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"No trees in the wrong place"

defining multifunctional priorities for woodland expansion strategy to meet national climate targets in the Cairngorms National Park"

PhD Student

Baggio Compagnucci Andrea

Tutors

Dott. Alessandro Gimona **Prof. Roberto Canullo**

Coordinatore curriculum

Prof.ssa Anna Maria Eleuteri

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Chapter 1 – Introduction, aims and objectives

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1.1 Introduction

Forests are the repository of much of the world's biodiversity, and therefore foresters must assume a degree of responsibility for its management and conservation. (Kapos and Iremonger, 1998). Maintaining such biological diversity it is now one of the most important goals of managing forests in a sustainable way and to address this need, biodiversity conservation organizations have proposed nine templates of global priorities over the past decade. (Brooks et al., 2006). However, if forest conservation priorities are well recognised globally, the process for understanding the distribution of species and ecosystems locally is scale-dependent (Lindborg et al., 2017). On the other hand, data that are available at a global scale are still typically sparse and of varying quality, while locally choices are still driven by detailed data.

Managers and policy makers need to be cognizant of the biological significance of the forests they manage in a broad context, avoiding to compromise global biodiversity goals by managing their forests inappropriately. Therefore, to achieve this important management target it is crucial that managers be fully informed (Noss, 1999) on the status, condition, conservation value of each forest, and change in forest conditions over time. This thesis addresses these questions by examining a range of data-driven spatially explicit approaches with the purpose of supporting the assessment of potential impacts of different policy and climate scenarios on the Scottish forest-based sector. The specific forest types characterization and potentialities are guided by information at two levels: bottom-up models based on local characteristics of each site; and an overarching, top-down, national-level national policy for net carbon sequestration. In the forest Scottish context detailed local scale case studies are still lacking in incorporating the policy context and the ecosystem service approach, introduced in Chapter 2, to meet national strategic targets. Beside, human activities within and nearby the protected areas boundaries have increased the pressure on forest and the services they deliver, exacerbating the concept that management and land-climate systems need to coexist, and pursue the same sustainable development. The Cairngorms National Park represent an example, that may possibly encourage dialogue between different actors for the mutual advantage of using tools that facilitate the visualisation of constraints and opportunities for the forestry sector. In Chapter 3, the specific bio-physical assessment in the area and socio-economic drivers and barriers to change in Cairngorms is presented.

Forests as carbon sinks, therefore, are required to play a multifunctional role that includes, but is not limited to, biodiversity conservation and maintenance of ecosystem functions; yield of goods and services to the society; enhancing the carbon storage in trees, woody vegetation and soils; and providing social and economic well-being of people (Pandey, 2002). Evaluating how climate mitigation measures (e.g. woodland expansion) may have unconsidered effect on other ecosystems and forest functions has become increasingly important in the study of long-term maintenance of biodiversity (Peters and Darling, 1985, Van der Plas et al., 2018, Minang et al., 2014). However, works dealing with modelling of the impacts of climate change on woodland species dynamics in Scotland to address carbon sequestration, one of the pillars in global mitigation, are limited. An attempt to model the future distribution of broadleaved and conifer species at Scotland scale, and investigate the potential impact on soil carbon is described in Chapter 4.

Natural protected areas provide valuable services to society, including the supply and purification of fresh water (Postel et al., 2005, García-Nieto et al., 2013., Birch et al., 2014). Ecosystem services modelling tools has been widely compared (Dennedy-Frank et al., 2016, Rosenzweig et al., 2014, Cheaib et al., 2012, Vigerstol and Aukema, 2011), with a quite relevant concern in validation and accurancy, however successfully attempt in hydrological ecosystem (Redhead eta la., 2016) have recently raised the attention around InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs, Sharp et al., 2015). In Chapter 5, a methodological approach to nutrient and sediment retention taking account of modelling response to options of woodland expansion is established, through the integration of ideas developed in the Interim report for the Scottish Government (Gimona et al., 2019).

Adaptation and resilience cannot be achieved without credible and robust information on climate change and its variability is needed to inform decision-making. UKCP18 is the most up to date national climate projections for the United Kingdom and will provide users with the most recent scientific evidence on projected climate changes with which to plan. The UKCP18 trends can drive future fluvial flooding which, already increasingly nowadays, can heavily threats the hydraulic and biological process of the flood plains. Engineering solutions seem insufficient to maintain low flood risk without affecting biota components (Talbot at al., 2018, Nedkov et al., 2012), hence natural catchment-based adaptation measures (e.g. Natural Flood Management, NFM) are likely required (Wilkinson et al., 2019, Iacob et al., 2016, Nisbet at al., 2011). Afforestation is one of the measures that can increase infiltration rates associated with improved soil structure and macropore formation (Eldridge and Freudenberger 2005). Target areas for spatial decision support and NFM approaches can be implemented through the use of Geographical Information Systems (GIS) providing an excellent opportunity for integrating with multi-criterion evaluation results (Jankowski et al., 2001). A suitability model of the occurrence of flooding risk and opportunity for afforestation in the Cairngorms National Park has been developed. This work is described in Chapter 6.

The persistence of native species in fragmented landscapes is dependent on dispersal or foraging movements between habitat patches, which may be limited. Although corridors have been heralded as solutions, their effectiveness depends on species' movement behaviour, which has rarely been studied (Doerr et al., 2011). Chapter 7 brings together results of potential dispersal connectivity for generic species in broadleaved woodland and specialized birds for native conifer. The connectivity paradigm here is defined by a prediction of movement patters in complex landscape based on circuit theory software. Such models identified the spatial opportunities for new trees that can act as stepping stones, increasing connectivity and facilitating range expansion (Rossi et al., 2016).

Additional spatial data, not obtained by modelling methods, and the creation of the baseline land cover map to use for generation land use change scenarios is discusses in details in chapter 8; while results of four woodland expansion scenarios in the Cairngorms to meet the rate of national strategic target are presented in chapter 9. The outcomes are the simulation of forest land managers that can benefits from tools (e.g. spatial Multi Criteria Analysis, sMCA) to identify win-win functions and avoid unintented negative effects.

Chapter 10 draws conclusions regarding this work. The management of all natural resources must now meet both national and local targets and guidelines. To achieve this stakeholders such as policy makers, managers, ecologists, foresters, and field rangers must have access to both spatial data and tools. Combining, GIS, statistical spatial models, specific ecological software and open-source frameworks and integrating data in a computer-based platform to let decisions managers explore options is therefore crucial to simulate and define multiple benefits. The main task of this research was to find a means to implement and integrate all the specific outcomes in the Cairngorms National Park area and outline the implication, described in chapter 11, of such effort.

The specific objectives of this work therefore were:

1. To define areas for net positive soil carbon sequestration through woodland expansion in Scotland accounting for climate changes (2050-2070).

2. To parametrize and use model in water purification service in Scotland assessing some of the consequences of scenarios of broadleaved land use change.

3. To outline priority areas for implement Natural Flood Management in Scotland with the use of spatial analysis.

4. Defining the potential dispersal pathways addressing native conifer and broadleaved species to enhance connectivity in the Cairngorms National Park.

5. To discuss the usability and usefulness of MCDM methods from the viewpoint of supporting forestry decision making, identifying priority areas in the National Park for native woodland creation.

6. To map the options of woodland expansion in the Cairngorms National Park to meet national climate target.

7. To examine differences in the results of simulating different stakeholders opinions in defining priorities to the four chosen criteria.

8. To review the current Cairngorms National Park Forestry Strategy 2018.

Chapter 2 – Research context: the use of Ecosystem Services to define forest multifunctionality

2

2.1 Ecosystem Services approach

In the last 20 years the ES concept emerged through research projects aimed to enhance the protection of the system that host and sustain humanity (MEA 2005), such as the economic valuation of ES (Costanza et al. 1998, Kumar 2012), the environmental management (De Groot et al. 2010) and the classification of ES in broader groups based on the benefits they provide to society and economic prosperity (Watson et al. 2011). In same time the Common International Classification of Ecosystem Services (CICES) (Fig. 2.1) was proposed to account for spatial relationships between the source of the services and the beneficiaries (Haines-Young and Potschin 2011), although the debate is still open whether the biodiversity should be interpreted as an ecosystem service (Mace et al. 2012).

