	29	2,287,712	29	2,358,159	30	567,719	7	367,766	5
Wales	493,640	24	458,834	22	1,087,258	52	8,515	0	29,956	1
NI	694,181	51	188,301	14	301,462	22	95,514	7	81,335	6
Agricultural Sacrifice Scenario										
UK	10,366,648	43	5,817,400	24	6,602,581	27	724,055	3	855,481	4
England	5,445,030	42	2,882,553	22	4,289,835	33	42,822	0	385,911	3
Scotland	3,192,167	41	2,287,712	29	1,465,658	19	576,360	7	359,125	5
Wales	1,035,271	50	458,834	22	545,626	26	9,359	0	29,111	1
NI	694,180	51	188,301	14	301,462	22	95,514	7	81,334	6
Protected Areas Sacrifice Scenario										
UK	7,815,627	32	5,817,400	24	9,153,604	38	1,409,708	6	169,828	1
England	2,535,820	19	2,882,553	22	7,199,046	55	273,007	2	155,726	1
Scotland	3,558,338	45	2,287,712	29	1,099,488	14	926,346	12	9,139	0

Wales	817,966	39	458,834	22	762,931	37	37,087	2	1,383	0
NI	903,503	66	188,301	14	92,139	7	173,268	13	3,580	0

4.3.4 Data availability and access

This work represents a national scale, first look at space available in the UK for woodland creation. The analysis was limited to some extent by the lack of data available for certain areas. This is the case for the Agricultural Land Classification map in England. While the provisional (pre-1988) map is available nationally, this does not differentiate between grades 3a and 3b, which makes up 48% of land in England. Identifying 'best and most versatile' agricultural land (grades 1, 2 and 3a) is therefore not possible. While post-1988 mapping is available which does differentiate grades 3a and 3b, the coverage is very patchy. In other cases data exists but accessing this has not been possible. This is the case for the Agricultural Land Classification for Northern Ireland. For this analysis, current Arable and Horticultural land was used as a proxy, under the assumption that if the land is currently being farmed, it is likely to be of good quality. However, this is likely to underestimate the true area of good quality land for agriculture. For example, in England while 8.5 million ha of land are classified as ALC grade 3 or above, just 4.8 million ha are currently within the land cover class Arable and Horticulture. If judicious planning and policy development to promote large scale afforestation in the UK is to occur, these deficiencies in the existence, coverage and access to geospatial data need to be rectified.

4.3.5 Impacts of large-scale afforestation in the UK

All proposed woodland planting targets would see a transformational change in British land use, with large areas of land being converted to woodland, especially rough and improved grassland (Figure 4.5). With an average UK farm size of 81 ha (Defra et al., 2018), the establishment of 2,500,000 hectares of woodland (Committee on Climate Change medium ambition scenario, used as an example) would see the elimination of the equivalent to 31,000 farms, or 123,000 farms planting 25% of their land area, impacting the business models of these sites. Significant changes to land use policies and agricultural subsidies would need to be made to support this transition (discussed further in 4.3.6).

Studies have shown a preference for between 25% and 50% forest cover in a landscape, beyond this increases in forest cover are not appreciated by the public (van der Horst, 2006). Commercial forestry may also be viewed unfavourably by the public, especially in landscapes where it has not occurred historically (Ní Dhubháin et al., 2009). Therefore, care will need to be taken to plan planting appropriately at a local level, although the sheer scale of afforestation being proposed may make this a difficult task. More extensive, but less intensive tree planting may be one solution, for example through urban greening or silvoarable and silvopastoral agrofarming practices (Saunders et al., 2013), although the capacity for this in the UK would need to be assessed.

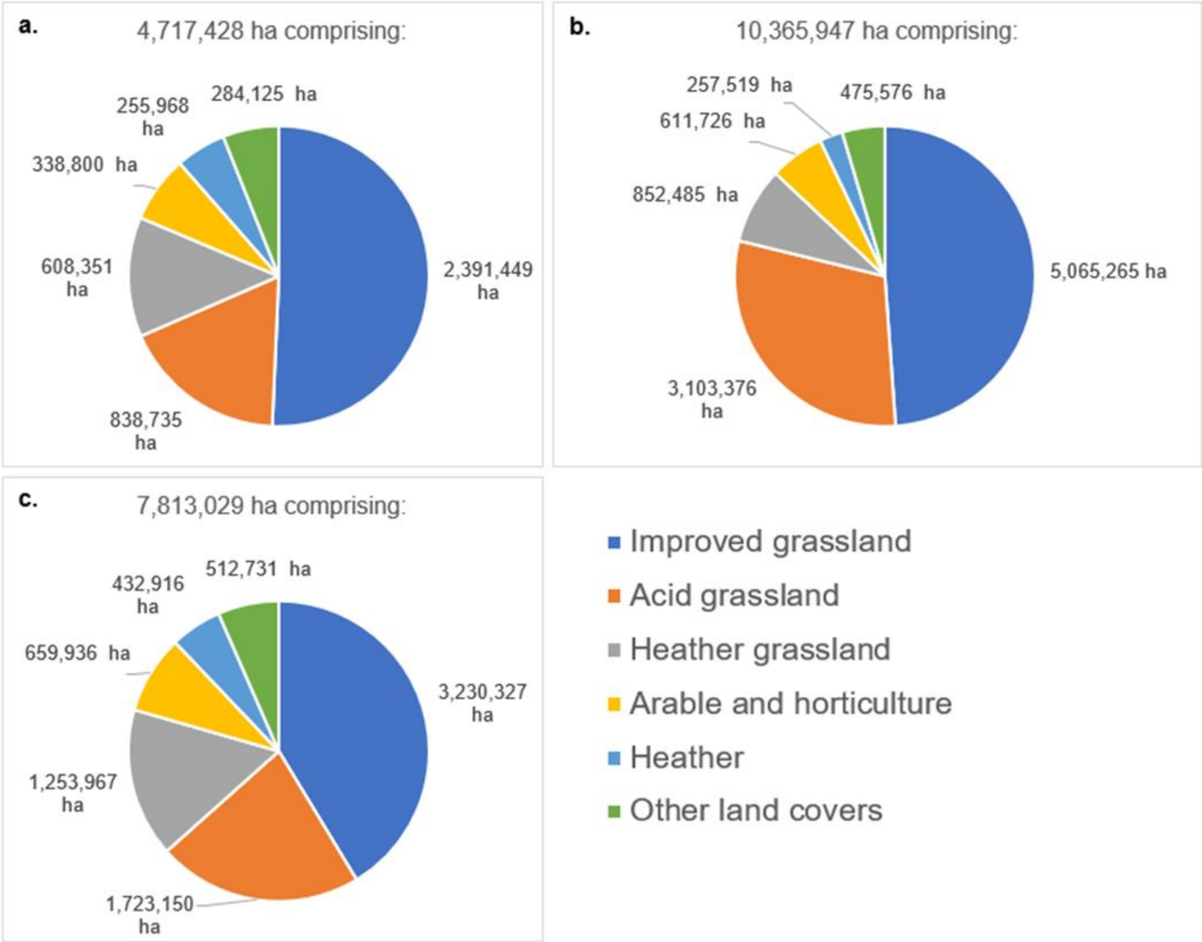

Figure 4.5: Approximate extent of different land cover types within areas found to be suitable for planting in the a.) Restrictive Scenario, b.) Agricultural Sacrifice Scenario and c.) Protected Areas Sacrifice Scenario.

4.3.6 Mechanisms for achieving proposed woodland creation goals

To meet most proposed targets, a rate of planting would be required which far exceeds that seen in recent years, or decades. Annual planting rates from the past half century range from a high of 30,270 ha in 1989, to a low of 5,440 ha in 2010 (Forest Research, 2019b). Therefore, it is clear that achieving a significant increase in planting, with some targets proposing up to 176,000 ha per year, will prove to be a substantial challenge (Figure 4.6). In order to meet this challenge, it may be prudent to learn from historical precedents. In the UK, the establishment of the Forestry Commission led to an increase in productive forest area from approximately 1.3 million ha to 2 million ha by the end of the 20th century (Aldhous, 1997). Internationally, Spain has seen 10 million hectares of woodland created since the mid-19th century (Vadell et al., 2016), while 1 million hectares were reforested in Romania in the mid-20th century (Palaghianu and Dutca, 2017). Large scale afforestation is therefore possible, and the mechanisms to achieve this are discussed further below.

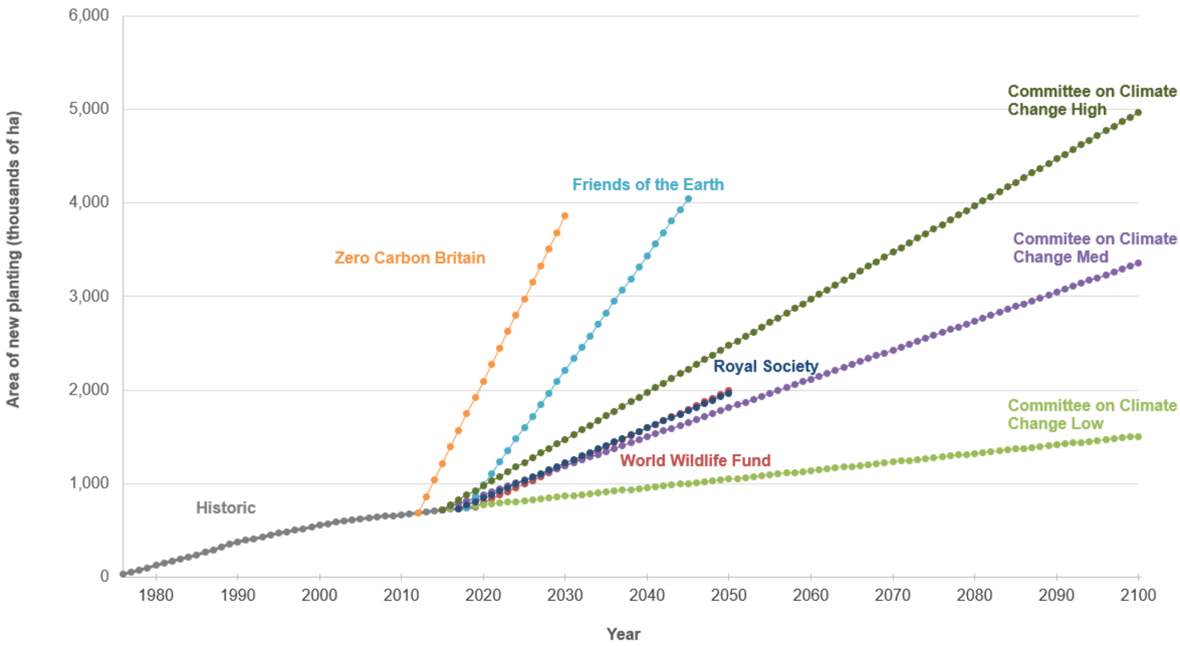

Figure 4.6: Cumulative area of historic and proposed new woodland planting in the UK. Historic values represent recorded areas from 1976 to present. Proposed values are calculated using average annual planting rate from the start/publication year through to the year identified for reaching the target by each scheme/report (see Table 4.4).

For much of its history, Britain had no formalised state forest policy, often taking a laissez faire approach to woodland management and making liberal use of cheap imports from overseas (Aldhous, 1997). As a consequence of centuries of deforestation (Smout, 2003) and little incentive for the establishment of new plantations (Aldhous, 1997), by the end of the 19th century, woodland area in Great Britain stood at around 1 million ha (Table 4.7), less than 5% of the total land area. This decline in wooded area accompanying a hands-off approach to woodland management and policy suggests that a proactive approach is required if afforestation is to increase substantially. This was recognised in 1919 when experiences from World War I led to the creation of the Forestry Commission in order to create a strategic timber reserve and lessen reliance on imports (Richards, 2003; Aldhous, 1997).

4.3.6.1 State forestry

For the first 60 years, additions to the UK woodland area came primarily from an increase in the area of the state forests, rather than private plantations (Table 4.7). This was achieved through the purchase of large areas of land for planting, such that by 1939 the Commission had become the largest landowner in Britain (Nail, 2008). This approach of governmental land acquisition offers one possible means of meeting proposed planting targets. However, while the policy was successful when implemented in the mid-20th century, that success may not necessarily be replicated if attempted today. Early expansion of state forests in the 1920s and 30s was enabled by the availability of cheap land (Forestry Commission, n.d.); however, the price of agricultural land in England has increased substantially in real terms since 1945 (Jadevicius et al., 2018). Today, much of the remaining non-forested land tends to be higher quality agricultural land, in scenic areas or in peri-urban locations, and the purchase costs could be prohibitive. While the Commission has compulsory purchase powers, these have not been applied in practice, with land instead being purchased from willing owners (Edlin, 1969). However, this approach may not remain feasible due to the large proportion of suitable land that may need to be planted (Section 3.3), leaving little room for flexibility if used as the only mechanism to purchase land from willing owners. Attempts to use compulsory purchase in a significant way are likely to be unpopular with landowners and politically charged (NFU Scotland, 2017). In Spain, a 'consortia' approach was used during the mid-20th century where the state carried out afforestation and management of the area, but did not take control of the land, although this was typically done on publicly rather than privately owned land (Vadell et al., 2016).

4.3.6.2 Private forestry

Efforts to encourage private planting in the UK were initially slow to take-off, with landowners having neither the money nor interest to carry out new planting (Aldhous, 1997; Forestry Commission, n.d.). However, by the 1940s the first of a series of fiscal schemes and grants was introduced to encourage private planting (Forestry Commission, 1956).

The 1980s saw controversy with large scale private planting in The Flow Country of Scotland, driven by generous grants and a tax system that allowed for significant returns on investment, at the expense of profound environmental damage and habitat destruction (Warren, 2000; Stroud et al., 2015). This is a striking example of the dangers of poorly planned woodland policy, but, perhaps also, of the speed at which afforestation can take place given appropriately generous financial incentives. The lessons learned during this time resulted in a ‘greening’ of forest practice, with more vigorous assessment of grant applications and a shift to multi-purpose forestry (Warren, 2000). Today, various grant schemes are still used to encourage woodland expansion by private owners and make this financially viable (Hardaker, 2018), with the private sector being responsible for the vast majority of recent new planting in the UK (Hopkins et al., 2017).

4.3.6.3 Encouraging private planting

While grant payments have been instrumental in encouraging woodland expansion in the UK (Thomas et al., 2015), recent surveys have found low uptake, or planned uptake of forestry by farmers (Hopkins et al., 2017). This can be seen in the national planting rates, with an average of just 9,590 ha of new planting each year in the UK between 2010 and 2019 (Figure 4.6). Numerous reasons for this lack of uptake have been reported in the UK and elsewhere (Lawrence and Dandy, 2014) and include: an application process perceived as highly complex and requiring external assistance (Thomas et al., 2015); delays in receiving income (Watkins, 1996); lack of financial incentives (Duesberg et al., 2014); a loss of both productive land (Watkins, 1996; Howley et al., 2015) and ability to demonstrate farming skill (Burton, 2004); a preference for food production, land-use flexibility, and the farming lifestyle (Duesberg et al., 2014), and tradition (Duesberg et al., 2014). There are also currently few financial incentives to increase uptake of agroforestry, and indeed in Scotland agricultural subsidies may be lost depending on planting densities (Saunders et al., 2013). Barriers such as these will need to be addressed if rates of private planting are to increase to meet proposed goals.

Table 4.7: Woodland area in the United Kingdom. Adapted from Aldhous (1997), with the inclusion of figures from Forestry Commission (2014) and Forestry Commission (2019a).