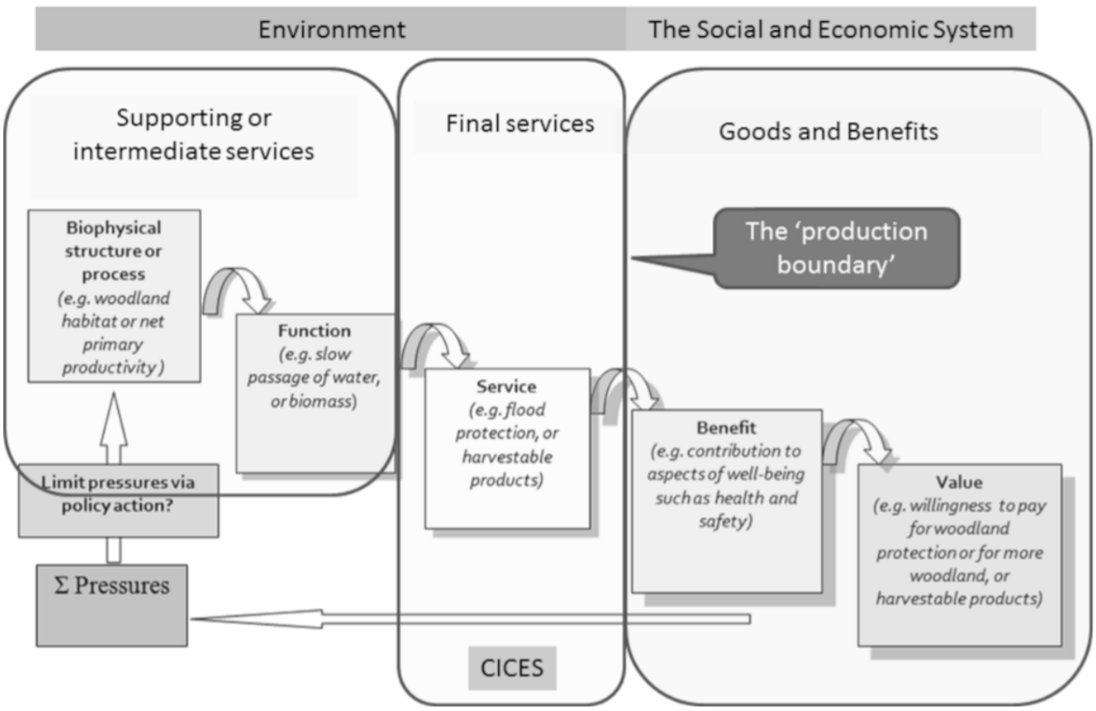


Figure 2.1: The cascade model adapted from Haines-Young and Potschin 2011

Humans shaped and manipulated the ecosystems greatly and extremely rapidly in accordance with the growing demands of food, fresh water, timber, fibre and fuel and this request has often resulted in conversion of natural landscape practices into human-dominated lands (Foley et al. 2005). In real world context, each ecosystem is not independent, but instead exhibits complex interactions with other nearby systems. For instance, the reduction of forests to increase timber and agricultural production is one of the most important drivers of species extinction rates which are now 1000 times higher than in the fossil record (Hassan et al. 2005). In addition, deforestation and the land use change on favour of intensive crop production activities, using high quantity of fertilizer, has increased N and P concentration in aquatic ecosystems, leading to low productivity itself, decreasing fish population, and the creation of hypoxic zones. Since 1850, roughly 35% of anthropogenic CO2 emissions resulted directly from land use (Houghton et al. 2001), and in particular forest ecosystem have been overused to meet the resources demand of world population and support the induced pressure (Kochli and Brang 2005, Haberl et al. 2007, Lafortezza et al. 2013). Recent studies suggest that the world will need 70 to 100% more food by 2050 (Akram-Lodhi 2008) and in the upcoming decades, how we find alternatives ways to meet the increasing demand of resources and services will determine the future condition of the ecosystem and the life style of the global population (Bennett and Balvanera 2007, Herrick and Sarukhan 2007). We already experience worldwide situations where the provision of one service has a direct or indirect impact on another service. This interaction suggest that ES research should be addressed to improve our understanding of the relationships among ESs and our landscape management (Bennett et al. 2009). Furthermore, the multifunctionality of the various ecosystems focus on maintaining same level of use of the services (optimum) is unrealistic while compromising is an achievable target in line with the real ecological process. The future research challenges will therefore consider the analysis of trade-off between ESs and the identification of good compromises (synergy) to enhance sustainable use of multiple ES to avoid unwanted irreversible effect on land use and maintaining of long-term ES provisioning.

2.2 Forest landscape planning: priorities and multifunctionality

Each choice involves the definition of priorities. If you recently move to another country, you might consider what kind of life style to have among different options. For instance, you might find yourself having to decide whether you want to cycle to work or you want to buy a car and once you are in the bicycle shop you have to decide between a road bike or a mountain bike, moving forward, once you are on the road you will select the way to cycle to work among the cycle network available and so on. Most of us are not in the comfortable position to buy or achieve everything we desire. We must put thoughts into every purchase and consider how it affects our bank account. We also must think about what type of satisfaction that purchase will give us. As a result, to get one thing that we like, we usually have to give up another thing that we also may like. Therefore, making decisions requires trading off one item against another. However, what is essential is how you prioritise your choices and how much knowledge you have on the topic.

The roots of this concept come from the Latin word Priorem (*previous, in front, better*) commonly used to define the leader of a monastery or mendicant order (Prior) which was described as "First among equals", hence we can define the prioritisation as "what you put in front or what is the better choice for you" among multiple possible options.

Managing public land and landscape in general also involves trade-offs. Choices about whether, where and when harvest timber or and where to expand agricultural rather than forest lands involve trade-off among numerous factors that differ across landscape and over time (Kline and Mazzotta 2012). However, land is a limited resource that fulfils multiple functions (Seppelt et al. 2013) and the planning process might not be simple as the choice between the bicycle and the car example because requires the correct definitions of constraints and opportunities, the formulation of alternative plans and the evaluation of their consequences. In fact, the management decision can cause undesirable effects if it lacks understanding of the complex nature of ecosystem which lead to the multifunctionality of land systems (Bennett et al. 2009). In other words, maximization of a single function (or ES), for instance productivity, can feedback negatively on several others ecosystem services and functions with cascading effects on human well-being (Holling 1996, Bennett et al. 2009, Maes et al. 2012). Because most of the ESs shown to be interactive is important to assess how the services are distributed across the landscape and how they interact each other. The number of publications has risen rapidly in the last decades since the concept of trade-off has been originally developed in economics and the study of the relationships has been used to describe different types of compromises occurring in management of ES (Lee and Lautenbach 2016, Cord et al. 2017), compromises between ES (Bennett et al. 2009), between generations (Rodriguez et al. 2006), between ES provision and demand (Mouchet et al. 2014) or between beneficiaries (Martin-Lopez et al. 2012).

Following the general use in literature the relationships of ES pairs can be categorized into "trade-off" (one service increase while another one decrease), "synergy" (both services increase or decrease together) and "no-effect" (Jopke, 2015, Vallet, 2018).

In many cases, the study of the relationships involved the assumption of a linear correlation between ESs (Raudsepp-Hearne et al. 2010, Vallet et al. 2018) while they were trying to assess landscapes as areas which consist of bundles of services. In opposite direction Koch et al. 2009 demonstrated the absence of a linearity between ESs. Therefore, the discussion around relationships between ESs is still very much open and further study will be surely contribute to define the relevant indicators, the biophysical processes, and to examine the interconnection among them, and even their relationships across scales.

Recent studies (Sweeney et al., 2004; Daily et al., 2009; Polasky et al., 2011) have identified the upstream and downstream relationship on some of ESS confirming how consequences and causes of the same stress can define different pressure locations when land use change transformation occurs. It is clear that understanding the interaction of all these spatial and temporal complexities is essential to define the trade-off and identify the drivers in order to find solutions for adaptive capacity and regional vulnerability (Lindner et al. 2010). Moreover, this challenge represents the main goal to locate whether and where the woodland can be moved helping the effective management practice (Rodriguez et al. 2006; Bennet et al., 2009; Garcia-Gonzalo et al., 2015).

In the forestry sector, the conservation and protected areas became a primary need to continue to deliver the ESS as the correlation assessment show direct relationship between designations and multiple services such as water purification, carbon sequestration, and crop pollination (Naidoo at al., 2008) yet it has been demonstrated that protected sites deliver overall higher levels of ESS than non-protected sites (Eastwood et al., 2016). These outcomes highlight even more the importance of

ESS if we consider that some of the benefits are addressed even outside the borders of the areas of protection and conservation. According to many authors (Naidoo et al., 2006, Turner et al., 2010), it is important that woodland management consider potential corridors for improving connectivity through a landscape ecological prospective emphasizing the biodiversity retention and reducing the fragmentation of the habitat (Merendeler et al., 1998; Van der Horst & Gimona., 2005, Gimona et al., 2015). Because the location and the extent of afforestation could be detrimental or beneficial, spatial analysis is an essential tool for minimizing conflicts and maximizing synergies, therefore spatial targeting for decision making requires a definition of the appropriate ES scale (Swetnam et al., 2011) and level of knowledge.

2.3 The importance of stakeholders and the participatory approach

Similarly, if the planning process need a deep understanding of the interaction between ESs and the formulation of options to consider the consequence of the landscape strategies, a significant contribution to narrow the appropriate conservation program can arrive from the stakeholders. Their participation and transparency in prioritizing the target ecosystems and services aim to focus attention on what is important for different sectoral interests, such as conservationist (Logsdon et al., 2015; Ujházy et al., 2020), local users (Palacios-Agundez et al., 2014), businesses (Houdet et al., 2012; Zucchella et al., 2019), and tourism (Brown and Weber, 2012; Pomeroy and Douvere, 2008). In South Carolina, USA, Ureta et al., 2020, resident of the state became aware that the forest ecosystem's direct linkage to water-related ecosystem services, therefore, they opt to choose to conserve the ecosystem that also enhances water quality as their primary priority ecosystem service. Thus, in planning for conservation interventions, transdisciplinary methods that integrate collective social opinion with biophysical angles can foster a wide consensus among all the stakeholders involved. Following the argumentation, multiple studies have been undertaken in the direction of mapping different ecosystem services interest, perception and knowledge linked with different stakeholder profiles (García-Nieto et al., 2015) and spatial scale difference (Hein et al., 2006) using workshops (Palomo et al., 2011), questionnaires (Martín-López et al., 2012) or interviews (Klain and Chan, 2012).

2.4 Tools for modelling ecosystems and services

2.4.1 Geographic Information Systems (GIS)

The modern concept of landscape ecology has developed in the last few decades following the growing trend of technological application and solutions such as geographical information systems (GIS) and remote sensing (RS). Is largely recognised that a combination and integration of these technologies brought a substantial revolution in the ecological studies offering numerous advantages in data collection, modelling, result analysis and visualization (Longley et al., 2005; Steiniger and Hay, 2009). In a structure as heterogenous as an ecosystem it is particular appealing the use of technologies that can pragmatically represent biological and physical processes, linked in practice with their ecological dimension. Therefore, representing all these complexities in different spatial explicit levels is strategic and recently become a regular proceeding (Singh et al., 2010).

GIS are nowadays fully embedded in the ecological studies and represent a fundamental tool to manage data over a wide range of spatial and temporal scales and create the platform to bridge landscape spatial structure and statistical and mathematical models. A platform that can handle a huge quantity of data such a GIS requires from research institutes and academies to migrate parts of big-data analysis into “the cloud” computing (Salt et. Al., 2018). High Performance Computing via shared pools of configurable machines become important to research because of its accessibility and potentially lower infrastructure requirements and costs. Some software and applications are already ideally suited to take advantage of this computing development, in particular those with perfectly parallel processing. Salt et al., 2018 at The James Hutton Institute, demonstrated that the technology and tools exist to automate the running of computationally-intensive modelling tasks. Utilising cloud computing infrastructure as a service permits to tackle large datasets, distributed over many computations on a need-by-need basis. This approach is extremely scalable; it allows substantial reduction in computation time when modelling large areas and enhances the reproducibility of results, hence is extremely suitable to macro-ecological studies to understand environment and species richness, habitat and species transitions and losses, landscape level solutions to adaptation and mitigation strategies to global climate change.