	Effective date of survey or census									
	1895	1913	1924	1939	1947	1965	1980	1996	2014	2019
	Woodland area (thousands of ha)									
Private										
Great Britain	1076	1267	1204	1197	1205	1085	1216	1554	2268	2325
England					673	651	692	758	1087	1093
Scotland					436	351	422	669	942	988
Wales					96	83	102	127	189	192
Northern Ireland			16	16	15			19	50	51
State										
Great Britain	27	27	50	179	252	655	892	852	809	801
England						234	255	223	215	215
Scotland						304	498	508	477	469
Wales						117	139	121	117	117
Northern Ireland			2	8	9			75	62	62
Total Woodland (UK)			1272	1400	1481			2500	3139	3187

4.3.7 Future work and considerations

Large scale afforestation for carbon sequestration is a complex undertaking, and this work largely concerns just one aspect of this – the availability of land. While we explore the feasability of meeting proposed planting targets, we do not assess the validity of the targets themselves. In creating these, assumptions are made concerning both the level of carbon sequestration required, and the area of afforestation required to achieve this. The former will depend on factors such as future levels of emissions, and the uptake and success of other mitigation measures such as direct air carbon capture, both of which are highly uncertain (IPCC, 2018). The latter will depend on factors such as the impact of CO_2 fertilization (Jiang et al., 2020), the species planted (Wang et al., 2017; Kirby and Potvin, 2007), and management regime used (Noormets et al., 2014). Therefore, while results presented here demonstrate an ability to meet proposed planting targets, whether these in turn will meet climate change mitigation targets is beyond the scope of this work.

In addition to rates of carbon sequestration, the kinds of trees, how they are grown, and where they are grown all determine the magnitude of additional ecosystem services provided by woodland, and who benefits from these (Chazdon and Brancalion, 2019). Future work will need to model the spatial variability of ecosystem service delivery from woodland to ensure maximum benefits are derived from large scale afforestation. In essence, while the present study explores where planting *could* take place in the UK, the next step will be to identify where it *should* take place and how this can be achieved.

4.4 Conclusion

There are a variety of proposed targets for woodland creation in the UK, ranging from hundreds of thousands to millions of hectares within the next 10 to 80 years. Numerous constraints dictate where these woodlands could not, or potentially should not, be established. Of these, those that can be described as 'policy constraints' – protected areas and good quality agricultural land, occupy the greatest area in the UK.

Sufficient space is available to meet the highest proposed woodland creation goals, even if planting is prevented on moderate to good quality agricultural land and within protected areas. However, this would require planting on a large proportion of available land, which is unevenly distributed across the UK. This would leave little room for flexibility to allow for woodland creation in the most optimal locations, both to optimise the provision of additional ecosystem services, such as air pollution control and recreation, and prevent negative impacts upon local communities and landscapes. This lack of flexibility could also complicate the practicalities of either acquiring land from existing owners or encouraging them to plant upon it.

While this initial analysis suggests that meeting national planting targets is possible, the scale of change being proposed and impact it could have on British landscapes is significant. Meeting many of these proposed targets would result in a transformational change in British land cover, which could result in tens of thousands of farms being converted to forestry, and the replacement of hundreds of thousands to millions of hectares of grassland, arable and horticultural land and other land covers. These more ambitious targets would also require rates of planting that far exceed those seen historically, while planting rates in recent years have been comparatively low. Expansion of British woodland in the early to mid-20th century was driven primarily by an increase in state forest area; however, the conditions that enabled this do not necessarily apply today. More recently, grant schemes have been used to encourage planting by private landowners, although participation has been low, with a variety of reasons for this being identified. These barriers will need to be addressed if targets for planting, and therefore carbon sequestration and storage, are to be met.

4.5 Acknowledgements

This project was supported by the Centre for Global Eco-Innovation, part funded by the European Regional Development Fund, in collaboration with LUC (grant reference 19r16p01012).

5. Spatial distribution of national-scale afforestation targets for new tree planting: a multiple ecosystem service approach

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Abstract

Large-scale afforestation has the potential to be amongst our most important natural climate solutions, and recent years have seen a range of national and international targets proposed for new tree planting. These planting targets, which primarily aim to deliver carbon sequestration and storage, also have the potential to deliver additional ecosystem services if targeted appropriately. In this paper, we explore how new woodland creation could be sited to optimise for the provision of selected ecosystem services, using a recently proposed national afforestation target in England as a case study. Using spatial data and predictive ecosystem service modelling, we quantify the potential provision of three example ecosystem services from new woodland creation at a national scale: carbon sequestration, recreation, and flood mitigation. While rates of carbon sequestration are largely high throughout the study area, provision of flood mitigation and recreation are more localised, with the lack of congruence between services meaning all three cannot be fully optimised for simultaneously. Results show that spatially targeting woodland creation at the national scale results in the highest level of ecosystem service benefits, but risks overwhelming landscapes with new planting. Spatial targeting within smaller spatial units, such as political and administrative sub-divisions, results in more evenly distributed planting, but lower ecosystem service benefits, largely through decreases in flood mitigation potential. All

scenarios however have the potential to deliver far greater benefits compared with randomised, untargeted planting.

5.1 Introduction

It is becoming increasingly clear that rapid, large-scale action is required to reduce risks related to global climate change (Rockström et al., 2017). Afforestation and reforestation has the potential to be amongst our most effective natural climate solutions (Bastin et al., 2019; Griscom et al., 2017; Lewis et al., 2019), and recent years have seen numerous initiatives aimed at encouraging tree planting at both the international (Kula, 2010; United Nations, 2015; IUCN, 2020) and national (Andrasevits et al., 2005; Palaghianu, 2015; Department of Communications Climate Action and Environment, 2019) scales. Strategies for creating new woodland will be required to meet these goals, initially addressing issues such as availability of space, trees and financial capital (Whittet et al., 2016; Burke et al., 2021), implementation of planting, and longer-term issues such as ongoing maintenance and monitoring.

Spatially targeted planting could ensure that preferred environmental and societal benefits are gained from these large-scale afforestation efforts. In addition to carbon sequestration through tree growth, and storage in tree biomass, litter and soil (Dewar, 1990; Cannell and Milne, 1995), forests can provide a range of ecosystem services (Lake et al., 2020). Two examples of these are recreation and flood mitigation. There is increasing recognition of the positive impact woodland recreation can have on wellbeing, with trees being linked to improvements in both physical and psychological health (Bell and Ward Thompson, 2014; Goodenough and Waite, 2019). Mature forested catchments have also been shown to provide higher evaporative losses and reduce peak flows associated with smaller storms compared with grassland (Dadson et al., 2017). With studies highlighting the contribution of greenhouse gas emissions to flooding (Pall et al., 2011) and extreme precipitation events (Min et al., 2011; Dadson et al., 2017; Christidis et al., 2021), this subject of flood mitigation is of increasing importance. The degree to which these ecosystem services are provided depends on the location, extent, configuration and condition of the woodland from which they originate. The spatial distribution of ecosystem service provision is therefore heterogeneous, and woodland in one location will not necessarily provide the same benefits as another, even if it has similar characteristics (Gimona and Van Der Horst, 2007).

Recognising this spatial variability, recent years have seen the use of various methods to synthesise the results of multiple ecosystem service assessments (Cortinovis et al., 2021), and identify ‘hotspots’ where provision is highest, primarily for the purposes of conservation (Cimon-Morin et al., 2013). These can be defined both as key areas that provide more than one ecosystem service, or a large proportion of a single particular service (Egoh et al., 2008; Cimon-Morin et al., 2013). A range of methods have been used to identify these hotspots. These include a richest cells approach where locations with the highest value are chosen, often using expert opinion or quantiles to define a cut off, measures of spatial clustering such as the Gi* statistic, counting the number of services in an area to give a measure of diversity, or the use of heuristic optimisation algorithms (Schröter and Remme, 2016; Cortinovis et al., 2021).

Past efforts to identify ecosystem service hotspots using these techniques have often been descriptive, mapping the current provision of services in order to find synergies between them (Egoh et al., 2008; Hou et al., 2018; Raudsepp-Hearne et al., 2010; Roces-Díaz et al., 2018; Wu et al., 2013). These can be contrasted with predictive studies, which explore how this provision could change under differing land-use scenarios (Crossman and Bryan, 2009; Davis et al., 2019; Finch et al., 2021; Gimona and Van Der Horst, 2007). While these have largely been carried out at local and regional scales, predictive modelling of ecosystem service hotspots at a national scale has the potential to be a useful tool in implementing emerging national scale afforestation targets.

Large-scale afforestation can be driven by a range of approaches, and at a range of spatial scales (Burke et al., 2021; Lawrence and Ambrose-oji, 2015). For example, a centralised, 'top-down' approach was seen in the United Kingdom during the first half of the 20th century, driven by the purchase and afforestation of vast areas of cheap, marginal quality land by the state (Nail, 2008). These efforts were instigated due to a need for timber following shortages in the wake of the First World War (Richards, 2003), with other ecosystem services, or indeed disservices these swathes of new woodland could generate, being of secondary concern. Planting may also be carried out by private landowners, although often still with the encouragement of government grants or a beneficial tax regime. Recent years have seen a shift to private woodland ownership and planting in a number of countries including Spain (Vadell et al., 2016), Romania (Palaghianu and Dutca, 2017), and the United Kingdom (Hopkins et al., 2017). In the latter case, while efforts to encourage private planting were initially slow to take-off (Aldhous, 1997), the introduction of a generous grants and tax system in the 1980s saw large scale private planting in ecologically sensitive areas, leading to profound environmental damage and habitat destruction (Warren, 2000; Stroud et al., 2015). The lessons learned during this time resulted in a 'greening' of forest practice in the UK, with more vigorous assessment of grant applications and a shift to multi-purpose forestry (Warren, 2000), and there is now increasing recognition of the range of benefits provided by woodlands in forest policy (Forestry Commission, 2017).

Spatially targeting woodland creation, to optimize the provision of ecosystem services, has the potential to ensure that benefits are maximized from the creation of new woodland, such as that being proposed for carbon sequestration purposes. While much existing work has focused on mapping the current provision of services, the development of predictive ecosystem service models, datasets and techniques means that levels of provision from hypothetical *new* woodland can be determined, and the optimal location for the desired services chosen.

In this paper, we explore how new woodland creation could be sited to optimise for the provision of selected ecosystem services, using a recently proposed national afforestation target in England as a case study. Using predictive ecosystem service models and spatial data, we model the potential provision of three key ecosystem services, used as examples to demonstrate the approach: carbon sequestration, recreation, and flood mitigation. We quantify the level of spatial correlation between these three ecosystem services, and identify locations where woodland creation could deliver multiple benefits.

Drawing on the range of different strategies for large scale afforestation experienced historically in the United Kingdom and Europe, we explore the impact of planning within multiple spatial units at a range of scales from the national scale, analogous to a centrally administered planting scheme, to local scale, analogous to a scheme administered by local government authorities. We measure the impact that the boundaries within planting decisions are made has on both the level of ecosystem service provision, and the resulting patterns of afforestation. The results of these scenarios are then compared with a random planting scenario, where the creation of new woodland is untargeted. Finally, we discuss the implications of new tree planting, including potential impacts on local communities, and how the approach could be extended for use within differing planning systems and to achieve differing woodland creation priorities.

5.2 Materials and methods

5.2.1 Case study and study area

To demonstrate the use of an ecosystem services-based approach to large-scale afforestation planning, England was used as a case study (Figure 5.1). As part of the United Kingdom, England has seen substantial planting in recent history (Aldhous, 1997), but remains amongst the least wooded countries in Europe (FAO, 2020). Recent years have seen a range of proposals aimed at increasing woodland area in the UK, primarily for the purposes of carbon sequestration and storage (Burke et al., 2021). These proposals, which involve the creation of 9,200 ha (Thomson et al., 2018) to 176,000 ha (Centre for Alternative Technology, 2013) of new woodland each year, provide an opportunity to explore ecosystem services-based planting in a pragmatic manner.

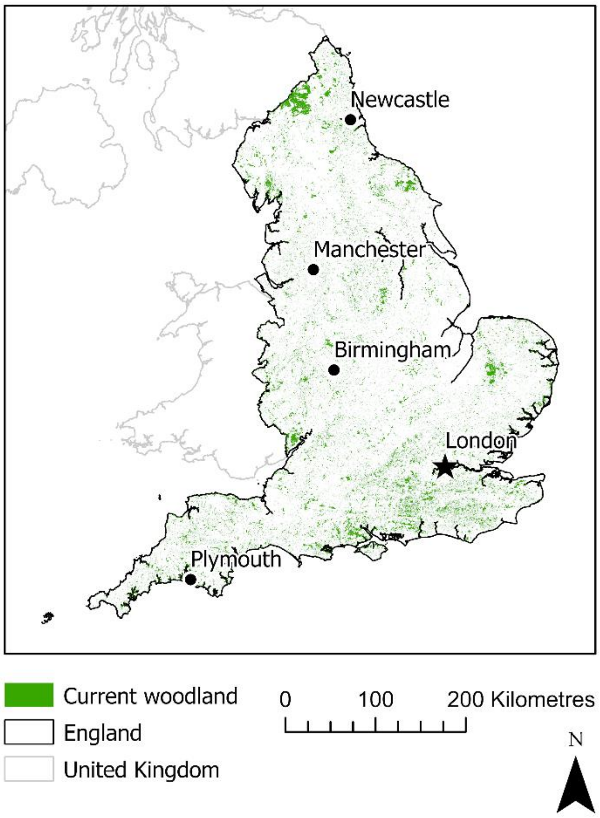

Figure 5.1: Map of the study area, England, showing current woodland area (Forestry Commission, 2018). The wider United Kingdom and selected major cities (© OpenStreetMap, openstreetmap.org/copyright) are included for context.

In this case study, we use the example of the *medium ambition* scenario proposed by the UK Climate Change Committee (Committee on Climate Change, 2018; Thomson et al., 2018), an independent statutory body formed to advise the UK government on climate change mitigation and greenhouse gas emission reduction. This medium ambition scenario outlines a suite of measures intended to reduce greenhouse gas emissions from the Land Use, Land Use Change and Forestry sectors, assuming reasonable uptake of currently available technologies (Thomson et al., 2018). Amongst these is a proposal to plant 10,000 ha of new woodland in England each year until 2100, resulting in the creation of approximately 840,000 ha of new woodland, of which 30% is assumed to be conifer species (represented by Sitka spruce), and 70% broadleaved species (represented by Beech). While primarily intended to facilitate carbon sequestration and storage, this large-scale new tree planting offers an opportunity to provide a range of additional ecosystem services, if planned for appropriately.

5.2.2 Modelling of ecosystem services

The potential provision of ecosystem services from new woodland creation in England was modelled in a spatially explicit manner. To demonstrate the use of an ecosystem services based approach to afforestation planning, three services were modelled as examples. As a reduction in anthropogenic greenhouse gas emissions is the primary aim of many proposed tree planting schemes in the United Kingdom, carbon sequestration was selected as one of these. In addition to this, recreation, and flood mitigation were also included. These services were selected as they are of increasing importance globally, and in the United Kingdom specifically, and woodland is known to be an important asset in their provision (Section 5). While three services were modelled here, future work could involve the use of additional or different services, should appropriate data be available (Section 5.4).

For each of the services, modelling was carried out for the whole of England at a spatial resolution of 1 km^2. As in comparable studies (Davis et al., 2019; Finch et al., 2021; Hou et al., 2018), we used a combination of existing predictive ecosystem service models and custom measures derived from spatial data, to create layers showing potential ecosystem service provision from new woodland.

5.2.2.1 Carbon sequestration

A spatial data set quantifying the potential for carbon sequestration was calculated using existing models and data, with a methodology comparable with that previously used to estimate timber production and carbon sequestration in the UK (Finch et al., 2021; Haw, 2017). This is a two-stage method, where yield classes were estimated first and where then converted to carbon sequestration rates. The Forest Research Ecological Site Classification (ESC) model (Bathgate, 2011) was first used to predict the yield class of woodland established within each grid square of the study area. Yield class is an index used to describe forest productivity, based on the maximum mean annual increment of cumulative timber volume achieved by a tree species under specified conditions, measured in cubic metres per hectare per year (Mathews et al., 2016). The model uses climate characteristics (accumulated temperature, continentality, aspect and moisture deficit) and soil characteristics (moisture regime and nutrient regime) to assess the suitability of a site for the growth of a given tree species, including a prediction of its maximum potential yield class (Bathgate, 2011). The model was run for each grid cell in the study area, for both Sitka Spruce (the representative coniferous species) and Beech (the representative broadleaf species). The result was two layers identifying the maximum potential yield class of both species across England (Figure 5.2).

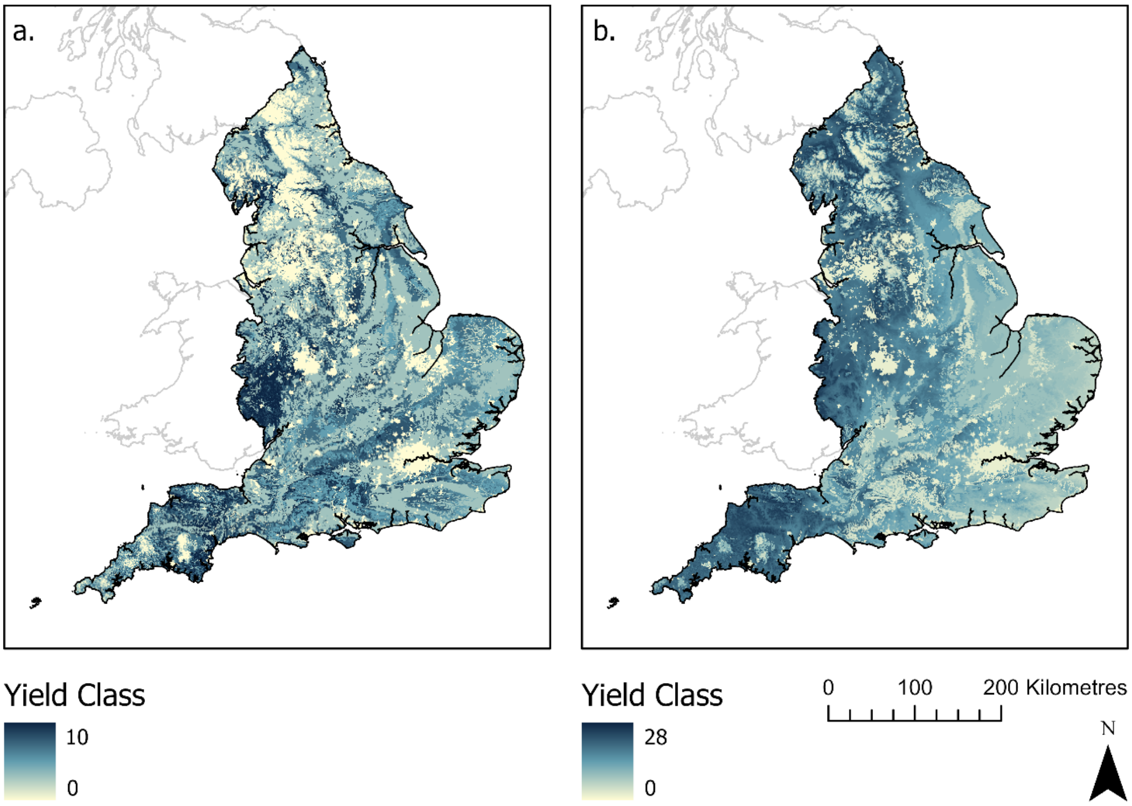

Figure 5.2: Predicted maximum potential yield class of a.) Beech and b.) Sitka Spruce within England, obtained from the Forest Research Ecological Site Classification model (Bathgate, 2011).

Lookup tables were used to convert yield class to rates of carbon sequestration (Randle and Jenkins, 2011). The lookup tables provide rates of carbon sequestration per hectare for a given tree species of a given yield class, under a specific management regime. As in the UK Climate Change Committee medium ambition scenario, we assume planting with a spacing of 1.2 m for Beech and 2 m for Sitka Spruce (Thomson et al., 2018). As rates of carbon sequestration vary with a tree's age, we calculated the average carbon sequestration over the first 100 years of growth, and assume that no thinning has occurred. Carbon sequestration values for a range of even numbered yield classes are given in the lookup tables used. Yield classes outside this range, and odd numbered yield classes, were interpolated using a simple straight line fit (Figure A1, Figure A2). Annual rates of carbon sequestration were then calculated for each grid cell, assuming each cell was planted with 70 ha of Beech, and 30 ha of Sitka Spruce.

5.2.2.2 Recreation

Potential for recreation was calculated using the ORVal model (Day and Smith, 2016). This is a statistical recreational demand model that allows visitation and monetary welfare values to be generated for existing and hypothetical new greenspaces at specified sites (Day and Smith, 2017). It has previously been used in a variety of studies to estimate the recreational value of existing sites (Clark, 2017; Day et al., 2018; Petersen, 2018; Petersen, 2021), and to predict future provision under potential land use scenarios (Davis et al., 2019; Finch et al., 2021).

The model was used to estimate the monetary welfare value of a new woodland site created at the centre of each grid cell in the study area. Each site was defined as 1 km^2 in area, and composed of 70% broadleaf woodland and 30% coniferous woodland.

5.2.2.3 Flood mitigation

Calculation of flood mitigation potential was based largely on UK Environment Agency data that identifies potential areas for Working with Natural Processes (WWNP), also known as Natural Flood Management (Hankin et al., 2017). This dataset identifies areas where tree planting has the potential to mitigate flood risk, including flood plains, riparian zones, and on slowly permeable soils (Figure 5.3), with trees slowing overland flow, enhancing canopy evaporation, increasing floodwater storage, and dissipating flood energy.

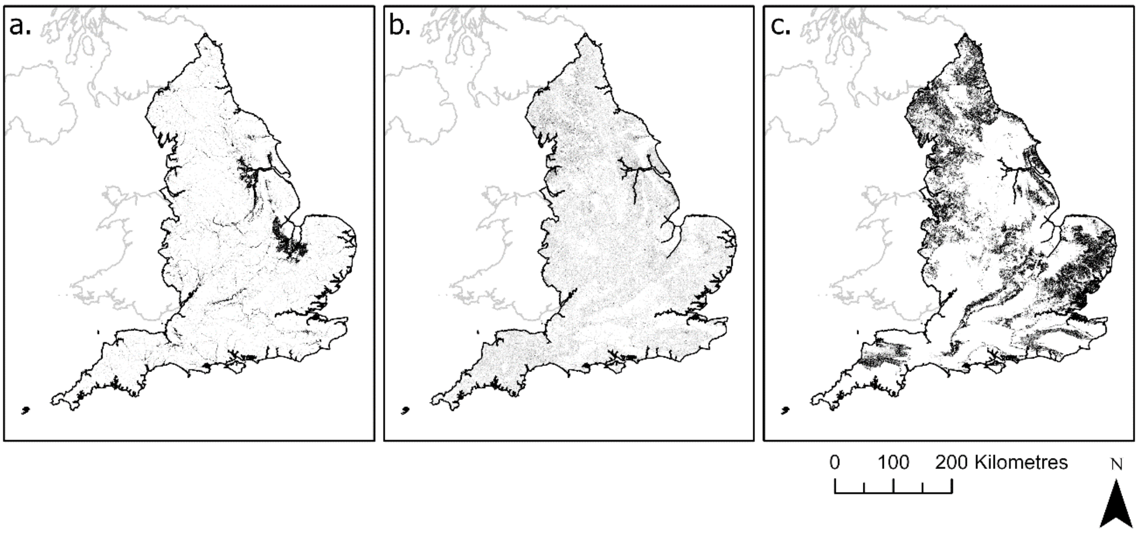

Figure 5.3: Areas in England with the potential to manage flood risk by tree planting on: a.) floodplains, b.) riparian areas, c.) slowly permeable soils (Hankin et al., 2017).

The Environment Agency WWNP dataset is categorical, with two classes: areas that could provide a flood mitigation benefit if planted, and areas that could not. The dataset was extended by calculating distance to river mouth for sections of river in locations where planting could provide a benefit. Modelling suggests that forest restoration in distal headwaters, far from the river mouth, is effective at reducing peak flood discharged, compared with restoration near the catchment outflow, which can increase peak magnitude (Dixon et al., 2016).

Rivers were first identified using Ordnance Survey Open Rivers (Ordnance Survey, 2019), a vector polyline dataset that identifies the alignment of water courses in the UK. Artificial canals were first removed from the dataset under the assumption that these are unlikely to flood. In the dataset, water courses are split into separate sections at each confluence (river junction). For each of these river sections, its distance to the mouth of that river was calculated (Figure 5.4a). This was then interpolated using inverse distance weighting (IDW) to create a continuous raster surface showing an average distance to mouth for nearby river channels (Figure 5.4b). This was clipped using the WWNP dataset to only include areas where planting has the potential to provide a flood mitigation benefit (Figure 5.4c). The result is a layer identifying where planting could aid flood management, and to what extent this could potentially reduce flood peaks. Finally, the layer was standardised such that all values lay between 0 and 100 (Section 5.2.4), and an average value calculated for each 1 km^2 grid cell.

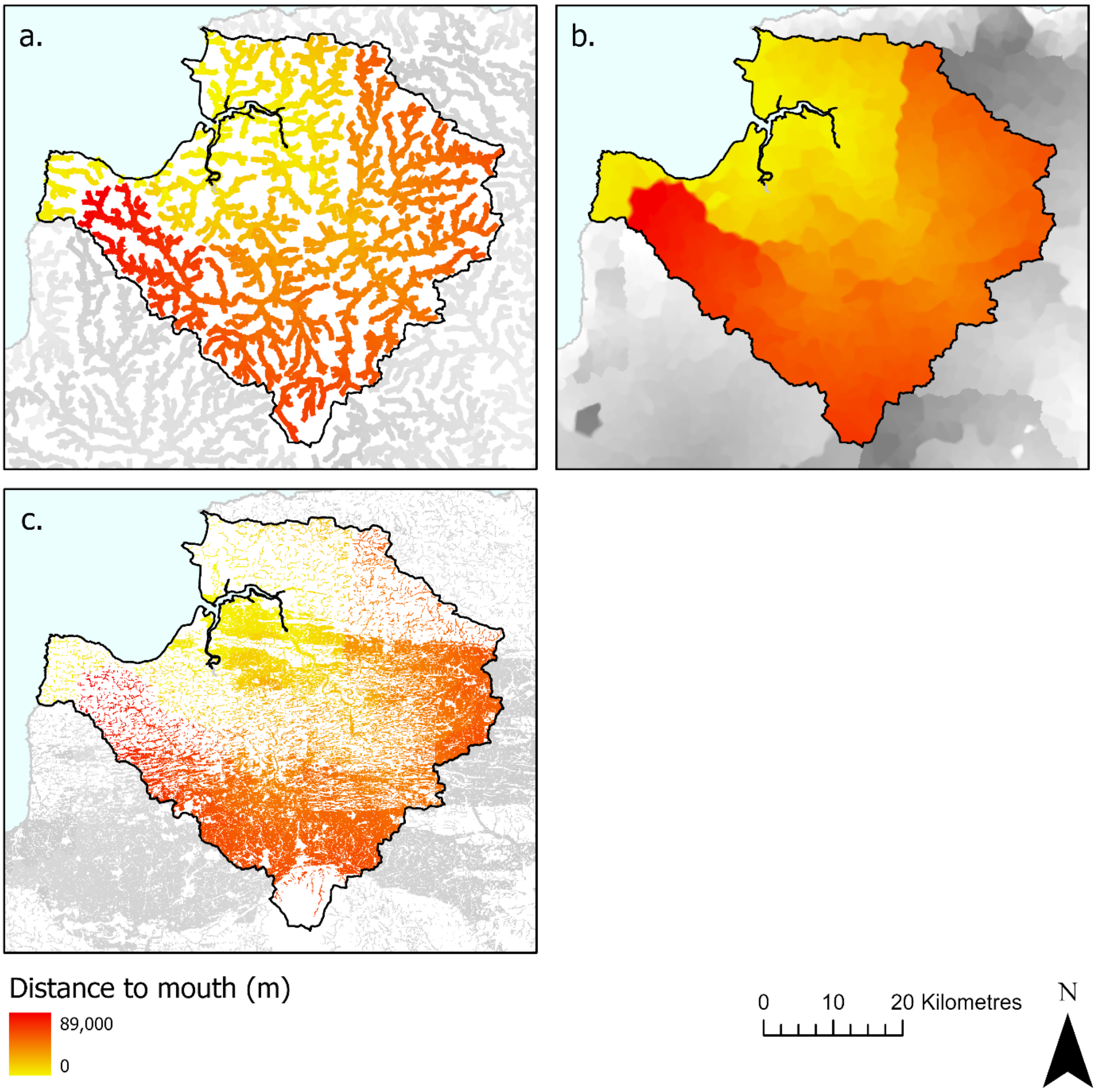

Figure 5.4: Creation of the flood mitigation layer, focused on a single example catchment for clarity. a.) Distance to mouth for sections of river channel. b.) Interpolated continuous raster. c.) Results clipped to only cover the WWNP dataset.

5.2.3 Removal of unsuitable areas

Tree planting is not possible, or suitable, in all areas. Physical and environmental constraints on afforestation were identified using constraint maps produced in Burke et al. (2021). Physical constraints include areas where large-scale tree planting is not physically feasible such as existing woodland, water, rock and coastal sediment, above the climatic treeline, and urban and suburban areas. Environmental constraints identify areas of peat and bog. Afforestation in these areas can lead to net CO_2 emissions (Sloan et al., 2018), and as a result the UK Forestry Standard prohibits planting on peat exceeding 50 cm in depth, and on sites that would compromise the hydrology of bog or wetland habitats (Forestry Commission, 2017). Cells where over half the land was covered by these physical and environmental constraints were designated as being unavailable for planting, and were removed from the analysis.

5.2.4 Quantifying the provision of multiple ecosystem services

Areas where afforestation could deliver multiple ecosystem services were identified using an 'intensity' approach, widely used in studies on multiple ecosystem services (Cortinovis et al., 2021; Schröter and Remme, 2016). Each ecosystem service provision layer was first normalised such that its values lay between 0 and 100:

$$x_{norm} = \frac{x - \min(x)}{\max(x) - \min(x)} \times 100$$

For each cell, the mean level of ecosystem service provision from the three services modelled was then calculated, resulting in a combined ecosystem services value. In this instance, all three services were assigned equal weights, although future work could assign weights to prioritise specific ecosystem services (Section 5.4.6). Following a richest cells approach (Schröter and Remme, 2016), these combined ecosystem service values were then ranked from high to low, and the top ranked cells selected for planting.