Finally, 3D GIS technique recently became a useful branch of research studies to promote the development of landscape ecology and visualize specific results and future landscape options. These challenges were explored with a case study of virtual forest landscape in the Cairngorms National Park (CNP) which was used to test preferences for scenarios of future woodland expansion (Wang et al., 2020). Audience feedback suggested that the enhancement of user interaction through VR has potential implications for the planning of future woodland to increase the effectiveness of their use and contribution to wider sustainable ecosystems.

2.4.2 Generalized Additive Models (GAMs)

The use of GIS tools in ecology and the parallel increasing application of advancedstatistical techniques contributes to construct predictive habitat distribution models. Accessibility to modelling gained success and momentum since start to investigate the implication of land use change on the distribution of flora and fauna connecting the geographical abundance of species with the landscape characteristics (Guisan and Theurillat, 2000; Mourell and Ezcurra, 1996). The variety of these models have grown exponentially since a wide range of application started from biogeography (Bohm and Popescu, 2016), to climate change research (Alexander et al., 2018), and even habitat or species management is represented with very specific model design to simulate plant communities (Yee et al., 1991; Bolker eta al., 2003; Brzeziecki et al., 1993), aquatic plants (Vis et al., 2003; Lehmann et al., 1997), fish (Olden et al., 2002; Oberdorff et al., 2001), and terrestrial species richness (Elith et al., 2009; Berry eta al., 2003).

Surely statistical consideration "*a priori*" should drive the orientation of the method for implementing while conceptual considerations link the trade-off between generality, reality and precision forged by Levins, 1966 where only two out of three model properties can be simultaneously improved. Since the introduction of its generalized form, linear regression (GLM) evolved allowing that response follow any distribution from exponential family (e.g. Gaussian, binomial, Poisson and gamma). A locally weighted extension of GML is the Generalized Additive Model (GAM) in which part of the linear predictor is specified in terms of a sum of smooth functions of predictor variables (Wood, 2006). While methods have been introduced for testing the accuracy of predictive models (e.g. bootstrap, cross-validation), GAM approach enables the use of more complex models for the "random effect" component of data, thereby improving our ability to model correlated data. Given that the applied modelling involves using computer programs, gam() functions have been developed as packages in the years for both licenced and free available software. R is one of such software currently available at https://www.r-project.org, it runs under a variety of platforms, and has MGCV packages (Wood, 2000) for the use of gam() function. Although R was born exclusively to be use as a statistical framework, it is undergoing an object-oriented transition integrating statistical and spatial analysis. Over the last twenty years, ecologists realized that the explained component in the spatial dependence of ecological process (King et al., 2004; Reich et al., 2004). is quite high, and the availability of computer hardware and software have been considered equally important as examining the spatial relationships of these process.

2.4.3 Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST)

Ecosystem services modelling is fundamental to identify priority areas of intervention and can anticipate consequences of changes made to ecosystems (Vigerstol and Aukema, 2011). Among some of the emerging tools for quantifying ES are the Multi-scale Integrated Model of Ecosystem Services (MIMES), the ARtificial Intelligence for Ecosystem Services (ARIES), the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST), Co$ting Nature, and the reconceptualized application of the Soil and Water Assessment Tool (SWAT) (Boumans et al., 2015; Gómez-Baggethun et al., 2014; Bagstad et al., 2013a; Sharp et al., 2015, Mulligan et al., 2010; Arnold and Fohrer, 2005). The majority of these tools requires different inputs and parametrization of the underlying biophysical models however, they find shared fields in simulating the effect of land management over natural systems. InVEST framework is a modular modelling tool elaborated by Natural Capital Project and used worldwide from large to local level able to valuated a considerably number of ES simultaneously.

Recently the model has been used and tested on future-oriented land use change (Nelson et al., 2009) to address policy actions(Lawler at al., 2014) and devise payment schemes (Daily et al., 2009), management actions (Goldstein et al., 2012) and future initiatives (Guerry et al., 2012). Globally, InVEST is one of the most used tools to assess multiple ES and has been used on a wide range of landscape pattern and ecological question such as hydrological service in Spain (Terrado et al., 2014), pollination service in Czech Republic (Zulian et al., 2013), to infer land degradation in Ethiopian Great Rift Valley (Cerretelli et al., 2018) and biodiversity in Costa Rica (Vallet et al., 2016). The wide range of uses of InVEST tool is due to its simplified approach however, some of the most used terrestrial modules such as Sediment and Nitrogen retention also have some limitation that might affect the results.These processesare particularly sensitive to the land use considered and substantial error could be introduced when using global land use/cover maps that do not resolve local details (Cerretelli et al., 2018), therefore high detailed dataset and good quantity of time dedicated in parametrizing models and verify its sensitivity, are crucial in defining strategies for ESs. Nevertheless, modules like Water Yield, that form the basis of the Sediment and Nutrient retention models, can produce accurate estimates of water yield in the UK river catchments (Redhead et al., 2016) and the values used are transferrable to other UK catchments.

2.4.4 Circuitscape

Forest commodities on an international market are predicted to be under pressure in the future due to the increasing global demand and economic globalization, therefore habitat diversity and network linked to these global drivers seems to follow the same trend (Nabuurs et al., 2007). It is highly recommended to avoid habitat destruction that can lead to resistance to dispersal of woodland species and prevent the vulnerability of these ecosystems (Wright et al., 2006; (Merriam, 1984; Sutcliffe and Thomas, 1996; Bruinderink et al., 2003). It is therefore crucial to expand and maintain a good woodland network given the future climate and land use projections.

A fragmented landscape, trims the movement and dispersal capacity cascading affects species resilience and persistence (Etienne et al., 2003). However, if ecological corridors have the potential to reduce isolation and fragmentation between habitat (Vermeulen, 1994; Charrier et al., 1997; Clergeau and Burel, 1997) increasing biodiversity, conversely, the forest system can trade-off with other service, facilitating the spread of pathogenic tree disease leading to change in forest composition (Brown and Webber, 2008), or a corridor for one species may act as a barrier for another (Fjellstad, 1998; Mauremooto et al., 1995). A recent study in UK have demonstrated how important to integrate and consider tree disease management with the ecological requirements of red squirrel suggesting that in Sitka spruce dominated forests, a presence of approximately 20% of other conifers, such as pine species or larch, is recommended to ensure a more dependable seed food supply (Shuttleworth et al., 2012). Thus, landscape connectivity affects various ecological process and modelling became a regular method to measure its role. Simple approach such as the "least-cost" pathway for years was the basis of minimum cumulative resistance model applied to ecological species and habitat studies to simulate the efficiency of the landscape as a function of the distance travelled and the costs traversed (Stevenson-Holt eta al., 2014; Watts et al., 2010; Pliscoff et al., 2020; Marrotte et al., 2017). It is nonetheless that scholars have found a new efficient approach based on "circuit theory" to define functional connectivity and the impact of land use change establishing new ecological corridors. This new method has been recognised to offer multiple advantages in: i) accounting for multiple dispersal pathways; ii) showing the degree of redundancy; iii) predicting movement patterns and fates of random walkers in complex landscapes (McRae et al., 2008; Gimona et al., 2012).

2.5 Integrated spatially explicit models

Although the EU Biodiversity Strategy 2020 requires from the Member States an assessment of the ESs (EC, 2011; Maes et al., 2013, 2014), a standardized approach still seems to be lacking for woodland expansion assessment. As long as land management increases in public attention, its importance, complexity, and request of transparency from decision processes is needed. Within the consultation in the informative process the demand to integrate very heterogeneous data arised. For instance, using ES approach and the multiple dimension of them may lead to the integration of bio-physical measures with subjective opinions, rather than GIS datasets and model simulation. This integration in the context of woodland management, has been used by multiple studies (Yousefpour and Hanewinkel, 2009; Cademus et al., 2014; Temperli et al., 2012; Kašpar et al., 2015). Methods have been proposed for modelling potential timber production (Allison et al., 1994; Bateman and Lovett, 1998) for recreational activities (Bateman et al., 1999; Tenerelli et al., 2016) and carbon sequestration (Seidl et al., 2007, Bottalico et al., 2016) integrating drivers of land use and climate changes. Although an holistic approach has led to great discoveries and remarkable progression in the integration of natural and human component to create sustainable solutions, further effort is still needed to blend and consider simultaneously the spatial dimension and human perceptions of the same ecosystem.

The spatial dimension of ES is a key issue for stakeholders since they are more interested to know ''where'' to implement planning than ''why''. Usually, they have clear ideas of local and regional problems, but they need operational and spatial solutions (Fürst et al., 2014). Therefore, integrated methodologies and tools—such as Multicriteria Decision Analysis (MCDA) method provide unique and useful solutions in facilitating the work of decision makers. Exploring the balance between the pros and cons can be tricky for the quantity of information to consider. Here, GIS and MCDA provide the efficiency to visualize multiple information illustrating the per-formance of alternatives across criteria, exploring trade-offs, formulating a decision and testing its robustness. GIS-based and MCDA examples have been widely applied in the definition of the land suitability for recreation (Miller et al., 1998), for animal and plant species (Store and Kangas, 2001), geological risk (Pradhan and Lee, 2010), zoning for conservation (Genetti et al., 2008), urban planning (Dai et al., 2001), renewable energy assessment (Polatidis eta al., 2006), and regional planning (Bailey at al., 2006).

2.6 Summary

Land management is defined by the presence of human activities, that affects the biotic and abiotic component of the landscape (Van Oudenhoven et al., 2012). The introduction of the ecosystem service concept in land management has shown that these elements are not separate entities, but rather interlocked components of a shared structure. Therefore, it is important that managers and policy makers recognise this complexity and make an effort to manage an area so that ecological services and biological resources are conserved, while sustaining human use, in other words, try to effective deliver a multifunctional forest. There is indeed proof of substantial ecological, social and economic benefits when the research of multifunctionality is prioritised in the forest planning process and when the sustainable use of ecosystems can generate a “win-win situation” (De Groot et al, 2012).

The review of multifunctional concept has led to the identification of appropriate set of indicators (criteria) for planning (Bibby, 1998) that are crucial in the management of conservation areas. Particularly, in forest context this selection exercise have to narrow criteria at landscape scale, and prioritize the part of landscape that has potential for afforestation according with existing understory. definition and criteria for the concept of multifunctional forest management in a modern context.

Furthermore, the definition of key criteria to outline multifunctionality, tools, methods and participants are considered equally important and should receive similar attention, especially when land suitability is the final outcome and the level of complexity between function s increases. (Fig. 2.2).