5.2.5 Construction of planting scenarios

Five scenarios were constructed, with planting locations being identified within different spatial units in order to explore the use of an ecosystem services-based approach to afforestation planning under differing scales of decision-making. The first, the *national* planting scenario, is analogous to a large-scale centrally administered planting scheme. It purely maximises the provision of multiple ecosystem services from new woodland at the national scale by selecting the 8,400 1 km^2 cells containing the highest combined ecosystem services values from across the whole of England, identifying 840,000 ha of land for afforestation as proposed in the *medium ambition* planting target.

Subsequent scenarios were constructed using administrative subdivisions of regions, counties, districts and parishes as spatial units (Table 5.1). In these scenarios, it is assumed that each spatial unit plants a proportion of the national target relative to its land area, analogous to a planting scheme devolved to local authorities and communities. For example, in the *districts* scenario, The City of Lancaster district has a land area of 57,621 ha, making up approximately 0.44% of the land area of England. In this scenario, it would therefore plant 0.44% of the national 840,000 ha planting target, equal to 3,710 ha. Following the approach used in the *national* planting scenario, cells within the City of Lancaster District were first ranked by their combined ecosystem services value from high to low, and the top 37 selected, identifying the 3,700 ha of land required for afforestation. This process was repeated for each district in England, resulting in approximately 840,000 ha of land being identified for afforestation.

Table 5.1: Spatial units used in construction of the planting scenarios.

		Land Area (ha)		
Spatial Unit	**Count**	**Average**	**Largest**	**Smallest**
National	1	13,046,148	13,046,148	13,046,148
Regions	9	1,449,563	2,385,107	157,351
Counties	48	271,793	865,697	290
Districts	314	41,548	502,617	290
Parishes	10,739	1,215	25,556	0.038

The current distribution of woodland in England varies substantially. These region to parish scale scenarios therefore evaluate a method where planting is more evenly distributed across the country, and ensures that no single area is overwhelmed by new woodland. It also allows the impact of planning at different levels to be explored, from national government, to local parishes. These scenarios were compared with a *random* planting scenario, where grid squares were selected from across England at random for planting, to simulate untargeted woodland creation.

5.3 Results

5.3.1 Spatial variability of ecosystem service provision

Each of the three ecosystem services modelled were found to have substantial spatial variability and varying levels of provision (Figure 5.5). Potential rates of carbon sequestration were found to range from 184 $tCO_2e/km^2/a$ in the least productive areas, to 1,429 $tCO_2e/km^2/a$ in the most. Potential welfare values from recreation were more variable, ranging from £3,185/km^2/ a in the least valued areas, to £1,685,796 in the most. As potential for flood mitigation was calculated as a normalised index, values for this ranged from 0 where planting is deemed to have no positive impact on flood mitigation, to 1 where it is deemed to have the most.

Rates of carbon sequestration were found to be highest in the west of the country, due to more favourable climatic conditions resulting in increased tree growth, whereas welfare values for new woodland recreation sites were highest near urban areas, due to the presence of populations that can utilise these sites. Land with potential for flood mitigation from afforestation was found to be relatively evenly distributed across the study area, with the highest values largely being found in inland areas where the distance from river mouth, and therefore potential benefit, is greatest. No spatial correlation between the three services was found using Pearson correlation coefficients (Table 5.2).

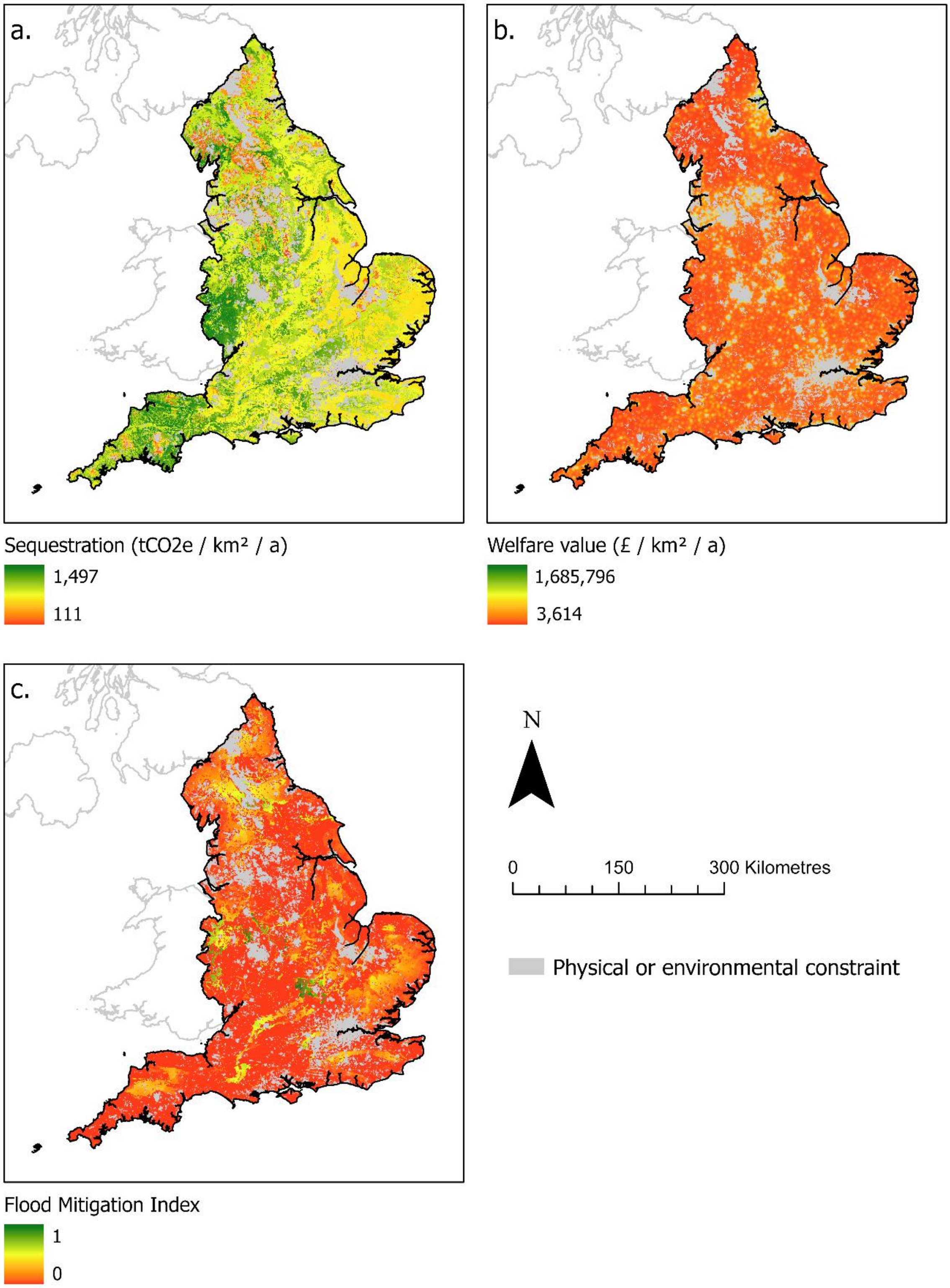

Figure 5.5: Distribution of ecosystem service provision from potential new woodland. a.) Carbon sequestration, b.) recreation, c.) flood mitigation.

Table 5.2: Pearson correlation coefficients for the three ecosystem services modelled. $P < 0.01$ for all coefficients.

	Carbon Sequestration	**Recreation**	**Flood Mitigation**
Carbon Sequestration		-0.056	-0.041
Recreation			-0.098
Flood Mitigation			

Normalising each of the three ecosystem service maps to lie between 0 and 100 and combining them with an equal weighting results in a map of combined ecosystem services value (Figure 5.6). This was found to vary from 0.34 to 69, demonstrating that while afforestation in all unconstrained locations within the study area would provide ecosystem services, even if to a very small degree, no singular location is optimal for all three ecosystem services studied here (as this would result in a combined value of 100).

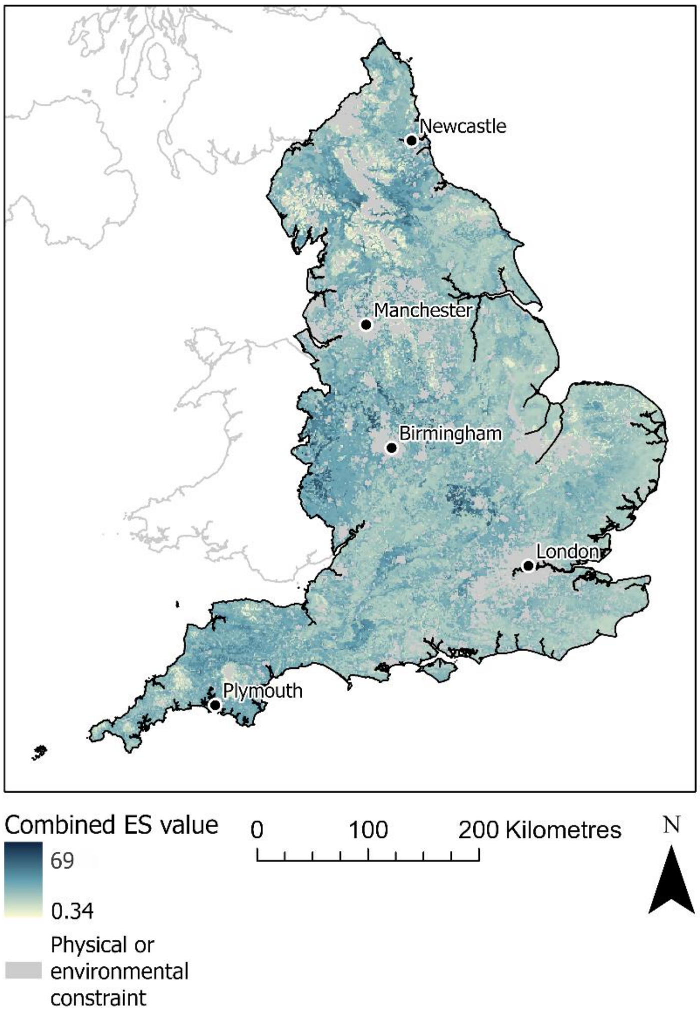

Figure 5.6: Distribution of combined ecosystem service value from potential new woodland.

5.3.2 Planting scenarios

From this map of combined ecosystem services value, five hypothetical planting scenarios were explored, using different levels of spatial planning unit (Figure 5.7). Each planning-level leads to a differing proposed planting distribution. The *national* scenario, where the national tree planting target is distributed at a national-level, results in larger concentrated areas of afforestation, especially to the north, west-midlands and south. Whereas in the *regions* and *counties* scenarios, where the national target is distributed proportionally across each region and county, brings more planting to the east of England. The *districts* scenario results in more distributed planting, with the larger continuous areas of new afforestation, seen in the national-level modelling, largely gone. Meanwhile, the *parishes* and *random* scenarios both display mostly singular cells of new planting, distributed throughout the study area.

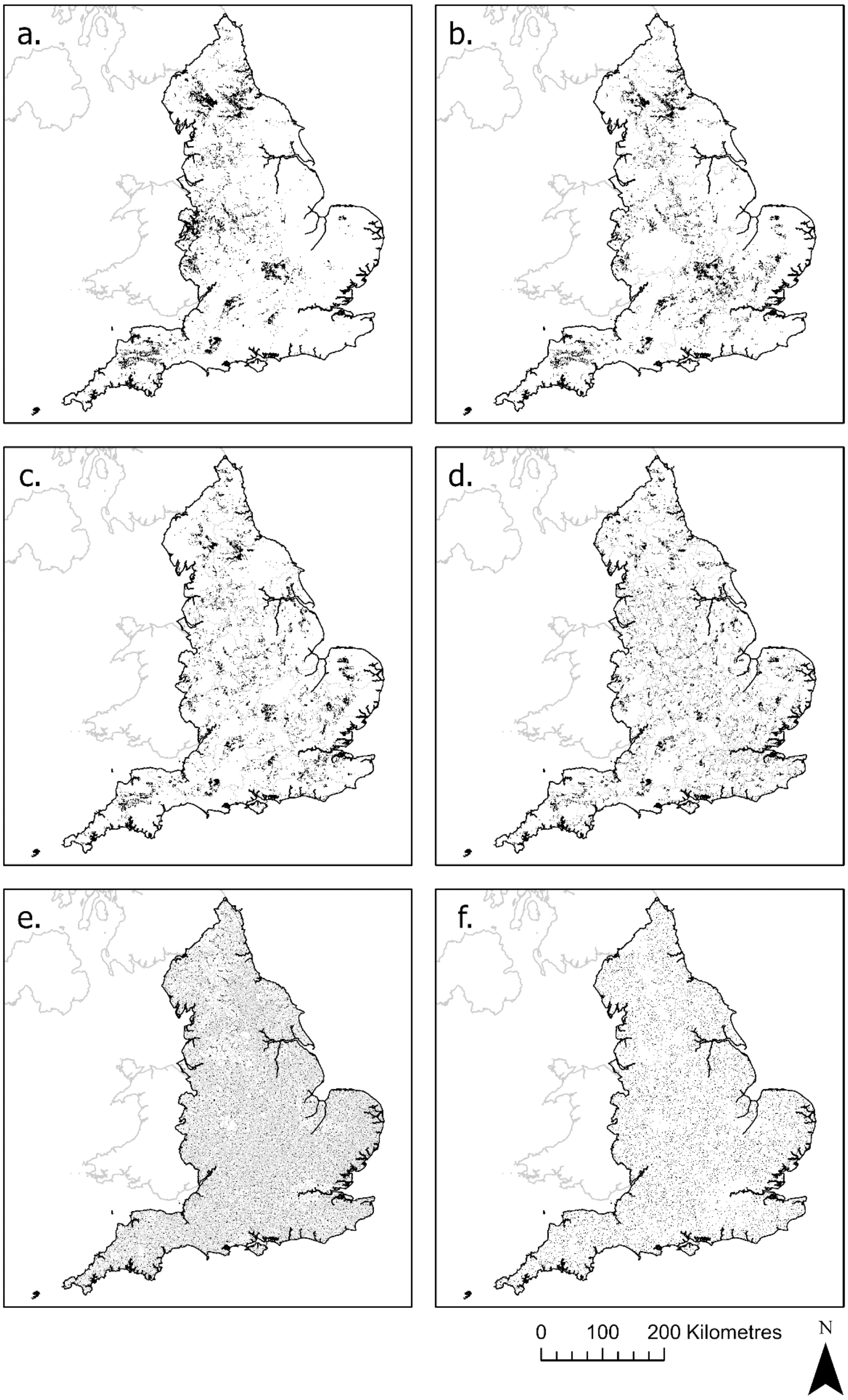

Figure 5.7: Proposed patterns of afforestation from modelled scenarios. a.) national, b.) regions, c.) counties, d.) regions, e.) parishes, f.) random. Total proposed new planting in all scenarios is approximately 840,000 ha, the Committee on Climate Change medium ambition scenario (Section 5.2.1).

These differing patterns of planting result in differences in potential overall levels of combined ecosystem service provision from proposed new woodland creation. Planting according to the *national* scenario would result in the highest overall level of provision, with a mean combined ecosystem services value of 40.5. This mean value was then found to decrease as the spatial unit within which locations for planting are identified decreases in size (Table 5.3, Table 5.4). All planned scenarios are higher than untargeted woodland creation in the *random* scenario, with a mean value of 24.2.