Finally, it can be highlighted that, in many cases, forest managements which accentuate the economic return such as the intensification of productivity for timber production can have a negative effect on other services. Bennet et al., 2009 have highlighted the difficulties to

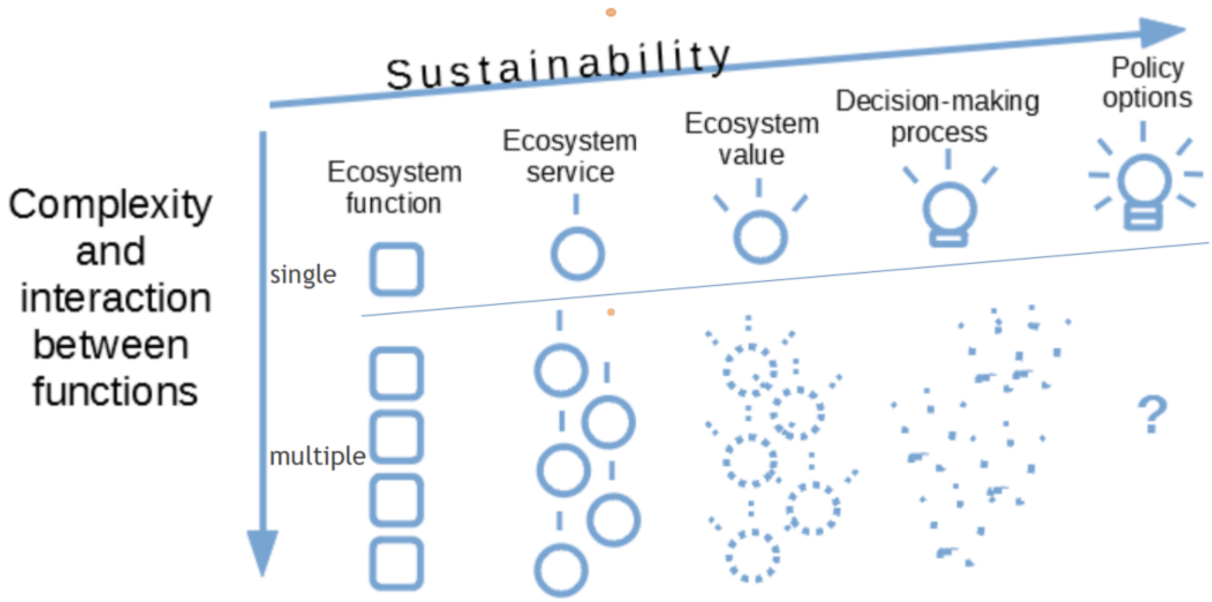


Figure 2.2. Scheme to highlight the difficulty to define a clear list of policy options when interaction between ecosystem complexity (number of functions) and sustainable decision making process increases.

simultaneously produce multiple, positive ecosystem services by the forest, due to the trade-offs among different, or even competing functions; optimisation of one service may cause substantial declines in other services. There is therefore a need to develop and test approaches to the quantification of realistic options between multiple forest ecosystem services.

Chapter 3 – Study area, drivers and barriers of change 3

3.1 The Cairngorms National Park

Cairn Gorm, meaning Blue or Green Hill in Gaelic, is the sixth highest mountain in Britain, and gives its name to the whole surrounding area above 600m in altitude located in the northeast of Scotland, despite neither the highest nor the most prominent mountain in the range. Cairngorms National Park (CNP) was established in 2003, at the time already the UK's largest National Park and expanded into Perth & Kinross council in 2010. The CNP (Fig 3.1), previously 3,800 sq km now covers 4,528 sq km and is twice the size of the Lake District National Park and the Loch Lomond and the Trossachs National Park. It is now home to over 18,000 people. Here tourism makes up abut 80% of the economy with 9 millions visitors recorded in 2018 (ST, 2019). However, the top 5 positions in distinctive industries list are mostly occupied by forestry (CNP, 2013).

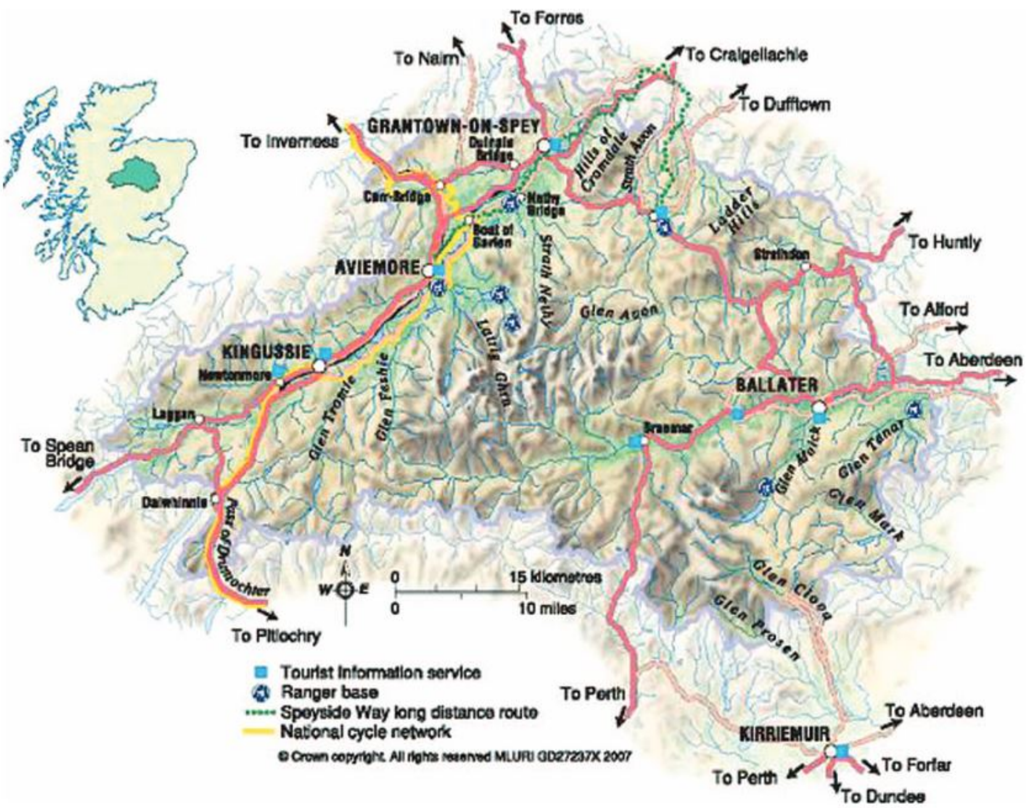


Figure 3.1. Map of the Cairngorms National Park (Dinnie et al., 2012)

The reason of this attraction is due to its ionic landscape and habitat richness, where forest occupy a big portion. Existing woodlands are in fact home to the last remaining stronghold of the Capercaillie (*Tetrao urogallus*) and an incredible diversity of other flora and fauna. Within the Cairngorms parkland there is ample space to create more forests and woodlands enhancing the habitat network. This newly forest could therefore have substantial impact on the recreation, water quality, natural flood management functions while increasing the strength of the forestry industry.

3.2 Bio-physical characteristics

The park has a large highland plateau which is separated by the surrounding uplands by the valleys of the Dee and the Spey (Brown and Clapperton, 2002) and is categorised as a "protected landscape" with development limited within its borders. Its altitude varies from the top of Ben Macdui (1309m) to the lowest point along the Dee river (134m) in the proximity of Dinnet.

The plateau mentioned above was formed by a granite pluton intruded during the Devonian geological period (425Myr) into Caledonian metamorphosed sedimentary rocks which form the generally lower in altitude areas of the park. Glacial erosion in the Cairngorms has been confined mainly to the deepening and extension of pre-glacial valleys and the formation of corries in valley heads. The history of glacier ice cover in the Cairngorms suggests that three contrasting relief-forming environments operated in the recent geological past and have mainly contributed to define the landforms currently present in the areas (Hall et al., 2013) while the progressive erosion along structural lineaments (lines of weakness in the underlying geology such as faults, joints and fold axes) produces troughs incised into the landscape now recognised in the valleys (Brice et al., 1998).

The resulting shape is a landscape dominated in the majority by large river valley bordered by gentle hill slopes accidentally featured by sharper landforms. Soil formation and properties are the direct result of superficial deposits and the underlying geology which can be divided to three main zones. Podzolic soils are well represented (50%) in the CNP and are mostly confined to the lowlands, succeeded by subalpine soils (18%) above 550 metres altitude and peat soils where conditions are wetter and poorly-drained (13%) while mineral soils are scattered throughout the park (4%). Eight Sites of Special Scientific Interest (SSSI) in the Park are considered to have soils of international importance, and 12 have soils of national importance (CNPA, 2006c).

Summers are mostly cool within the Cairngorms area, with an average range of temperature between 8.1 and 15°C recorded in the period of 1980-1998. In winter, average mean monthly temperatures can vary from 3.7 to 2.5°C. Frosting event are common. Mean Annual precipitation ranges from up to 2250 mm within the Cairngorms to as much as 900 mm within the northern valley. Average annual number of days with lying snow varies ~60 days in the lowlands to 200 days in the summits. Prevailing winds direction in the Cairngorms are from south-west (CNPA, 2006a).

In this particular physical geographic context the flora and the fauna of the Cairngorms developed in accordance with the complexity of the landscape. This natural zonation mostly due to the altitude, is characterized by low nutrient tolerant plant communities and a diverse quantity of habitats as described in the Cairngorms Biodiversity Action Plan (CP 2002), which groups the habitats into four classes: farmland and grassland, montane, heath and bogland, wetland and water, and woodland.

The Cairngorms Massif offers extensive areas of artic-boreal healths, sedge and rush heaths, and moss heaths. This montane habitats and the passage with the sub-montane zone is often blurred and identified as a transition zone. Widespread artic alpine species here are the typical dominating communities. Where soils become more acidic the habitat migrates to acid grassland communities where the form of the vegetation is dominated by grasses and herbs on a range of lime-deficient soils. Such soils, usually have low base status, with a pH of less than 5.5. This habitat type includes a range of types from open communities in the lowlands, through closed pastures, to damp acidic grasslands on gleys and shallow peat (Jackson,2000). While a number of rare species grow on the sharp spurs of the cliffs (e.g. alpine saxifrage, Highland saxifrage), in the glen underneath a usually rich patches of forest (Tab 1) have developed over 16.5% of the total Cairngorms area. Within the park 81% of the area of tree cover is coniferous with three quarters of that being native Scots pine (60%). By far the most dominant broadleaf species is birch (16%) with other species, eg rowan and aspen making up only 3% of the tree cover.