The larger spatial units result in higher values for carbon sequestration and flood mitigation. Conversely, smaller spatial units, with the exception of the *parish* planting scenario, while having lower values for overall intensity, result in a slightly higher value for recreation, possibly due to increased planting in more urbanised areas. Values for carbon sequestration are comparatively high in all scenarios, suggesting spatial targeting for this service is less important compared with recreation and flood mitigation. It was also found that the number of cells selected in the *parish* scenario (7,820) is somewhat less than the goal of 8,400 (equal to 840,000 ha of afforestation). This is because the amount of planting assigned to some parishes is less than 1 km^2, so no cells were selected, while others did not contain enough unconstrained land to make up the required number of cells.

Table 5.3: Ecosystem service provision in each of the modelled scenarios (normalised values).

		Combined ES Value			Carbon Sequestration			Recreation			Flood Mitigation		
Spatial Unit	**Cells Selected**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**
National	8,400	40.5	34.8	68.9	72.2	1.2	100.0	12.2	0.19	100.0	37.2	0.0	100.0
Region	8,399	39.3	24.2	68.9	71.3	1.2	100.0	13.0	0.2	100.0	33.5	0.0	100.0
County	8,402	37.7	18.0	68.9	71.2	1.2	100.0	13.8	0.2	100.0	28.2	0.0	100.0
District	8,370	36.4	6.0	68.9	70.5	1.2	100.0	14.7	0.2	100.0	23.9	0.0	98.3
Parish	7,820	31.6	1.2	68.9	68.8	1.2	100.0	13.2	0.08	100.0	12.8	0.0	98.3
Random	8,400	24.2	1.0	61.6	56.4	1.2	98.9	9.2	0.0	72.0	6.9	0.0	95.9

Table 5.4: Ecosystem service provision in each of the modelled scenarios (raw values).

		Carbon Sequestration ($tCO_2/km^2/a$)			Recreation (£/km^2/a)			Flood Mitigation (0 – 100 normalised index)		
Spatial Unit	**Cells Selected**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**
National	8,400	1,112.1	126.9	1,497.3	208,919	6,841.2	1,685,796.1	37.2	0.0	100.0
Region	8,399	1,099.7	126.9	1,497.3	222,967	7,436.8	1,685,796.1	33.5	0.0	100.0
County	8,402	1,098.3	126.9	1,497.3	235,127.9	6,601.1	1,685,796.1	28.2	0.0	100.0
District	8,370	1,088.9	126.9	1,497.3	250,604.4	6,601.1	1,685,796.1	23.9	0.0	98.3
Parish	7,820	1,065.0	126.9	1,497.3	225,316.9	4,951.1	1,685,796.1	12.8	0.0	98.3
Random	8,400	892.9	126.9	1,481.4	158,651.3	3,613.7	1,214,901.9	6.9	0.0	95.9

5.3.2.1 Distribution and size of afforestation

The amount of planting being considered in this case study is substantial. With a current woodland area in England of 1,308,000 ha (Forestry Commission, 2019a), the 840,000 ha of new planting proposed is equal to an increase of over 60%.

Of the 314 districts in England, 11 were identified as having no land available for planting. Of these, 10 were highly urbanised London boroughs where all cells were deemed unsuitable for planting when land with environmental and physical constraints was removed (Section 5.2.3). The other is the Isles of Scilly, where no data was available for the Ecological Site Classification model used in production of the carbon sequestration layer (Section 5.2.2.1).

Of the remaining districts with available space, all would receive new planting under the *districts* planting scenario. Under the *national* planting scenario however, 92 would receive no planting, while 10 would receive over 25% of their land areas as new woodland. Similarly, planning planting within larger spatial units, such as in the *national, counties* and *districts* scenarios, results in the creation of large, contiguous areas of new woodland, on the scale of, and in some cases exceeding, the largest existing woodland areas in England (Table 5.5). Previous studies have shown a preference for between 25% and 50% forest cover in a landscape (van der Horst, 2006), so concentrated planting at this scale has the potential to have a negative impact on local communities.

Table 5.5: The three largest contiguous woodland areas under each of the planning levels, compared with the largest existing woodland areas in England.

Woodland area (hectares)					
National	**Regions**	**Counties**	**Districts**	**Parishes**	**Existing**
654	570	284	127	12	465 (Kielder Forest)
613	379	217	125	11	208 (The New Forest)
554	256	158	100	9	137 (Thetford Forest)

5.4 Discussion

5.4.1 Impact of planning scale on ecosystem service provision and distribution

Much existing work has been carried out exploring the impact of scale and resolution on the perceived spatial pattern of ecosystem service provision (Burke et al., 2020; Hou et al., 2018; Roces-Díaz et al., 2018; Zen et al., 2019). This work highlights the related, but distinct, issue of spatial unit size on the optimisation of ecosystem service delivery. Given an areal afforestation target, our results indicate that delineating hotspots within larger spatial units, such as nationally in the case study presented here, provides greater benefits compared with planning within smaller spatial units, which offer a lesser opportunity to optimise multiple ecosystem service provision.

Using the *districts* spatial unit as an example, *Southwalk London Borough* district contains the lowest combined ecosystem services value (9.37) of the 301 districts in the study area with land available for afforestation, with a normalised value of 1.15 for carbon sequestration, 26.98 for recreation, and 0 for flood mitigation. That is, according to these results, planting in any other district with available land, has the potential to provide greater combined ecosystem service benefits. Planting a 1 km^2

grid cell in the *City of Stoke-on-Trent* district, for example, which has the highest maximum ecosystem service intensities within its boundaries (68.9), could provide over seven times the benefit, as this cell has high values for carbon sequestration, recreation, and flood mitigation (84.61, 29.44, and 92.73 respectively). More generally, of these 301 districts, 235 see more planting in the *districts* scenario than in the *national* scenario. That is, in the *districts* scenario, a proportion of planting in these 235 districts could provide greater benefits if instead moved elsewhere, as is the case in the *national* scenario.

5.4.2 Spatial implications for afforestation schemes

Resources for afforestation are finite, and there are both costs and practical issues associated with the establishment of new woodland (Whittet et al., 2016). Planning for afforestation at the national scale, or within large spatial units, may therefore be most desirable in order to obtain the greatest ecosystem service benefits from these efforts. Conversely, schemes that confine planting only to specific areas, such as selected districts, counties or regions (Cornwall Council, 2021; Lancaster City Council, 2019; Surrey County Council, 2021; Wirral Borough Council, 2020 etc.) may be a suboptimal use of resources.

Similarly, while England is used as a case study in this work, it may be that planting elsewhere within the wider United Kingdom, or even globally, could provide greater benefits. Lewis et al. (2019) for example suggests that the most efficient route to climate change mitigation through afforestation is the establishment and restoration of forests in tropical and subtropical regions, as trees in these areas are fast growing, have little impact on albedo, and could offer additional benefits, such as alleviating poverty, if well managed. It could therefore be argued that if purely concerned with maximising ecosystem service provision, and carbon sequestration in particular, planting in areas such as these should be prioritised.

5.4.3 Role of spatial targeting

While this work indicates that planning new planting within larger spatial units offers the greatest benefit when considering multiple ecosystem services, it also demonstrates that spatial targeting, even within smaller spatial units, provides significantly greater benefits compared with random, untargeted planting. This highlights the importance of considering multiple ecosystem services, even in local scale planting projects. While this smaller scale planting may result in lesser overall ecosystem service benefits, it may also better fit with local priorities and needs. Funding for afforestation is often driven by a complex mix of organisations with unique interests and spatial footprints. Local government for example will initiate afforestation projects not to maximise ecosystem service provision at a global scale, but for the benefit of local residents and constituents.

Our results support previous work mapping the provision of ecosystem services at a national or regional scale, finding that while hotspots do exist, levels of congruence between services modelled are generally poor (Egoh et al., 2008; Hou et al., 2018; Wu et al., 2013). With each of the three services modelled having unique distributions, each also has a different sensitivity to the spatial units within which planning takes place. Potential for carbon sequestration for example is high throughout much of the study area, and as such, it is high in each of the targeted scenarios tested (Table 5.3, Table 5.4). Areas with high potential for flood mitigation on the other hand are generally confined to certain inland locations. Potential flood mitigation benefits are therefore lower when afforestation is planned within smaller spatial units, and planting is distributed to spatial units in more coastal areas, suggesting that planning within larger areas is required to optimise for this service effectively.

5.4.4 Scale of ecosystem services

While afforestation often takes place within political or economic subdivisions, such as local government authorities or privately owned land, ecosystem services will cut across these boundaries. Carbon sequestration for example can be considered a "global" ecosystem service, with consumers receiving the same benefit regardless of their location relative to the woodland that provides it. Other services such as recreation are more localized, with benefits being provided only to those in proximity to the trees from which the flows of benefits originate (Cimon-Morin et al., 2013). These flows may also be directional, so trees planted for flood mitigation purposes may only benefit those downstream of them, for example, even if those upstream are in closer proximity.

5.4.5 Disbenefits of afforestation

Proximity to large scale afforestation may also have negative consequences. In our assessment, the *national* planting scenario results in larger continuous areas of planting, concentrated in certain areas, compared with the more distributed planting seen with smaller spatial units (Table 5.5). With studies having shown a preference for between 25% and 50% forest cover in a landscape (van der Horst, 2006), this concentrated planting has the potential to have a negative impact on local communities, although higher resolution work would be needed to measure this at the landscape scale.

5.4.6 Future work

Future work could explore how benefits from new planting can be shared amongst a population equally, which may involve weighting the distribution of afforestation by population, or more equitably, by assessing current baseline service provision and attempting to bring all areas up to a common level. Alternatively, it may be decided to minimise losses from replaced land cover, both in terms of the monetary cost of land conversion, or by quantifying the value of services lost when land cover in replaced, in essence calculating the net, rather than gross, value of new afforestation (Davis et al., 2019). While there is perhaps no singular correct answer to the "best" place for new afforestation, it is important that these issues are explored. In evaluating an existing national subsidy-based planting scheme in Scotland for example, Gimona and Van Der Horst (2007) found that existing approaches to spatial targeting of woodland creation were no better, or even worse, in terms of ecosystem service delivery than if the trees were planted randomly. The present study demonstrates the use of a combined ecosystem services approach for the translation of an aspatial woodland creation target into a spatially explicit and optimised plan for its execution. Here, we demonstrate its use to plan for the creation of woodland to prioritise for the provision of carbon sequestration, recreation, and flood mitigation ecosystem services, with each service being assigned an equal weight and importance. However, this approach could also be used for the optimisation of woodland creation according to differing priorities, and within different political and planning systems. It could, for example, account for the provision of other services, should suitable data be available. In addition to those modelled here, woodland is known to provide a range of benefits, from to air quality improvements to biodiversity (Lake et al., 2020), to the extent that it is unlikely that its true value to humanity could ever be fully appreciated.

Refinements could be made to improve the accuracy of modelled ecosystem service distributions. In this case study, planting in each grid cell is considered independently. That is, planting in one cell does not impact the ecosystem service benefit of planting in another, i.e., there is no spatial interaction. In reality, the presence of a woodland recreation site in one location, for example, is likely to reduce the value of a second in close proximity to it, as there is already an alternative site available, while the value of a single, large site formed by planting in two adjacent cells is likely to be different to the value of two smaller sites considered in isolation. Similarly, planting an area of woodland for the purposes of flood mitigation may reduce or remove the need for further planting in the same catchment. A more dynamic approach, with cells chosen for planting iteratively and accounted for in subsequent modelling of service provision, would likely result in different spatial patterns of ecosystem service supply, and therefore hotspot locations. A more dynamic approach could also be applied to other areas. In this work for example, a species split of 30% Sitka Spruce and 70% Beech was used for all cells in the study area, including cells where these species cannot grow well, or at all. In practice, species would be chosen to be well suited to the local conditions at the planting site, with the different characteristics of these different tree species dictating the ecosystem services they provide.

The approach used here is scale independent. In this work, we demonstrate the approach at a national level, with a spatial resolution of 1 km^2, and hotspots identified within spatial units of varying sizes. With appropriate data however the approach could be used at a range of scales, from local to global. While the national scale results produced in this case study provide an indication of the most effective areas for afforestation, more detailed mapping could be carried out to optimise specific planting projects, such as those undertaken by large landowners or individual farms and estates, with afforestation by private landowners being the primary means of increasing forest area in the UK in recent years (Burke et al., 2021).

5.5 Conclusion

Through the use of spatial data and predictive ecosystem service modelling, we demonstrate the heterogenous spatial distribution of potential ecosystem service provision from new woodland creation at a national scale. Congruence between the three services modelled is low, such that it is not possible to fully optimise for all simultaneously. However, ecosystem service hotspots were used to identify areas where new woodland creation has the potential to deliver multiple ecosystem service benefits.

Results show that spatial targeting of woodland creation at the national scale has the potential to bring the greatest ecosystem service benefits, but risks overwhelming landscapes with new planting. Spatial targeting within smaller spatial units, such as administrative boundaries, results in more distributed planting, but reductions in benefits compared with nationals-scale planning. Importantly, spatial targeting within even the smallest spatial units was found to bring far greater benefits compared with randomised planting, illustrating the importance of considering ecosystem services in planning locations for woodland creation.

Although demonstrated here at a national scale, using national planting targets, the generalized nature of the approach developed means future work could see it implemented to optimise for differing priorities as expressed through multiple ecosystem service benefits and disbenefits and at a range of spatial scales.

6. Discussion

6.1 Research outcomes and implications

The aim of this thesis was to explore issues surrounding:

1. The quantification of current natural capital assets and associated flows of ecosystem services

2. The estimation and optimisation of future flows of ecosystem services resulting from the creation of new assets.

Through this, the thesis has addressed a number of knowledge gaps, by exploring the following key research questions:

6.1.1 How do we map current natural capital stocks, and value the ecosystem services arising from them?

In this thesis, two natural capital quantification approaches were investigated in contrasting case studies. In the first (chapter 3), a land cover and benefit transfer approach was used for the first time at the individual farm scale to estimate the value of current natural capital stocks in monetary terms. This demonstrated how local scale monetary valuations of natural capital could be produced rapidly by exploiting existing data sets. The method combined existing land cover products with valuation data primarily derived from the Office for National Statistics' UK natural capital accounts (Connors and Philips, 2017), allowing for valuations to be produced rapidly and at low cost, especially when compared with more complex modelling tools which can have substantial requirements in terms of input data, time and expertise (Jackson et al., 2017). The simple nature of this approach means both land cover and valuation data could be easily changed or updated, for example to reflect changes in valuation methodologies, or changes in land use within a farm. This could also be done to assess the impacts of potential future changes in land use on ecosystem service provision, forming a tool to help inform management decisions, which could be explored in future work. However, it was also recognised that there are uncertainties associated with the use of land cover-based proxies for the quantification of ecosystem services (Eigenbrod et al., 2010), and such this chapter also identified a number of refinements to the method which could improve accuracy, specifically through consideration of the condition, configuration (including connectivity) and location of natural capital assets, rather than simply their extent.

Importantly, this work highlighted how the characteristics of land cover data, including spatial and thematic resolution, can have a significant impact on the valuations produced for a farm, the implications of which are discussed further in section 6.1.4. While the impact of land cover data on valuations was the primary focus of this chapter, it should also be noted that the valuation of ecosystem services in monetary terms will also be a source of uncertainty, including through the selection of services to be valued, choice of valuation technique, and inputs and assumptions used in this technique (Boithias et al., 2016). This too will need to be rigorously explored and accounted for in future work if the valuation of ecosystem services is to become a useful tool for the management of land.

6.1.2 Can mapping potential natural capital asset distributions, and modelling future ecosystem service provision from these, be used as a tool to target new habitat creation?