	Total area (ha)	Total area (%)	Total volume (K m^3)
Scots pine	36,900	60	7,204
Sitka spruce	5,600	9	1,843
Lodgepole pine	300	5	743
Larches	2,600	4	64
Other conifers	1,600	3	552
All conifers	**49,800**	**81**	**11,040**
Birch	10,200	16	858
Other broadleaves	1,900	3	220
All broadleaves	**12,100**	**19**	**1,082**
All species	**62,300**	**100**	**12,126**

Table 3.1. Estimation of forest cover (area and volume) in the Cairngorms National Park (National Forest Inventory, 2015)

CNP's forests are the remarkable highlight of the scenic landscape. The distinctiveness of these forests is defined by their exceptional height and the high proportion of native tree species they contain (commercial forest are predominantly Scots pine). These forests represents the core of the native pinewoods and are of great antiquity. Roughly 13,258 hectares are documented by the Forestry Commission's Caledonian Pinewood Inventory and these sites are considered the residual part of the original Caledonian pine forest, a community of ancient Scots pine, birch and juniper. Despite their mostly high-altitude distribution, the Native pine woodlands often are located on lithologies with strongly leached podzolic soils and support a very highly specialized selection of plants and animals (with 16-20 breeding bird species) rather than a highly distinctive species abundance (Lust at al., 2001). Finally Summers et al., 1995 state that no mammal species were associated with Scots pine in Britain. However, Mason 2000 claim that patches of planted Scots pine of ca. 500 trees/ha, can facilitate free movement of red the squirrel.

Oak and Birch are, on the other hand, the dominant type of broadleaved woodland in the Cairngorms but still quite uncommon. If most of the pine woodlands are located along the glen of Spey (north), the majority of deciduous trees are found in the south and east of the CNP, along the Deeside, and in the Angus Glens. Most broadleaved woodlands are highly dominated by downy and silver birch and are supported by wet soils. The high quantity of deadwood and the fragmentation and the configuration of the landscape (mosaic of woodland and open space) provide the general condition to sustain the appropriate habitats. The number of specialized flora and fauna species associated with birch is higher than for other tree species in Europe (Branquart et al., 2005), while oak has been associated with 2300 different species (Mitchell et al., 2019).

Plantation in the CNP also has the potential to maintain some wildlife but are subject to silvicultural system whose prevailing management approach is clearfelling with artificial regeneration and a rotation age of ~60–80 years (Mason et al., 2007).

3.3 Ownership and landholding

There are over 150 different land-holdings in the ranging in size from less than 100 hectares to over 40,000 hectares and up to 75% of these are private ownership while the remaining land is divided between Charitable conservation bodies, such as Mar Lodge Estate in Deeside and Abernethy Estate in Strathspey, and Government Agencies, as The Forestry Commission, Scottish Natural Heritage and Highlands & Islands Enterprise. (CNPA, Estate Management). Since the Neolithic period, Cairngorms area was deeply modified by modern introduction of pastoralism and the development of larger the settlements such as Grantown-on-Spey and Ballater. Today the main land use is sheep farming and, to a lesser extent, cattle farming. Only since the 60's coniferous forestry become important (MacMorran, 2008).

The Scottish debate around private land and how this has been managed, became quite intense and controversial in the last years, with various attempts by private shooting estates that, by managing their game for sustainable harvests, to claim that they are practicing conservation of a sort (Adams, 2012). Conflict emerged after several organisations pushed Cairngorms National Parks Authority to address the problems associated with grouse moors: destruction of habitats, destruction of the landscape, destruction of wildlife and destruction of the rural population. (Kempe, 2016). Gamekeepers burn patches of heather in rotation to provide a mixture of areas, with young shoots suitable as food for red grouse and older heather that provides cover, they also kill predators of red grouse and their eggs, while birds of prey are illegally shot, trapped or poisoned on many grouse moors (Etheridge, Summers & Green 1997; Potts 1998; Green & Etheridge 1999). Moreover, on occasions when CNPA have tried in the past to stand up against the “self-management” of the landscape as a means to be used to support field sports, private estates have held a strict position, and the resulting controversy needed to be solved by the intervention of the Scottish Government (SG, 2020). In this crossfire between conservationist and landowner the real loser seems to be the public domain that now needs to pay through incentives schemes to restore wildlife. In this sense, the citizen who has seen tax rises over the last 10 years might ask “for whom incentives and for what reason”. However, if the fact that half of Scotland is owned by just 500 people has been revealed (Wighman, 2018) the names on the individual pieces of the mosaic are still missing and the question: “who own Scotland?” remains unanswered.

Wighman, 2013 shows that 750,000 acres of Scotland is registered in tax havens posing problems for law enforcement and tax authorities. However, public representative like Minister for Energy, Enterprise and Tourism rejected proposals to reveal the beneficial owners of companies that own land (Scottish Parliament, 2012) - although the Government were eventually persuaded to do so in the Land Reform (Scotland) Act 2016 (Combe, 2016) – it is easy to understand the seriousness of I think strategy the discussion around ownership.

3.4 National policy context and spatial planning in Scottish forestry.

There is a notable number of strategy and policy documents of national and devolved governments in the UK that strengthens the forestry benefits (Scottish Executive, 2006, Scottish Government, 2009, Department for Environment Food and Rural Affairs, 2013; Department of Agriculture, Food and the Marine (DAFM), 2014). Additionally, the Scottish Biodiversity Strategy and the Scottish Forestry Strategy (Forestry Commission Scotland, 2006, McIntosh, 2006) have narrowed the path to ‘progressive forward-looking strategies’ for woodland restoration (Hobb, 2009), while the Land Use Strategy engages the ecosystem approach and aims at adaptation to climate change. The north-east of Scotland case study from Muñoz et al., 2015 summarized very well the synergies and conflicts between the woodland expansion plans and the numerous attempts to deliver a woodland multifunctionality. While this multifunctionality remains the main challenge of the national agenda a very strong alignment of the national planning context with the climate change target exist. Muñoz et al., 2015 analyse the way that the forest development entrusts the management to land managers (private and public) while coordination and monitoring is carried by planning authorities, and argue that coordination between policies and planning is not sufficient. Implementation of land use changes largely relies on the intentions of farmers and land owners to convert their land into a woodland and agree with the policy recommendations. Spatial explicit approaches have been applied as an attempt to deliver a more functional woodland in Scotland through project as 'the right trees in the right places' (Forestry Commission Scotland, 2010a) and 'The forest of Cairngorms' (CNPA, 2008) within the CNP. Despite these attempts, Muñoz et al., 2015 observed that in the recent years forest cover declined while (Scottish Government, 2013) the current planting rates are insufficient to meet the goal-oriented aspirations for forest expansion. Therefore, in a context where changes in land use systems, and so the delivery of the public good, are resolved by who owns and manages the land, technical guidance for planning needs to clear up inconsistencies between policies and planning actors and instruments. This paradigm of land use change is strongly connected with the presence of economic, social, physical-environmental and operational factors that influence woodland creation on private land (Thomas et al,. 2015) while woodland targets encouraged by various grant schemes, with over £0.5 billion paid in grants from 2005–6 to 2014–15 within England, Scotland and Wales (Forestry Commission, 2015), are still underachieved due to insufficient incentives (Burton, 2004). Moreover, afforestation on farms is framed as balance to the ongoing productivist orientation of farmers: most farmers prefer not to afforest, and if they do afforest, prefer to do so on poorer quality land (Hopkins et al., 2017).

In 2018 the Cairngorms National Park have released the its own Forestry Strategy (CNPA, 2018) in which the aspiration to enhance habitats on a landscape scale is strengthened addressing climate change, timber, access and health, environmental quality, business development, community development, and biodiversity. The guideline is driven also by key policy documents like Cairngorms Nature Action Plan (CNPA, 2019a), Active Cairngorms (CNPA, 2015), the Cairngorms Economic Strategy (CNPA, 2019b), Local Development Strategy and the Local Development Plan and aims to strengthen a forest habitat network between Spey, Dee and Tay river catchments in order to enhance migration and colonization of woodland species and, deliver multiple benefits. Despite some skepticism about planning for woodland expansion at national level (Sutherland et al., 2006), the Cairngorms Forestry Strategy 2018 seems to have searched for a solution that considers space, the biophysical and built components of the environment. However, such frameworks risk becoming ineffective in the resolution of trade-offs because they are based on data that do not totally consider multifunctionality (Native Woodland Model, 2004). Therefore, to successfully implement such an ambitious plan, spatially explicit, integrated approaches are needed. Some examples of good practice which adopted the Ecosystem approach and operating to local level (Baggio Compagnucci et al., 2015, Castellazzi et al., 2016, Gimona et al., 2016) have been considered rewarding by the policy actors.

In the following chapter the dynamics of the potential carbon sequestration by the trees in Scotland is discussed throughout a paper which is currently under submission. The double intention is to use the carbon sequestration service as the overarching component of the multi-criteria analysis applied to the Cairngorms National Park where soil and trees are treated as part of the same system and incorporating the attempt to resolve the debate of how much trees can contribute to reach net-zero emission national target into the local objectives.

Chapter 4 – How will woodland expansion help Scotland meet its climate goals? The potential for carbon gains and losses.

(Paper under submission)

Baggio Compagnucci A.[ab], Aitkenhead M.[b], Hewitt R.J.[bc], Ovando P.[b], Canullo R[a]. Gimona A.[b]

[a] *The James Hutton Institute, Craigiebuckler, AB158QH Aberdeen, Scotland, UK*

[b] *School of Biosciences and Veterinary Medicine, Plant Diversity and Ecosystems Management Unit, University of Camerino, 62032 Camerino, MC, Italy*

[c] *Transport, Infrastructure, and Territory Research Group (tGIS), Geography Department, Faculty of Geography and History, Universidad Complutense de Madrid (UCM), Madrid, Spain*

4

Abstract

The reduction of GHG emissions is a key policy commitment of the countries that signed the 2015 Paris Agreement. In the land sector, woodland expansion is often considered an important means to contribute to this commitment, through offsetting emissions. In response, the UK and the Scottish Governments have initiated a substantial programme of woodland expansion. To assess the potential contribution of woodlands in Scotland we estimated the potential for net carbon storage in the landscape, accounting for the interactions between climate, soil-related factors and planting practices. We present detailed spatial results for where in Scotland woodland expansion would be likely to result in overall carbon gains, accounting for the spatial variability of timber yield classes (YC) as well as present and future climate (RCP 8.5). The results provide a precautionary lower limit for the net carbon storage expected, that may help to minimise the risk to afforest in unsuitable locations. We show that, while there is a large amount of land that can potentially achieve net carbon storage through afforestation, especially in the lowlands, this often does not apply to the uplands. Soil type and the intensity of soil-disturbing resulted crucial in determining whether the modelled net carbon storage over the next decades is likely to be positive. Upland ecosystems, whose soils are rich in carbon, resulted vulnerable to loss, particularly with intensive commercial planting practices. While the prevalence of mineral soils in the lowlands makes them a safer option, it also exposes these areas to potential conflicts with agricultural activities. Compared to the global UK carbon footprint, the magnitude of the offset obtained in 30 years – if afforestation goals are reached – is likely to be lower than 1% for the UK and around 12% of the total Scottish footprint. While this is valuable, and can provide other multiple benefits, it reinforces the need to pursue a systemic approach that seeks reductions in emissions in a multitude of ways throughout all sectors.

4.1 Introduction

The reduction of GHG emissions is a key policy commitment of the signatories of the Paris Agreement (UNFCC, 2015). This entails reductions in all sectors of the economy through Nationally Determined Contributions (NDCs, e.g. Pauw and Klein, 2020). According to The IPCC Climate Change and Land Report (IPCC, 2019) to limit the global temperature increase to 1.5°C, the land-use sector needs to contribute to decarbonising the economy and offsetting remaining emissions.

Forests worldwide have the potential to play an important role in the mitigation of climate change by acting as a sink that can offset some of the human GHG emissions (e.g. Andregg et al., 2020; Bastin et al., 2019). Forest planting is therefore encouraged through numerous initiatives, such as the Trillion Trees Initiative, a joint venture of BirdLife International, Wildlife Conservation Society and World Wide Fund for Nature. However, trees are not necessarily a silver bullet (e.g. Holl and Brancalion, 2020) and can sometimes cause large trade-offs, e.g. through GHG emissions from ground preparation, or impacts on carbon sinks (Seymour, 2020). To reach the intended mitigation effect, it is therefore important that implementation plans take into account local ecological attributes, to ensure that the right tree is planted in the right place (e.g. Fady et al., 2021) and undesirable trade-offs of afforestation are minimised.

To help meet the international commitments in the UK as a whole, and in Scotland in particular, the land use sector's climate policy comprises ambitions to expand woodlands from 18% to 21% of Scotland's total land area by 2032, with 15,000 hectares of newly planted forest per year from 2024/25 (Scottish Government, 2018).

In planning such expansion soil attributes are of particular relevance, as the dynamics of soil carbon are crucial for carbon storage (e.g. Hofmockel et al., 2011). It is known that the total amount of carbon stored in soils is far higher than the amount stored in vegetation (Tarnocai et al., 2009), and especially so in boreal ecosystems (e.g. Simola et al., 2012; Köchy, Hiederer, & Freibauer, 2015), where plant growth and decomposition are often restricted by nutrient limitation and climate. Consequently, it is important to minimise the release of soil carbon by new planting as this reduces the net carbon captured and can undermine the effectiveness of new forests as carbon sinks.

Recent evidence from Scotland (Matthews et al., 2020; Frieggens et al. 2020) has also confirmed that soil attributes are a key local variable in determining whether new woodlands are a net sink or a net source of ecosystem carbon. The reason is that on organic or organo-mineral soils, tree planting, especially with high soil-disturbing ground preparation methods (e.g. deep ploughing and turfing) applied to large planted areas, can release more carbon than the trees are likely to accumulate in the next 20+ years. Analysing a combination of species, their findings showed that most of Scotland's upland soils would be at risk of releasing carbon if planted with commonly used mechanised methods (see also e.g. Vanguelova et al., 2018). This has significant implications for the amount of land available for tree planting in Scotland and for the potential conflicts with other land uses. In particular, if a large portion of the uplands were not suitable for mitigation, the potential conflicts would mainly affect agricultural land in the lowlands, with higher opportunity cost for agricultural uses.

While the Matthews et al., 2020 study makes an important advance in our understanding of the role of afforestation on forest carbon potential, some key questions remain unanswered, which the present paper aims to address. Matthews et al., 2020 assumed fixed "representative" timber yield classes (rather low ones) for a number of species and limited their analysis to the present climate and they also used coarser resolution soil carbon data than in the present study. By refining the spatial and temporal dimension of the analysis, it becomes possible to highlight important heterogeneities which may not have been detectable in the earlier study. If such spatial or temporal heterogeneities were to exist, some areas previously deemed (in)appropriate for carbon storage might be seen to be (un)suitable either now, or in the future, once climate change effects are taken into account.

To address these research gaps, our study seeks to answer two key research questions: (i) where, in Scotland, would tree planting be likely to result in overall carbon gain, accounting for the spatial variability of timber yield classes? And (ii) whether climate change would alter the results we obtained for the present.

We believe that our findings have important implications, not just for Scotland, but for boreal ecosystems and carbon rich soil sinks across the globe.

4.2 Methods

To address these research questions, we built statistical models (Figure 4.1) using terrain, climate and soil variables to predict the yield class (YC) for eleven commonly used tree species (Table 4.1). YC is an index used in the UK of the potential productivity of even-aged stands of trees. It is based on the maximum mean annual increment of cumulative timber volume achieved by a given tree species growing on a given site and managed according to a standard management prescription (Matthews et al., 2016).

Each of the eleven models predicts YC at each location in Scotland, with a spatial resolution of 250x250 m. Predictions were for the present and for one climate change scenario (RCP8.5), a widely used emissions trajectory which assumes no mitigation (Ebi et al., 2014) using year 2050 and 2070 for which WorldClim 2.1 data were available, globally, it is also the trajectory followed so far (Schwalm, et al., 2020).

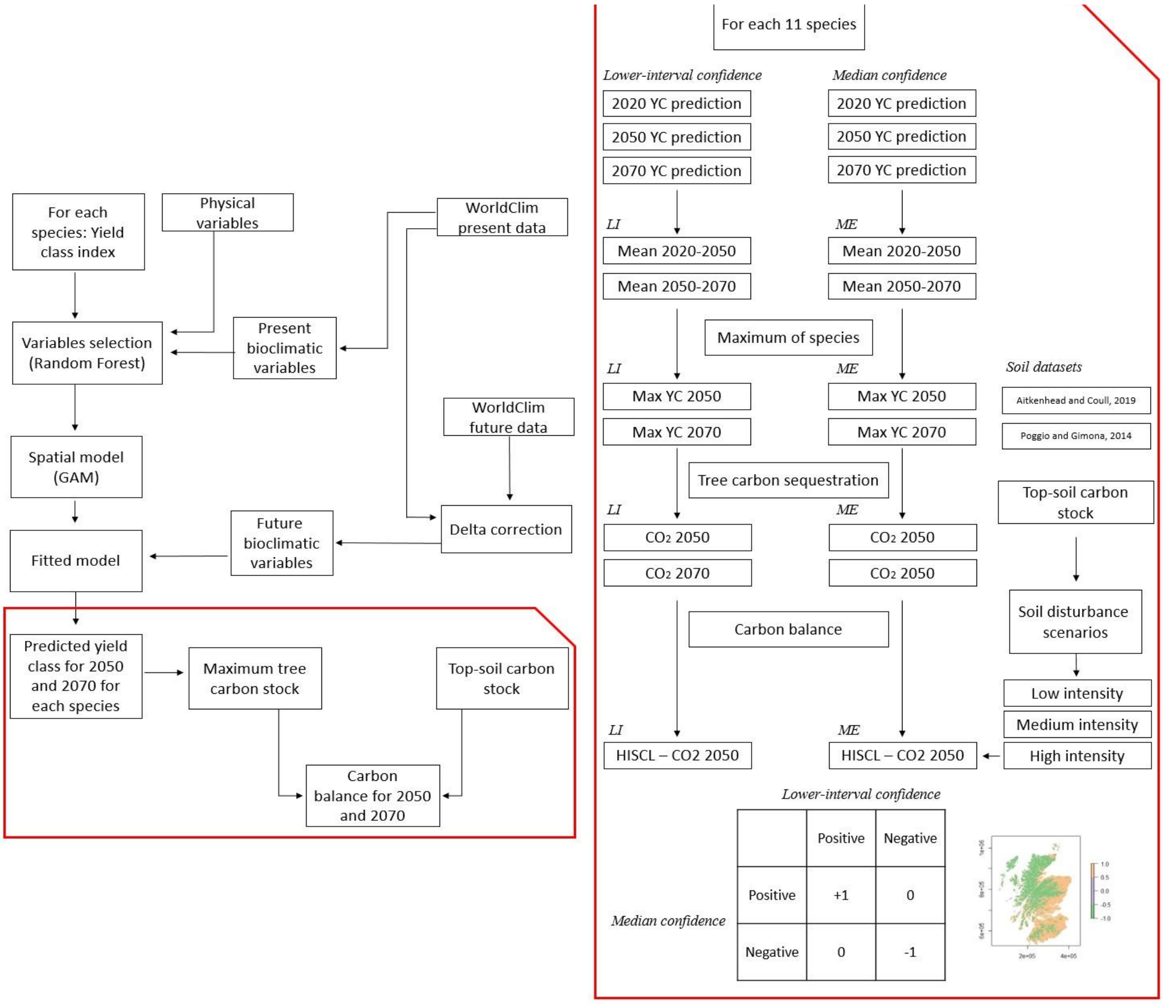


Fig. 4.1. Flowchart of the main steps of the proposed approach.

The predicted YC was then used to help define the quantity of carbon the growing tree biomass can store at the two-time steps. The total carbon (aboveground and root biomass) stock capacity, obtained through conversion factors, was then balanced with existing top-soil carbon content data to calculate the carbon budget for three scenarios of ground preparation intensity. The modelling was performed using R and GRASS-GIS software (R Core team 2020, Neteler et al 2012).