In the second case study (chapters 4 and 5), spatial analysis and predictive modelling techniques were used to estimate future ecosystem service provision from afforestation at a national scale. This work demonstrated how woodland creation could deliver multiple ecosystem service benefits, including carbon sequestration, recreation, and flood mitigation. The level of provision of these services was shown to vary depending on where the new woodland is to be created. The magnitude of this variability was also found to differ between services. While rates of carbon sequestration from woodland were, for the most part, similar regardless of where woodland was located, the provision of flood mitigation and recreation ecosystem service benefits were far more localised. The lack of congruence between these services also meant it was not possible to locate sites for new woodland creation that would fully optimise for all three simultaneously.

Compared with the relatively simple land cover and benefit transfer approach used to quantify current natural capital stocks in chapter 3, the spatial analysis and predictive modelling used in this case study are more complex and computationally intensive. While the ORVal and forestry Ecological Site Classification models, used in calculating recreation and carbon sequestration respectively, are user friendly, they are also to a large extent black boxes, with limited opportunity to modify how they operate. Calculation of flood mitigation, on the other hand, was based on a number of assumptions and simplifications requiring substantive data inputs and processing.

Monetary ecosystem service valuations derived using the land cover and benefit transfer approach (chapter 3), and ecosystem service modelling tools (chapter 5) were found to differ substantially, highlighting how these should not be considered the true value of a natural capital asset, rather a relative indication (appendix 3).

6.1.3 Do existing datasets provide an appropriate evidence-base to inform natural capital-based environmental management?

Through these case studies, this thesis identified and explored a range of issues surrounding the mapping, and measurement, of natural capital and ecosystem services for environmental management. A recurring theme has been the availability of data, and the suitability of the data that is available. Ideally, data should be of good quality, appropriate spatial resolution, readily accessible, timely, well documented and maintained, and consistent across the UK. In practice, key datasets were found to vary considerably across the Home Nations, with a lack of suitable data found particularly in chapter 4, which analysed afforestation in the UK at national scale. In chapter 4, the provisional Agricultural Land Classification (ALC) was used to identify agricultural land quality in England. While a national pre-1988 map exists, this does not differentiate between grades 3a ('good quality') and 3b ('moderate quality'), which between them make up 48% of land in England. More recent mapping, which does differentiate between these two grades, is available only for a limited number of local areas. Therefore, when constructing hypothetical planting scenarios, these had to take the broad approach of either allowing tree planting on both 'good' and 'moderate' quality agricultural land, or disallowing it, with no way to differentiate between the two. Constructing planting scenarios that more precisely distinguish agricultural land quality, such as preventing planting on the 'best and most versatile' agricultural land, which includes grades 1 ('excellent quality'), 2 ('very good quality') and 3a ('good quality'), was not possible.

In other cases, while the required data existed, it was not possible to gain access to it. This was the case with the ALC data for Northern Ireland. As this was unavailable, current Arable and Horticultural land, identified by the CEH Land Cover Map, was used as a proxy for good to moderate quality agricultural land, although this is likely to underestimate the true area. Similarly, maps of the boundaries of Local Nature Reserves in Northern Ireland and Registered Battlefields in Wales were not available, and so these could not be included in the protected areas constraint.

These issues highlight the data requirements for both research around, and planning of, large-scale afforestation, and land use change more generally. In particular, that there needs to be both improvements in our knowledge of the current state of land in the UK, and improvements in the accessibility, openness and consistency of this knowledge, in order to form an appropriate evidence base.

6.1.4 How does scale influence natural capital mapping and ecosystem service estimation?

In addition to the availability and consistency of data sets, this thesis also demonstrated that other characteristics of the data sets used could have a significant impact on the ability to quantify and value natural capital. The spatial and thematic resolutions of the land cover data in a land cover and benefit transfer approach to natural capital quantification is known to have a potentially significant impact on results. However, this is little discussed in existing studies, and indeed many spatially explicit ecosystem service assessments do not even state the resolution at which values are mapped (Schägner et al., 2013).

The implications of using different datasets were explored in chapter 3, which quantified how the characteristics of land cover data, and especially its spatial resolution, can impact valuations of natural capital at the farm scale. In this work, the use of different land cover products caused valuations to vary by up to 58%. The magnitude of these differences was shown to largely depend on the landscape structure of the farm for which natural capital was being valued. This was primarily due to the amount of woodland recorded by each product, with higher resolution products recording larger amounts of woodland, through the inclusion of smaller patches, leading to higher overall valuations. Differences in valuations were therefore found to be greatest at farms where the landscape was highly fragmented, with small patches of land cover, such as groups of trees scattered throughout the farm. At Site 3 for example, annual valuations were found to range from £478,186 to £870,135, a difference of 58%. Conversely, at farms where the landscape was more homogeneous, the differences in valuations produced using each land cover product were far smaller. At Site 5 for example, valuations ranged from £387.564 to £410,721 a difference of just 6%.

These results corroborate previous work which found that land cover data can impact biophysical quantification of ecosystem service flows (Kandziora et al., 2013). This highlights firstly the importance of using appropriate data when quantifying natural capital. Each of the three land cover datasets used in this thesis are widely available and have been used in a range of research projects and publications. The use of any of the three when valuing natural capital would, on the surface, seem reasonable. However, as this work illustrates, the choice of land cover mapping dataset can have a significant impact on the resulting valuations, and ultimately the perceived environmental value of the farms being assessed. Secondly, it highlights the importance of testing valuation approaches in a range of representative real-world settings. If valuations had only been carried out at Site 5 for example, which contains large continuous areas of woodland, results would suggest that coarser resolution datasets are suitable for producing farm scale valuations. Work at other farms however indicates that in other cases, the ability to distinguish small patches of woodland or other land cover types is required, and therefore a high-resolution product is needed.

The scale of the underlying datasets is not the only way that scale influences natural capital quantification. This thesis has also demonstrated how the scale of analysis can affect perceived patterns of ecosystem service provision. In chapter 5, it was shown that spatial targeting of woodland creation within larger spatial units has the potential to deliver significantly greater ecosystem benefits compared to targeting within smaller ones, when optimising for the provision of multiple ecosystem services. With resources for afforestation being finite, it may therefore be desirable for afforestation locations to be planned for at a national scale, rather than at a smaller scale, such as within individual local authority boundaries. However, whilst the greatest ecosystem service benefits were obtained from planning within larger spatial units, planning within smaller areas, such as parishes, also resulted in significantly greater ecosystem service provision compared to random, untargeted planting. This highlights the benefits of explicitly considering ecosystem services when planning changes in land use.

Maximising the provision of multiple ecosystem service benefits in absolute terms is one approach to managing afforestation planning, however other factors must also be considered. Funding for afforestation is often driven by a complex mix of organisations with unique interests and spatial footprints. Local government for example will initiate afforestation projects primarily for the benefit of local residents and constituents. Other approaches may include prioritising planting in areas that currently contain little woodland, or focusing the delivery of benefits to currently deprived areas, and could be explored in future work.

6.1.5 What are the constraints on increasing ecosystem service provision through the creation of new natural capital assets, such as woodland?

In addition to issues surrounding natural capital-based quantification and planning, a further sub-theme explored in this thesis has been the potential difficulties of implementing these plans, namely constraints on the creation of natural capital assets.

This was explored in chapter 4, which presented the first high-resolution national assessment of space available for woodland creation in the UK, accounting for a range of physical, environmental and policy constraints. Approximately 4.7 million ha of land was found to be potentially available for afforestation, even if planting was prevented on good to moderate quality agricultural land and in protected areas. While this is sufficient to meet even the most ambitious planting target of 4 million ha by 2100 (Thomson et al., 2018), this would require planting on a large proportion (85%) of unconstrained land, which is unevenly distributed across the UK. Meeting the target of 1 million ha by 2050, one of a range of Net Zero measures being considered by the UK government (Department for Business Energy and Industrial Strategy, 2017), would require planting on 21% of this available land. This could limit opportunities for spatially targeting woodland creation to maximise ecosystem service benefits. Large amounts of planting concentrated in specific areas also has the potential to negatively impact local communities in and around these areas (van der Horst, 2006), although quantifying this accurately would require more detailed work at the landscape scale compared with the national scale analysis presented here.

It may therefore be preferable to consider relaxing certain constraints, for example by allowing planting within protected areas, if doing so would bring suitable benefits. In essence, this would involve calculating the net, rather than gross benefits of afforestation, and should be an important factor in future work (Section 6.2.2).

6.2 Opportunities for future research

6.2.1 Improving the accuracy of natural capital quantification

It is unrealistic to expect the true benefits provided by nature could ever be fully quantified. However, in this thesis, a number of challenges were identified that provide opportunities to improve natural capital estimates.

In this thesis, a land cover and benefit transfer approach was used to measure current natural capital stocks within farms and estates. There are two main aspects to this approach: the land cover data, and the valuation data. Further work could improve the accuracy of both. In an accuracy assessment, all land cover maps tested in this work were found to contain inaccuracies. The UK Centre for Ecology & Hydrology Land

Cover Map 2015 (LCM2015) (Rowland et al., 2017) was found to be the most accurate product overall, with an accuracy ranging from 78% to 89% in the five farms tested. There is therefore room for more accurate mapping to improve the accuracy of valuations, either through the sourcing or creation of more accurate maps, or the augmentation of base mapping with additional more specialised datasets such as the CEH Woody Linear Features Framework (Scholefield et al., 2016) which records hedges and narrow lines of trees not included in any of the three products studied.

Future work could also utilise accuracy and quality metadata reported within land cover mapping products themselves. LCM2015 for example includes data summarising the number of pixels of each land cover class within each land parcel, with the dominant land cover being assigned to the polygon for display (Centre for Ecology & Hydrology, 2017). This has the potential to be used to account for mixed land cover within these parcels, which may contain patches of different land covers smaller than the minimum mapping unit of the product. This, along with estimated probability which is also reported for each polygon, could be used to better quantify and visualise the uncertainty associated with the product (Robinson et al., 2005). The second major aspect is valuation data. This work used an average monetary value for each natural capital asset, derived from the UK Natural Capital Accounts. While suitable for giving an indication of the relative value of different types of asset, this clearly has the potential for substantial inaccuracies. A native woodland near a population centre and containing public rights of way, for example, will provide a far greater recreation benefit than a remote conifer plantation sited on private land. Future work could therefore consider factors such as the condition, quality, configuration and location of natural capital assets when assessing their value (Figure 3.4), in addition to their extent, although this would be at the cost of additional complexity. This could be achieved both through refinements to the land cover and benefit transfer approach used here, or through the modelling and spatial analysis approaches explored in subsequent chapters.

While chapter 3 primarily explored the impact of land cover data on valuations of natural capital, there are also uncertainties associated with the translation of physical ecosystem service flows into a monetary valuation of the benefits these provide to society. These include the number of ecosystem services considered, the valuation methods used, and the input parameters used in these methods, which can have substantial impacts on valuations, and therefore the perceived value of a natural capital asset (Boithias et al., 2016).

6.2.2 Planning for the creation of natural capital assets

This work also identified potential challenges in the creation of new natural capital assets. Chapter 4 explored constraints on woodland creation in the UK, using three scenarios. However, there is room for future work to more deeply explore the considerations and compromises that will need to be made when planning for large-scale afforestation.

In-particular, the binary nature of the analysis, with land being designated either available or unavailable due to the presence of a broad constraint, left limited room for flexibility. It could be decided for example to allow for some planting to take place on good quality agricultural land under certain circumstances, potentially reducing food security but gaining climate security and other benefits. While converting an arable field would result in a loss of crops, it could also for example provide recreation opportunities if publicly accessible and located near an urban area, and contribute to flood mitigation if sited appropriately in a high flood-risk catchment. This would in essence be considering the net, rather than gross ecosystem service benefits of afforestation. Davis et al. (2019) for example used this approach to identify priority areas for saltmarsh reestablishment in Devon, UK, comparing potential economic benefits with costs such as lost agricultural output and property loss.

The binary nature of the analysis and 1 km^2 resolution also means that scenarios could only consider the wholesale replacement of existing land cover with large, continuous areas of new woodland. However, the integration of less intensive tree planting with existing land use through silvo-arable and silvo-pastoral farming practices could be explored. Rather than replacing agricultural land, tree planting in agroforestry systems has the potential to improve agricultural land through the provision of nutrient management and soil stabilisation, in addition to benefits such as carbon sequestration and the diversification of farm income (Saunders et al., 2013).

Similarly, the 'protected areas' constraint includes different types of conservation areas, designated for a range of purposes. Protected areas like the National Parks and AONBs cover diverse landscapes. While in some protected areas, factors such as the presence of rare habitats or aesthetic landscapes will prevent significant levels of planting, others present an opportunity for substantial areas of new woodland (National Parks England, 2020). Both the services that would be gained from planting, and those that would be lost from the change in land cover, would need to be considered, in essence calculating the net rather than gross benefit of afforestation. A similar approach could also be applied to chapter 5, which in this work focuses on the new benefits of afforestation only. This could be extended by accounting for the costs of woodland creation, both in monetary terms, such as the cost of acquiring land, and the ecosystem services that would be lost or altered from this creation.

Chapter 5 focused on just one method of targeting planting, calculating the average provision of the three ecosystem services modelled and selecting areas where this is highest. This optimises for the delivery of multiple benefits, and identifies areas where afforestation could produce multifunctional woodland, but is not the only approach that could be taken. Optimising different areas of woodland for different purposes is another option, and could result in native woodlands located near urban areas for recreation, and conifer plantations located in cheaper remoter areas for carbon sequestration, rather than attempting to identify locations optimal for both. This approach would also fit some policy-driven funding opportunities for new tree planting that are designed to achieve specific environmental benefits, such as flood mitigation.

Assumptions were also made in the modelling of ecosystem services, which could be explored further in future work. For example, it was assumed that the same combination of beech and Sitka spruce would be planted in all locations. Practically however, species would likely be selected to meet local conditions and requirements, which would impact the services provided. Additionally, in this analysis, each cell is considered independently. A more realistic approach would be to dynamically update the baseline tree map as the process ran, allowing the model to take into account existing and new tree planting before determining the next best place to plant. The presence of a new woodland recreation site in one location, for example, is likely to reduce the value of a second new site in close proximity to it. Similarly, the connectivity of cells could also be accounted for. The value of a single, large site formed by planting in two adjacent cells is likely to be different to the value of two smaller sites considered in isolation, which is not accounted for here.

Finally, future work could involve the modelling of further ecosystem services. In this research, three services were studied: carbon sequestration, recreation, and flood mitigation. These were selected as they are increasingly important, and known to be provided by woodland. However, they are far from the only ones, with woodland being known to provide a range of further services including timber production, biodiversity, pollution removal and soil conservation, amongst many others.

6.3 Summary and conclusions

This is a time of change and transition for land management in the UK, both in our understanding of British landscapes and their relationship with society, and in the makeup of these landscapes themselves. Initiated in response to these developments, this thesis first explored how natural capital and ecosystem services could be quantified within individual farms, amidst calls that post-Brexit, the UK's independent agri-environment policy should do more to support not just the production of food, but also the provision of environmental and societal benefits. These policies, and therefore the context within which this thesis is placed, continue to evolve. Beginning with Basic Payments and the Common Agricultural Policy, there has since been a move to transitionary payments as the UK exited the European Union, which themselves are now being phased out with the introduction of Environmental Land Management Schemes in England (Defra, 2021), and equivalent programmes in other Home Nations.