4.2.1 Estimation of the Yield Class

We focussed on the species in Table 1 as these are the most commonly planted in Scotland. We used the Ecological Site Classification Decision Support System (ESC-DSS), accessible on-line at the Forest Research website (http://www.forestdss.org.uk/geoforestdss/) to sample Forest yield class at 618 locations distributed in Scotland.

To capture a range of climatic conditions that might not be present in Scotland now but could emerge in the future, the sample was extended to the rest of UK with additional 353 sample location using a stratified random sampling strategy (Fig. 4.2) to span the environmental conditions found in the whole UK.

Common Name	Species code(1)	Growth model code(2)	Latin name	Native
Ash	[AH]	SAB	*Fraxinus excelsior*	Yes
Common alder	CAR	SAB	*Alnus glutinosa*	Yes
Douglas fir	DF	DF	*Pseudotsuga menziesii*	No
Downy birch	PBI	SAB	*Betula pubescens*	Yes
Lodgepole pine	[LP]	LP	*Pinus contorta*	No
Pedunculate oak	POK	OK	*Quercus robur*	Yes
Scots pine	SP	SP	*Pinus sylvestris*	Yes
Sessile oak	SOK	OK	*Quercus petraea*	Yes
Silver birch	SBI	SAB	*Betula pendula*	Yes
Sitka spruce	SS	SS	*Picea sitchensis*	No
Wych elm	WEM	BE	*Ulmus glabra*	Yes

Table 4.1 List of the 11 species obtained from Ecological Site Classification and their Latin names. In brackets species with pest or disease constraints in the UK. 1 In brackets species with pest or disease constraints in the UK.

2 Growth model code of the species representing timber volume growth according to the Forest Yield Model (FYM) (Matthews et al, 2016, see section 2.3), where SAB species include Sycamore-Ash-Birch, and common Alder, and BE represents Beech which is the FYM growth pattern used for Wych elm.

4.2.2 Estimation of independent variables

4.2.2.1 Physical variables

For the same locations, physical variables such as altitude, slope and aspect were determined using the Mapzen Terrain Service digital terrain model, accessed through the R function get_elevation_raster (Hollister, J.W., Tarak Shah, 2017) along with accumulated temperature and an indicator of wind exposure (DAMS score) from Forest Research ESC website.

4.2.2.2 Climate data

Along with the physical variables mentioned above we added climatic indicators as covariates. In particular, we used WorldClim 2.1 (WC21) (Fick and Hijmans, 2017). This is a database of spatially interpolated monthly climate data for global land areas at a very high spatial resolution (30 arc-second in the WGS84 reference system - approximately 1 km^2). This dataset also contains 'bioclimatic variables' derived from monthly patterns of temperature and precipitation. Present climate data refer to "9000–60 000 weather stations over a temporal range of 1970–2000" . WorldClim has been widely used in species distribution and ecological modelling (Macek. et al., 2019, Poggio et al., 2018), including forestry (Schueler et al., 2014, Marchi et al., 2019) and erosion (Panagos et al., 2017) and applied to hydrological and carbon assessment studies (Zomer et al., 2006).

For future climate we used the WorldClim 2.1 projections based on the Coupled Model Intercomparison Project 6 (CMIP6) (Eyring et al., 2016) for the time periods 2041-2060 and 2061-2080. The Bioclimatic variables used are listed and described in Table 2.

CODE	DESCRIPTION	UNIT
BIO1	Annual Mean Temperature	°C
BIO2	Mean Diurnal Range (Mean of monthly (max temp - min temp)	°C
BIO3	Isothermality (BIO2/BIO7) (×100)	-
BIO4	Temperature Seasonality (standard deviation ×100)	-
BIO5	Max Temperature of Warmest Month	°C
BIO6	Min Temperature of Coldest Month	°C
BIO7	Temperature Annual Range (BIO5-BIO6)	°C
BIO8	Mean Temperature of Wettest Quarter	°C
BIO9	Mean Temperature of Driest Quarter	°C
BIO10	Mean Temperature of Warmest Quarter	°C
BIO11	Mean Temperature of Coldest Quarter	°C
BIO12	Annual Precipitation	mm
BIO13	Precipitation of Wettest Month	mm
BIO14	Precipitation of Driest Month	mm
BIO15	Precipitation Seasonality (Coefficient of Variation)	-
BIO16	Precipitation of Wettest Quarter	mm
BIO17	Precipitation of Driest Quarter	mm
BIO18	Precipitation of Warmest Quarter	mm
BIO19	Precipitation of Coldest Quarter	mm
AT	Accumulated Temperature	°C
SPR	Standard Percentage Runoff (derived by HOST)	%
DAMS	The probability of damaging winds	-
ELEVATION	Mean elevation value	m
ASPECT	Mean aspect value	radiants
X	Longitude coordinate (British National Grid)	m
Y	Latitude coordinate (British National Grid)	m

Table 4.2. Bioclimatic and biophysical variables used.

The use of RCP8.5 is also a precautionary choice to avoid path-dependent failure in mitigation due to a potentially optimistic choice of emission trajectory, and leads to identification of 'no-regret' planting locations.

Data derived from the UK-Metoffice Global Climate Model (GSM) were not available in WC21, we chose WC21 data derived from the French CNRM-CM6-1 (Voldoire et al., 2019). The latter is a fully coupled atmosphere-ocean general circulation model of the sixth generation jointly developed by Centre National de Recherches Météorologiques (CNRM) and Cerfacs for the sixth phase of the CMIP6.

Future mean values of the 19 variables (Table 4.2) were downloaded and the respective surfaces were created for two time periods: 2041-2060 and 2061-2080. Each of these datasets is a multiple band raster that needs to be split, reprojected and downscaled due to the difference in spatial resolution of future dataset (2.5 arc-minutes) with climate baseline (30 arc-seconds). Several authors (Wang et al., 2016, Navarro-Racines et al., 2020) have already demonstrated that the use of relative changes in climate WorldClim data is possible and represents a rapid method (named the delta method) for increasing GCM resolution to a finer scale, and that this technique is applicable to ecosystem service studies (Poggio et al., 2018) and species distribution analysis (Wan et al., 2016).

Once the phase of data preparation was complete, we sampled the covariate surfaces using the coordinates of the 971 YC points mentioned above (Figure 4.2), located in Scotland, Wales and England. Sampling beyond Scotland aimed to capture the current conditions in Britain based on our assumption that the climate in the future will progressively migrate to the North, with Scotland seeing a climate similar to that historically observed in the south of Britain.

4.2.2.3 Hydrology Of Soil Types (HOST) data

Because trees are sensitive to soil water conditions, we used the Hydrology of Soil Types (HOST) data as indicators of water infiltration capacity (e.g. Boorman et al., 1995). This is a data set based over 24 000 soil profiles. These attributes were determined by means of pedotransfer and expert knowledge and used in the development of HOST. In this study we used the Standard Percentage Runoff (SPR) parameter, which is estimated from the HOST class. This is the percentage of rainfall that contributes to the increase in surface runoff. Point data corresponding to YC samples were extracted from the continuous HOST geographic data base for Scotland (Lilly, 2010), whereas point data for England and Wales were obtained from Cranfield University (LandIS, 2020).

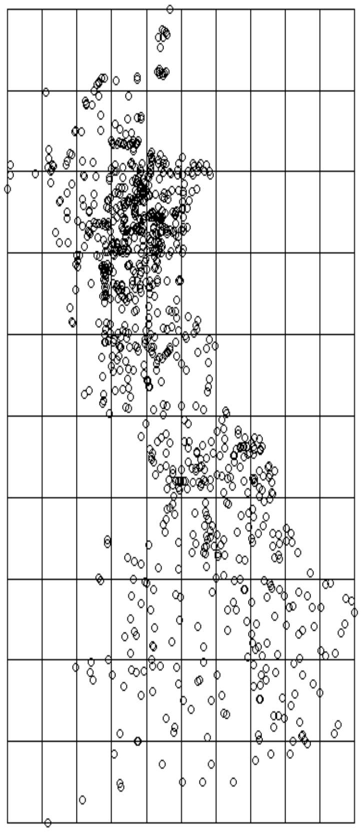


Fig. 4.2 Yield class and covariates sampling locations across the United Kingdom.

4.2.3 Modelling the Yield Class

The data set was split in a training and a validation set (70-30 %). The YC values at each location in Figure 2 were modelled as a function of the independent variables (covariates) described above, and summarised in Table 2.

A hybrid method was used by combining a Machine Learning algorithm, Random Forest (RF) (Breiman, 2001) with Generalized Additive Models (GAM, Wood, 2017).

There are several examples of the spatial use of Generalized Additive Models (GAMs; Wood, 2004; 2017) to investigate linear and non-linear relationships in ecological studies (e.g., Poggio and Gimona, 2014, Poggio and Gimona, 2015, Sinka et al., 2020).

The randomForest R-package (Liaw and Wiener, 2002) (RF) was used to automate model selection by estimating the importance of the covariates, while the GAM was used to estimate the trend of the dependent variable, as a function of the selected covariates. GAMs offer a more flexible approach than ordinary regression methods that allows modelling spatial autocorrelation through smoothing procedures (e.g. Wood, 2006; Poggio and Gimona, 2017). We added a bidimensional smoother of the coordinates for this purpose (see Wood., 2017). The fitted GAMs were validated using the validation set.

The GAM models were then used to predict YC at non-sampled locations for the eleven species in Scotland using the covariates. A set of three time-step predictions (2020, 2050 and 2070) was therefore obtained for each species.

We applied a precautionary approach to minimise the risk of overestimating the tree carbon storage potential (and therefore suitability for tree planting) at each location. We therefore calculated both the lower end of the 95% confidence interval of the estimates and the median. The first (GAMLI), was obtained by subtracting 1.960* standard error from the median. As a reminder, the median (GAMME) is the 50th quantile of the predicted estimate distribution.

4.2.4 Timber growth and tree carbon sequestration models

Each YC is associated with a tree growth curve. Therefore, each cell in our raster of YC predictions is also associated with a growth curve. Having predicted the potential YC volume (m^3 of timber per ha) at each location for all species of interest, we obtained estimates of the time-evolution of tree biomass from the age-related timber production tables of the Forest Yield Model (FYM) (Matthews et al., 2016). We applied the appropriate growth function derived from this source for each type of species and management regime. Here, a non-thinning management regime is assumed. We used the timber yield functions and carbon conversion factors estimated by Ovando (2020, see Appendix F for details) for Sitka spruce, Douglas fir, Scots pine, Sycamore-Ash Birch (SAB), Beech, and Oak species, and applied the same methodology to estimate Lodgepole pine growth and carbon conversion factors (in some cases the same conversion factor is applied to more than one species, see Appendix F for details).