Throughout this time, debate and discussion has also continued around how landscapes should be managed to respond to the growing threat of climate change. There has been increasing acknowledgement that large-scale action is required, and that tree planting can, and perhaps will need, to play a significant part in this. In 2018, as this thesis progressed, the UK's Climate Change Committee released a document proposing the creation of approximately 2.5 million hectares of new woodland by 2100 (Thomson et al., 2018), the latest in a growing list of calls for large-scale afforestation, which has continued to expand in subsequent years. While initial work in this thesis explored the benefits currently provided by British landscapes, these targets offered a timely opportunity to explore how these could change in the future. The next stages of the thesis, therefore explored the feasability and potential impacts of meeting these proposed targets, as well as how planting could be optimised to deliver multiple ecosystem service benefits.

Through this, this thesis has explored how the concepts of natural capital and ecosystem services can be practically applied to help address current issues in land management today. This work has explored and assessed datasets, and created tools and techniques to better understand the benefits provided by land, and to explore how transitions to new land use could secure further benefits. It is unrealistic to expect that the benefits provided by landscapes and ecosystems, and the interactions these have with society, will ever by fully understood and appreciated. However, this thesis has contributed to progressing knowledge in this area by addressing a number of research gaps. Specifically, the quantification of natural capital at a local scale within individual farms, the suitability of existing datasets for informing natural capital-based environmental management, constraints on the creation of natural capital assets, and the impact of scale on natural capital mapping and ecosystem service estimation. It has also identified a range of challenges and requirements for future research, relating to both the quantification and modelling of natural capital and ecosystem services, and the implementation of changes in land use to improve them. This work therefore represents a single step forward in a far longer ongoing journey. The potential benefits that well planned and managed landscapes can bring to society are great, the challenge, however, will be to make this a reality.

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Appendices

Appendix 1: Supplementary information for Chapter 4

Table A1: Notes on data sources used in constraints.

Constraint	Area	Source	Product	Notes
Base				
Land area	England, Scotland, Wales	Ordnance Survey (OS)	Boundary-Line	Mean high water line used
	Northern Ireland	Ordnance Survey of Northern Ireland (OSNI)	Open Data Largescale Boundaries – NI Outline	
Physical constraints				
Existing woodland	England, Scotland, Wales	Forest Research	National Forest Inventory (NFI)	Assumed woodland, broadleaved, conifer, coppice, coppice with standards, mixed mainly broadleaved, mixed mainly conifer, young trees, ground prep and felled categories included
	Northern Ireland	UK Centre for Ecology & Hydrology (UKCEH)	Land Cover Map 2015 (LCM2015)	Broadleaved woodland and Coniferous woodland classes used
Water, rock and coastal sediment	England, Scotland, Wales, Northern Ireland	UK Centre for Ecology & Hydrology	Land Cover Map 2015	Inland rock, saltwater, freshwater, supra-littoral rock, littoral rock, littoral sediment and saltmarsh classes used
Climatic treeline (600 m)	England, Scotland, Wales	Ordnance Survey	OS Terrain 50	

	Northern Ireland	Ordnance Survey of Northern Ireland	Open Data 50m Digital Terrain Model	
Urban and suburban areas	England, Scotland, Wales, Northern Ireland	UK Centre for Ecology & Hydrology	Land Cover Map 2015	Urban and suburban classes used
Environmental Constraints				
Peat	England, Scotland, Wales, Northern Ireland	British Geological Survey (BGS)	Geology 625k	Superficial geology layer Peat class used
Bog	England, Scotland, Wales, Northern Ireland	UK Centre for Ecology and Hydrology	Land Cover Map 2015	Bog class used
Policy constraints				
Protected areas	England	Natural England, Historic England	Site boundaries.	
	Scotland	Scottish Government, Scottish Natural Heritage, Historic Environment Scotland		
	Wales	Natural Resources Wales, Cadw		
	Northern Ireland	Department of Agriculture, Environment and Rural Affairs (Daera), Department for Communities		
Agricultural land	England	Natural England	Provisional Agricultural Land Classification (ALC)	
	Scotland	Hutton Institute	Land Capability for Agriculture Scotland	
	Wales	Welsh Government	Predictive Agricultural Land Classification	

	Northern Ireland	UK Centre for Ecology & Hydrology	Land Cover Map 2015	Arable and horticulture class used

Table A2: Protected and designated areas used or considered in the analysis.

Area	**England**	**Scotland**	**Wales**	**Northern Ireland**
National Park	Natural England	Scottish Government	Natural Resources Wales	None present
Area of Outstanding Natural Beauty	Natural England	None present	Natural Resources Wales	Daera
National Scenic Area	None present	Scottish Government	None present	None present
Heritage Coast[1]	Natural England	None present	Natural Resources Wales	None present
World Heritage Site	Historic England	Scottish Government	Cadw	Daera
Site of Special Scientific Interest	Natural England	Scottish Natural Heritage	Natural Resources Wales	None present
Area of Special Scientific Interest	None present	None present	None present	Daera
National Nature Reserve	Natural England	Scottish Government	Natural Resources Wales	Daera
Local Nature Reserve	Natural England	Scottish Government	Natural Resources Wales	Data not available
Special Area of Conservation[2]	Natural England	Scottish Natural Heritage	Natural Resources Wales	Daera
Special Protection Area	Natural England	Scottish Government	Natural Resources Wales	Daera

Ramsar Site	Natural England	Scottish Natural Heritage	Natural Resources Wales	Daera
Scheduled Monument	Historic England	Historic Environment Scotland	Cadw	Department for Communities
Historic, Registered or Designated Parks and Gardens	Historic England	Historic Environment Scotland	None present	Department for Communities
Registered Battlefields	Historic England	Historic Environment Scotland	Data not available	None present

[1] Some heritage coasts are defined laterally and do not extend a defined distance in-land, and therefore could not be used in the analysis

[2] Includes Sites of Community Importance (sites have been adopted by the European Commission but not yet formally designated by the UK government) and Candidate SACs (sites have not yet been submitted to the European Commission).

Appendix 2: Supplementary information for Chapter 5

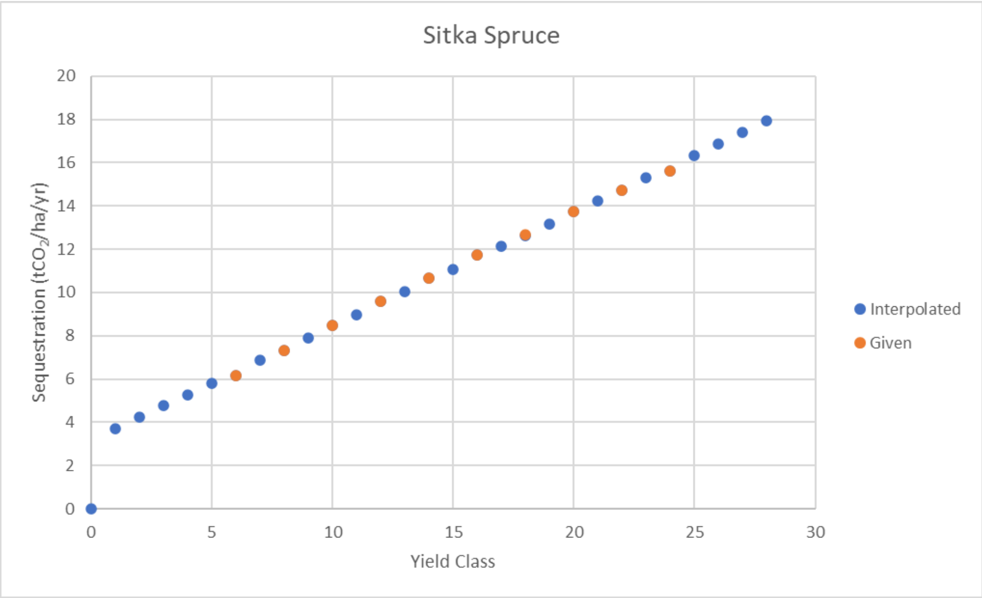

Figure A1: Interpolation of additional sequestration values for Sitka Spruce.

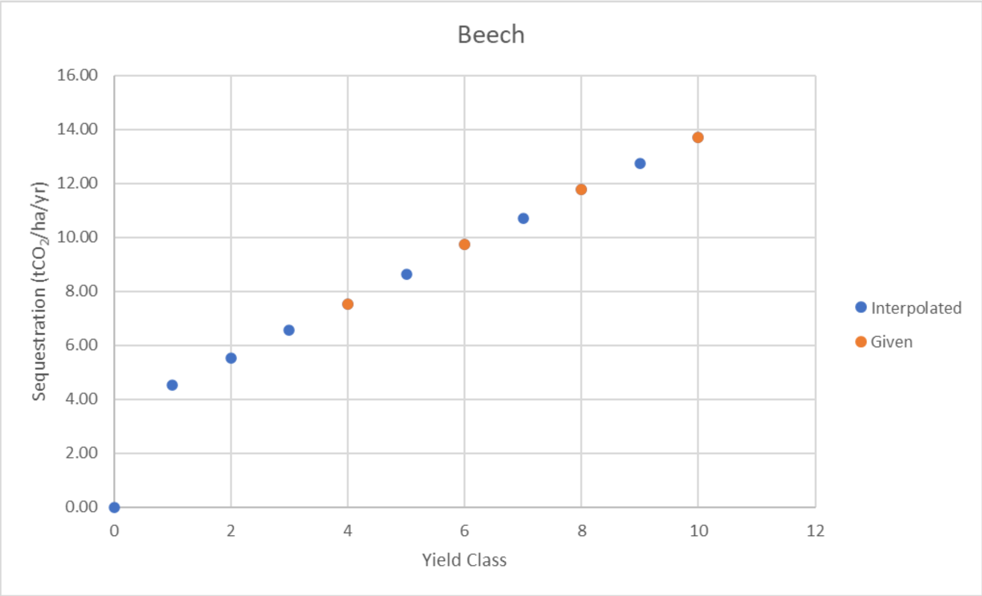

Figure A2: Interpolation of additional sequestration values for Beech.

Appendix 3: Supplementary information for Chapter 6

In this thesis, two case studies were explored across three analytical chapters. The case studies were carried out at contrasting scales, using contrasting methods, and in different contexts. However, both are also fundamentally linked, as part of a broader effort to understand, and ultimately enhance, the benefits we gain from nature.

To demonstrate this, the following section synthesises the results of these three analytical chapters. Beginning with a subset of farms studied in chapter 3, space available for tree planting within each of these is first identified, using the physical and environmental constraint maps created in chapter 4. Next, areas within each farm that would be afforested under each of the 1 km^2 resolution planting scenarios developed in chapter 5 are identified. This is explored in further detail for one farm, Site 3, by quantifying the existing land cover that would be sacrificed for new woodland in each of these scenarios. The validity of using 1 km^2 resolution data, created to plan afforestation at a national scale, to dictate planting within individual farms, is also discussed. Finally, monetary ecosystem service valuations produced for Site 3 using the land cover and benefit transfer approach (chapter 3), and ecosystem service modelling tools (chapter 5) are compared and contrasted, highlighting the uncertainties involved when estimating the benefits provided by natural capital assets.

In chapter 3, current natural capital stocks were quantified within five UK farms. Of these five farms, four are located in England (Figure A3), the area within which planting scenarios were constructed in chapter 5. They are therefore considered further here.

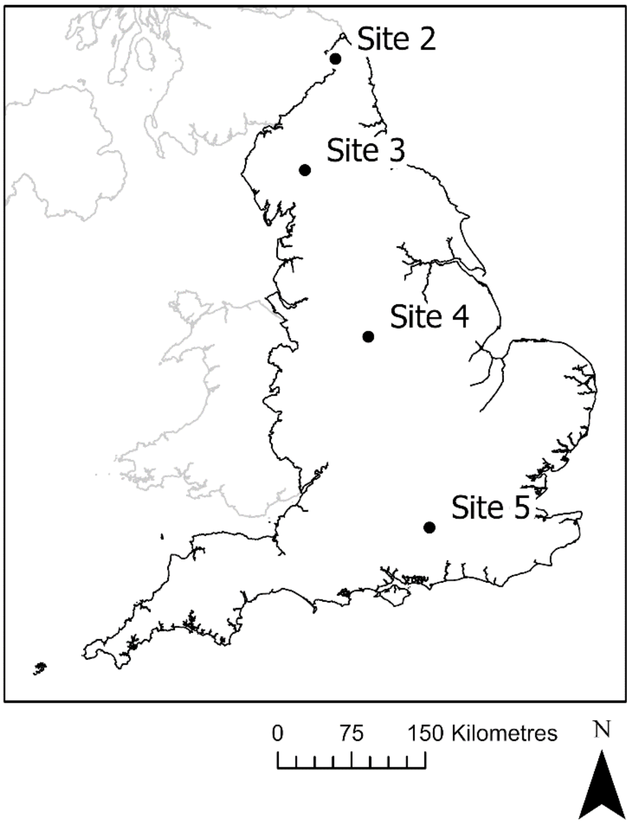

Figure A3: Locations of the four farms studied in chapter 3 that are located in England.

Of these four farms, all contain some land free of the environmental and physical constraints identified in chapter 4, and therefore potentially available for afforestation (Figure A4, Table A3).

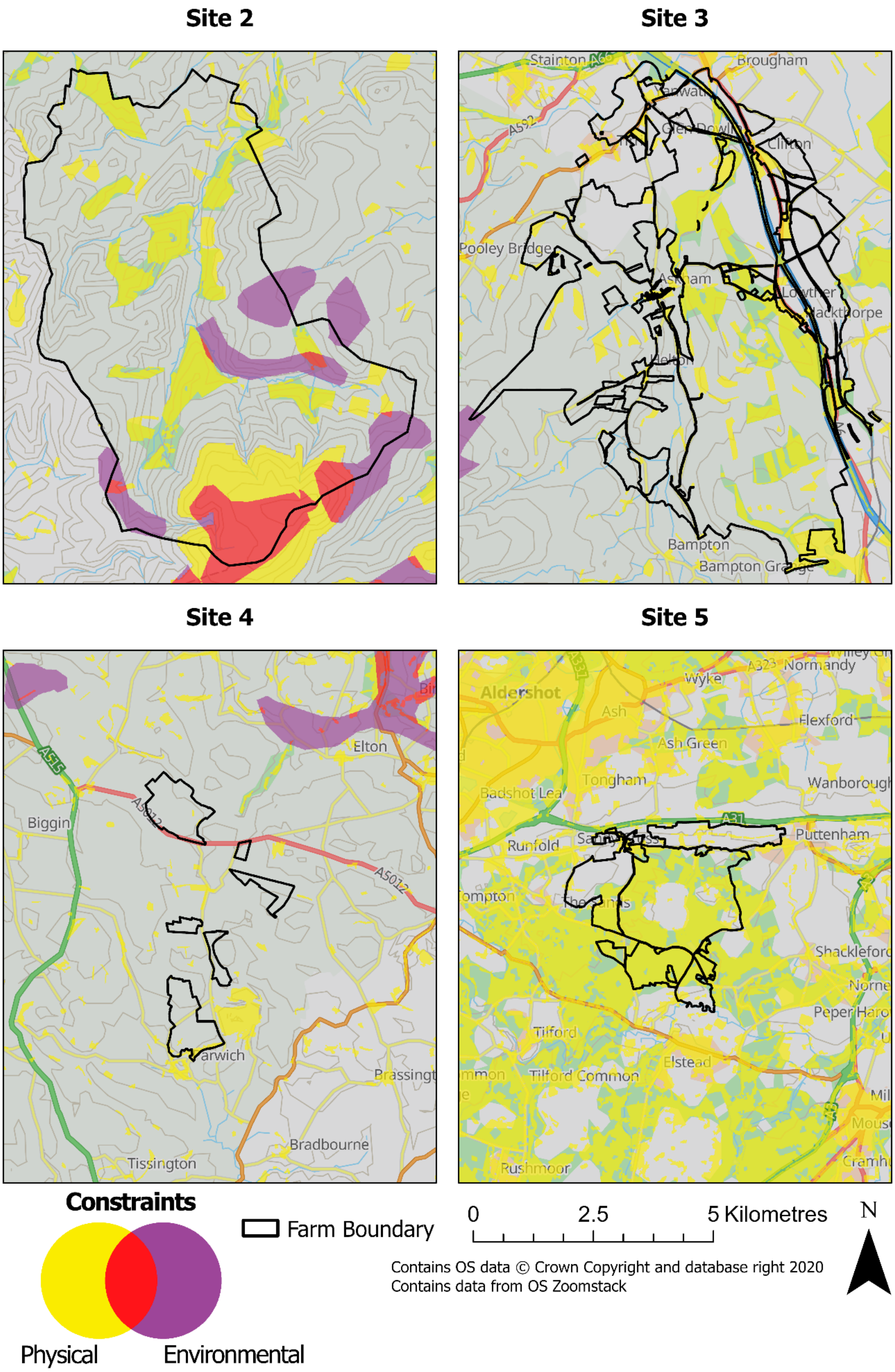

Figure A4: Environmental and physical constraints identified in chapter 4 at the four farms.

Table A3: Constrained and available land at each of the four farms.

Site	Farm Area	Available for planting		Constraints					
				Physical		Environmental		Physical and environmental	
	Ha	Ha	%	Ha	%	Ha	%	Ha	%
Site 2	4,897	3,213	66	1,021	21	365	7	298	6
Site 3	4,157	3,427	82	728	18	1.74	0.04	0	0
Site 4	315	306	97	9.27	3	0	0	0	0
Site 5	900	423	47	477	53	0	0	0	0

In chapter 5, analysis to identify locations where tree planting could optimise the provision of multiple ecosystem services at the national scale was conducted at 1 km^2 spatial resolution. Five planting scenarios were modelled, including spatial targeting of woodland creation at the national scale, and more distributed planting through targeting within smaller spatial units, such as counties and local authorities. Here, the results of this analysis are applied to farms studied in chapter 3 to explore the impacts of these afforestation scenarios at the local scale, and to identify opportunities for woodland creation that deliver multiple ecosystem service benefits.

When creating afforestation scenarios in chapter 5, the 10 m spatial resolution physical and environmental constraints identified in chapter 4 (Figure A4) were resampled to 1 km^2 using a majority rule. 1 km^2 grid cells covered 50% or more by physical or environmental constraints were designated as being constrained, and therefore unavailable for planting. Remaining cells, with less than half their area covered by physical or environmental constraints, were designated as being unconstrained, and could therefore be selected for planting in the afforestation scenarios.

Of the four farms studied in this section, all contain a number of unconstrained grid cells, and therefore contain land that was identified in the chapter 5 analysis as being available for afforestation (Figure A5).

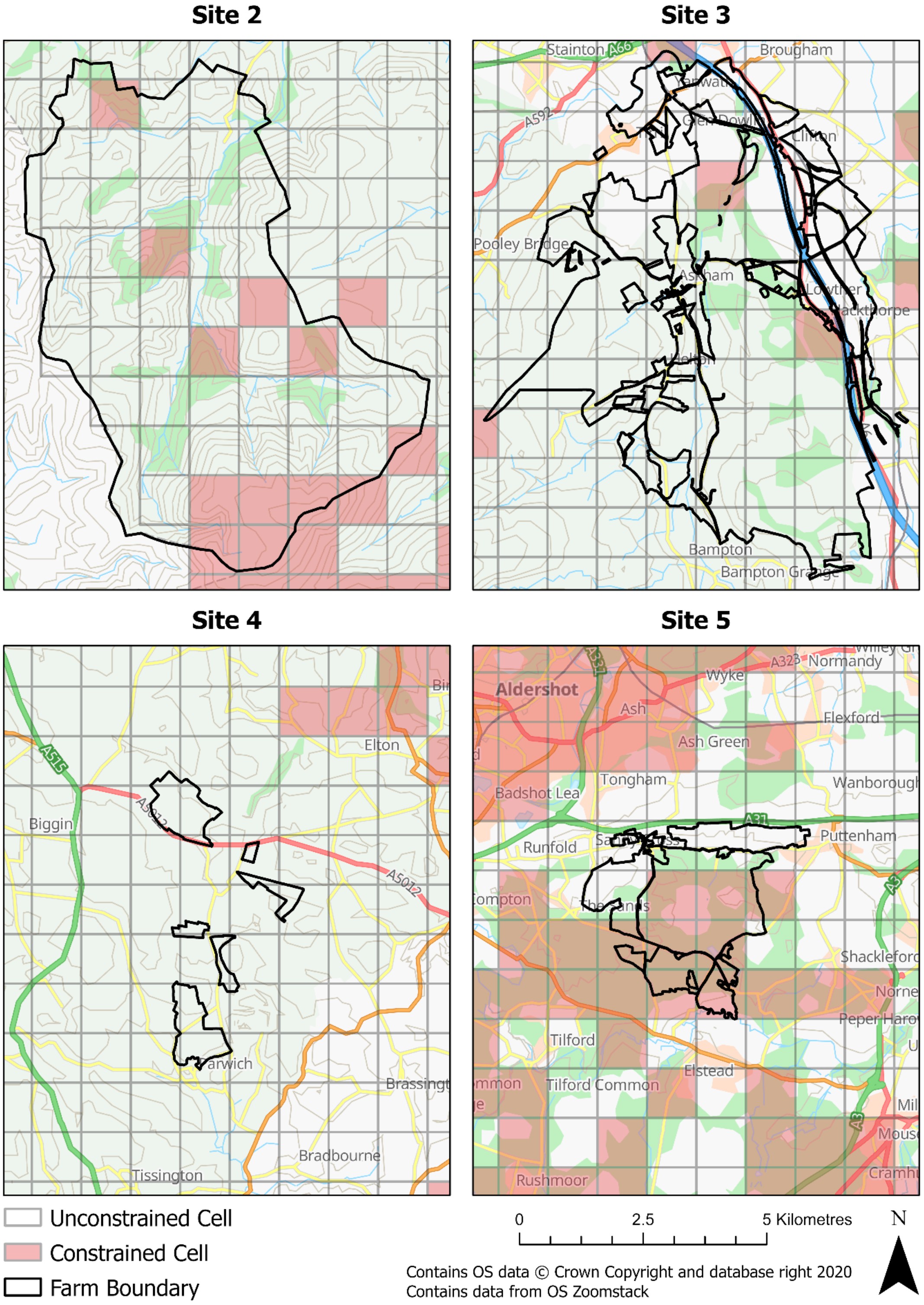

Figure A5: Constrained and unconstrained land at the four farms, as used in chapter 5.

Each of the four farms also contain land that would be planted in one or more of the five scenarios modelled in chapter 5 (Table A4). Notably, Site 3, a large mixed usage estate in Cumbria, sees both the most planting in each scenario, and also the greatest differences in the level of planting between scenarios (Figure A6), and so is examined here in further detail.

Table A4: Number of cells at each farm identified for planting in chapter 5 for each planning scenario.

	National Scenario	**Regions Scenario**	**Counties Scenario**	**Districts Scenario**	**Parishes Scenario**
Site 2	0	0	2	0	1
Site 3	45	21	18	15	4
Site 4	0	1	0	0	1
Site 5	0	0	2	2	3

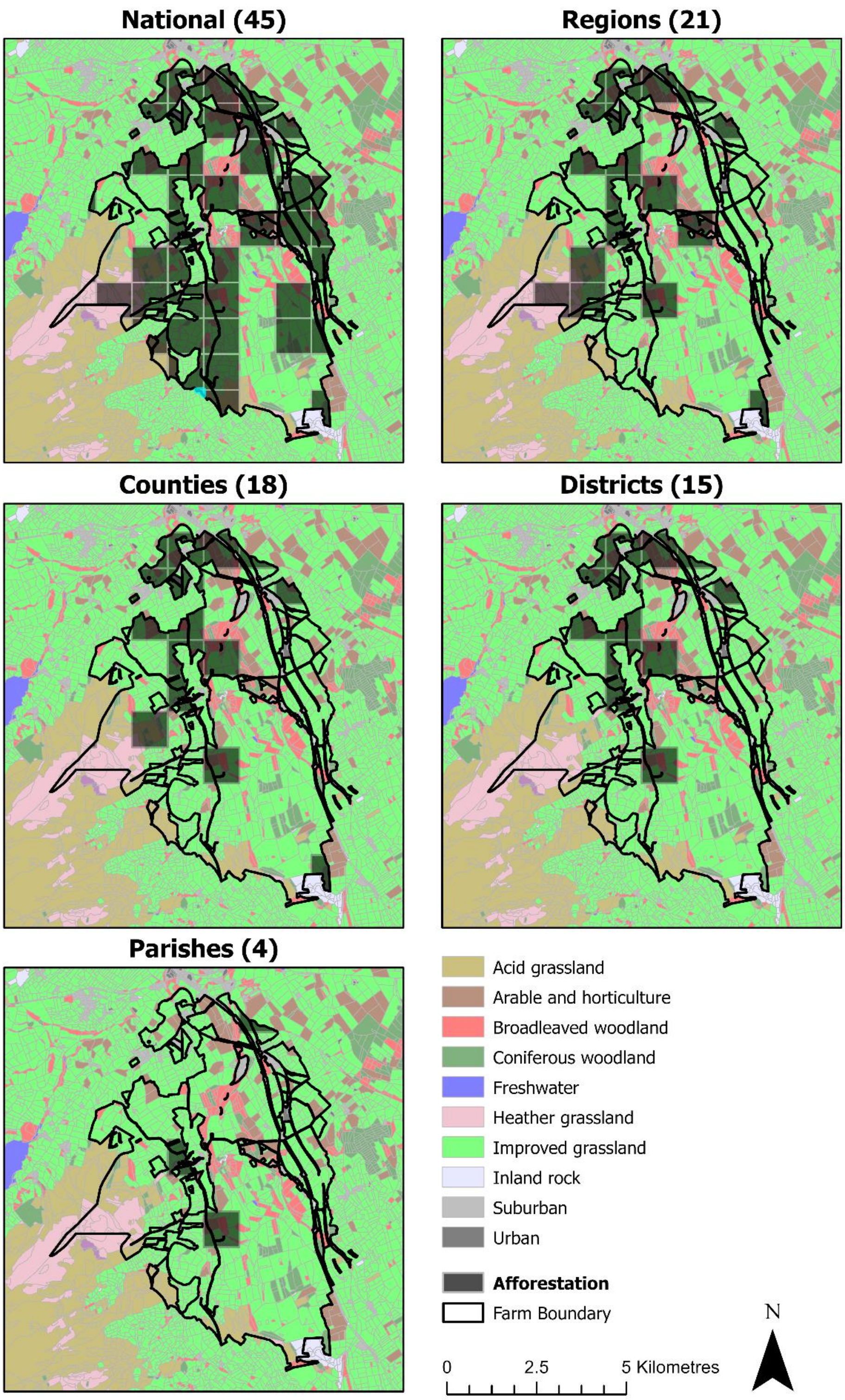

Figure A6: Locations where additional planting would occur at Site 3 under each of the scenarios developed in chapter 5. Numbers in brackets indicate the number of grid cells in the site that would be planted.

Any planting within a farm will necessitate the conversion of existing land cover. In all planting scenarios at Site 3, this replaced land cover is primarily improved grassland, typically used for grazing livestock, with smaller amounts of arable and horticultural land, and other land cover types (Table A5). In the *National* planting scenario, 2,477 ha, over half of the farm's area, would be afforested, which would require a substantial change in the farm's business model, and could impact food security if widely replicated elsewhere. In the *Regions*, *Counties*, *Districts*, and *Parishes* scenarios, afforestation is more widely distributed across the country, and less planting is proposed for this particular farm. It may therefore be possible to implement these plans with less disruption to existing farm activities. In all scenarios however, the amount of planting being proposed is substantial. Appropriate policies would therefore be required to encourage and support this transition in land use.

Table A5: Existing land cover that would be replaced by new woodland under each scenario at Site 3, measured using LCM2015.

Land cover	Current Farm Total (Ha)	Replaced in Each Scenario (Ha)				
		National	Regions	Counties	Districts	Parishes
Acid grassland	307	147	55	36	1	1
Arable and horticulture	301	247	131	111	110	18
Broadleaf woodland	431	193	101	82	78	19
Coniferous woodland	196	88	42	29	5	2
Freshwater	1	0	0	0	0	0
Heather grassland	167	99	99	14	0	0
Improved grassland	2697	1674	638	556	501	130
Inland rock	18	1	1	1	0	0
Suburban	36	27	17	15	15	6
Urban	1	1	1	1	1	0
Total	4157	2477	1085	844	711	175

In chapter 3, the land cover and benefit transfer approach was used to estimate the monetary value of ecosystem services provided by natural capital assets in individual farms, including Site 3. Using LCM 2015, which was found to be the most accurate land cover product, the total monetary value of these services was estimated to be £870,135 per annum. In the *National* planting scenario explored in chapter 5, 2,477 ha of new woodland would be created in the farm. This planting would increase the estimated value of ecosystem services provided at the farm to £2,239,940 per annum, accounting for services lost from replaced land cover. The afforestation planned in the

National planting scenario could therefore substantially increase the ecosystem service value of the site, at the expense of disrupting the existing farm business.

In chapter 5, the ORval model was used to value woodland recreation in monetary terms at a 1 km^2 resolution, while carbon sequestration was quantified in physical terms, and flood mitigation was quantified using a unitless value. Using the ORval model, woodland recreation at Site 3 was valued at between £20,093/km^2/a and £282,739/km^2/a, depending on the location of the woodland, with the highest values being found at the north of the farm, at the southern edge of the town of Penrith.

Using this approach, afforesting the 45 1 km^2 grid cells from the *National* planting scenario that intersect with Site 3 would result in an annual gross benefit of £3,182,927, not accounting for services lost from replaced land cover. This can be compared with the land cover and benefit transfer approach used in chapter 3 where woodland recreation was valued at a constant rate of £9,200/km^2/a, using data derived from the UK Natural Capital Accounts. Using this approach, planting the same 45 cells would result in a value of £414,000, substantially lower. Using different valuation approaches can therefore give very different results, highlighting how these should not be considered the true value of a natural capital asset, rather a relative indication.

The 1 km^2 resolution grid, used in chapter 5 to identify land for afforestation, is suitable for broad planning at a national scale, and for identifying sites that warrant further investigation, such as Site 3. However, the coarse resolution grid is clearly not optimal for planning for afforestation within individual farms and estates. This is evident in Figure 4 and Table 3, where some proposed new planting would replace existing broadleaf and coniferous woodland, and take place on physically unsuitable areas such as rock and freshwater, with these small features lost when the constraints data identifying them was upscaled to 1 km^2. These national scale, coarser resolution plans also leave little room for flexibility, with the choice of planting either 1 km^2 of new woodland or none at all being unrealistic, especially for smaller farms. While they may be useful for presenting a broad indication of opportunities for planting within a site, practically implementing this would require more detailed higher resolution mapping.