Timber yield is represented by non-linear functions per each type of species that depend on YC and stand age. The timber yield predictions are based on historical observation used to construct FYM timber production tables, and consequently may offer biased estimates of tree growth under changing climatic conditions.

Final estimates for the YC for each species were corrected by adjusting the tree timber growth curves using the mean of the predictions for 2020 and 2050, a process which we describe as follows. Firstly, we assume that any of the future growth predictions based on historical observations are likely to be inaccurate unless climate change is accounted for. Therefore, whenever we predict the YC using bioclimatic variables of the past we need to account for the concurring climate changes. For instance, at one particular location, a predicted YC index of 5 to 2020 for *Pedunculate Oak* might often have a higher (or lower) YC in 2050. Simply using the timber yield curve developed for the period before 2020 (which assumed constant climate) will lead to bias, since the YC curve for a given tree at a particular location is appropriate only for that location at the moment of planting. This assumes constant climate, and needs to be corrected as long as climatic conditions are changing. The adjustment is computed by taking the mean of the two growth predictions, for 2020 and for 2050, obtained applying -respectively- the 2020 and 2050 YC growth curves. Therefore, for each species at each pixel we corrected the growth trend projected for the future using a new growth curve that fits the middle point between the curves (see Fig 6 for an illustration). The same procedure was repeated to correct estimates for 2070. This adjustment technique returned bias-corrected future YC estimates for each species for 2050 and 2070.

It should be noted that most yield tables do not account for trees older than 150 years (and only 80 years in case of Douglas fir), therefore the volume prediction function, and all our analysis, are not affected by this age-range limitation. All the carbon sequestration and stock estimates provided here are for a maximum tree age of 50 years.

To translate timber volume into tree biomass and carbon stock in timber, branchwood and roots, we further use species-specific conversion factors that relate timber volume to total carbon stock in aboveground and root tree biomass (see Appendix F). Woodland soil debris carbon sequestration was estimated considering the Woodland Carbon Code (WCC) look-up tables (West, 2018).

Finally, at each location, we also mapped which species would be needed to obtain the highest tree carbon storage (Appendix A).

4.3 Carbon balance

Net soil carbon gains/losses due to woodland planting and growth need to be carefully considered, in view of potential GHG emissions due to soil disturbance during ground preparation, which are significantly higher when mechanical ground preparation techniques and organo-mineral soils are concerned (West, 2018). We estimated soil carbon stock up to 30 cm depth (topsoil) by averaging, for robustness, two published national datasets (Aitkenhead and Coull, 2019; Poggio and Gimona, 2014).

We then reclassified a Topsoil Soil Organic Carbon map (Lilly et al., 2012) in mineral, organo-mineral and organic soil following suggestion from map authors as mineral less than 12% of carbon content, organo-mineral between 12 and 35%, and organic above 35% (Appendix C). We then applied the percentage of carbon loss due to the disturbance level of ground preparation, as suggested by West, 2018. Initial carbon soil released was elaborated for three ground preparation intensity scenarios (i.e. methods): i) hand turfing and mounding (low), ii) ploughing (shallow turfing) and scarifying, and hand turfing and mounding (medium) and iii) turfing or tine, using double or single throw mouldboard plough (high). Carbon soil loss for each scenario (method) above was then balanced with the gain from tree biomass growth (maximum potential of predicted YC) estimated above, to show where trees would be able to offset the emission due to soil disturbance. For space availability reasons, here we show only the results for 2050 and for the high intensity scenario (HISCL), and a comparison map showing the (dis)agreement between the two sets of GAM predictions namely GAMLI and GAMME. The rest of the results and any related methodological clarification can be found in the supplementary material. This includes (Appendix E) a comparison of afforestation using only commercial species vs only broadleaved native species.

4.4 Results

Figures 3-8 show results for the two example species (*Sitka spruce* and *Pedunculate oak*) of the eleven tree species (details for all other species in Appendix A-D) listed in Table 1, for Scotland. *Sitka spruce* was chosen because it is the highest-yielding commercial species, while *Pedunculate*

Oak has the potential of relatively high carbon stocks among native species. These species were also chosen by the UK Climate Change Committee (2020) to illustrate their afforestation scenarios.

4.4.1 Variables selection

The set of covariates in Figure 3 are the twelve most important variables explaining the distribution of the two illustrative species. While the RF model selected different bioclimatic variables for each of the N species, accumulated temperature (at) and SPR index (HOST classification) were always ranked in the top five for importance.

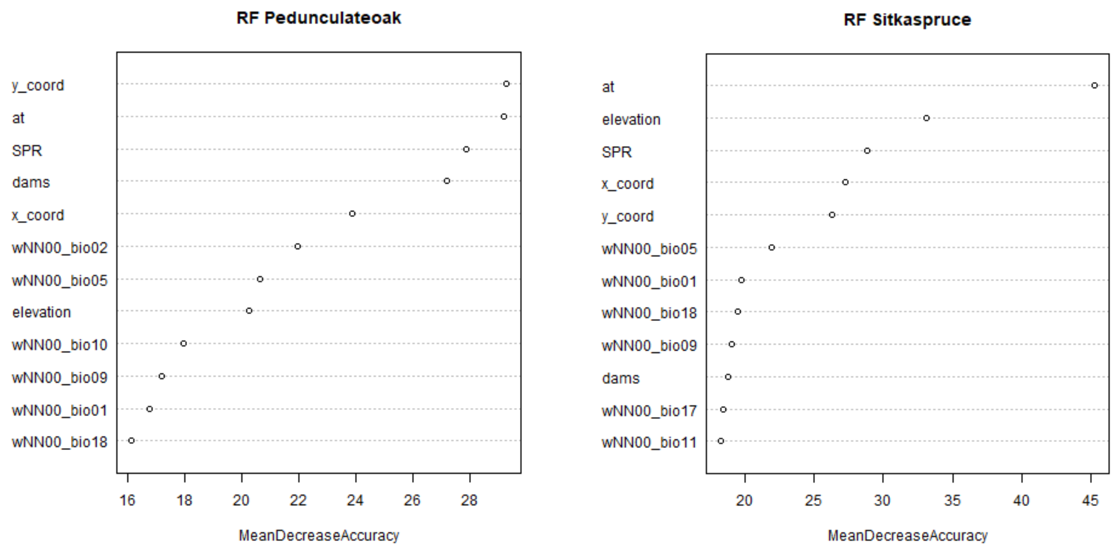


Figure 4.3. Random forest variable importance plots for *Pedunculate Oak* and *Sitka spruce* based on Mean Decrease in Accuracy (MDA).

The cross validation recorded a R^2 of 0.472 and 0.692 and a RMSE of 1.23 and 3.28 for *Pedunculate oak* and *Sitka spruce*, respectively.

4.4.2 Yield Class predictions

The GAM models using the covariates selected by RF had an adjusted R^2 of 0.812 and 0.739, explaining 85% and 77.7% of the deviance, respectively, for *Pedunculate oak* and *Sitka spruce*. For both species, most smoothing functions for the selected model indicate non-linear relationships are appropriate. According to the model response curves YC seems not to be affected by elevation until 550 m, falling off thereafter. The same is true of the dams score covariate which falls off only after a dams score of 18. Model assumptions were met, with normally distributed residuals centred on zero, as shown by diagnostic plots (Figure 4.4) which did not indicate any concerning patterns, except that the highest values of YC might be under-predicted (residuals vs linear predictor plots). This is unlikely to affect our conclusions regarding suitability, as higher yields are known to occur on mineral soils that are weakly affected by carbon release problems.

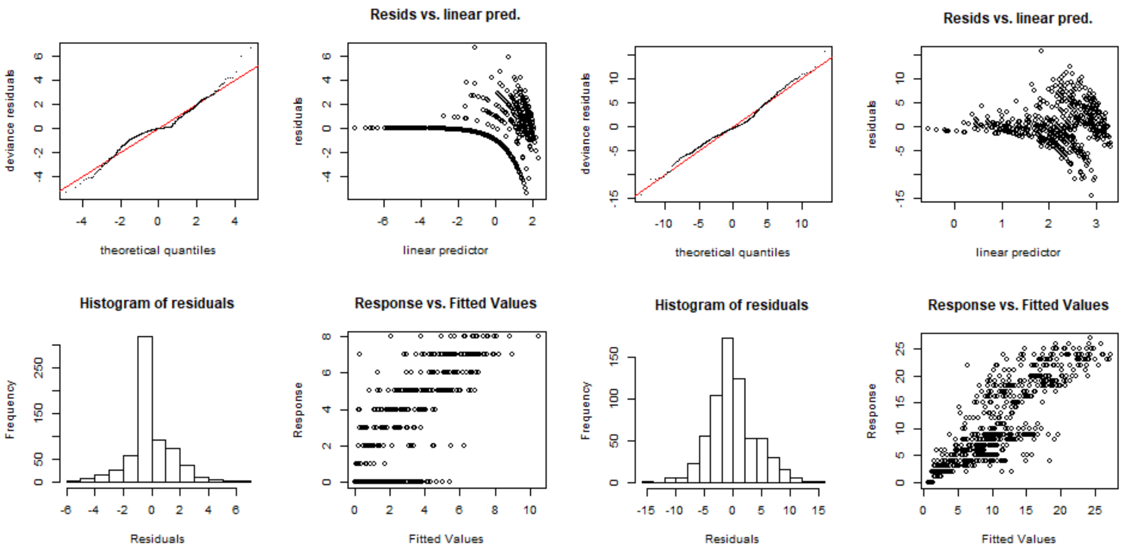


Figure 4.4. Diagnostic plots for the GAM models for*Pedunculate Oak* (left) and *Sitka spruce* (right).

Figure 5 shows the baseline (present) YC in the first map from the left, and the difference from the baseline in the other two maps which refer to 2050 and 2070. *Pedunculate oak* has a generalised positive trend of the YC between 2020 and 2070. The strongest changes characterized by a remarkably higher potential YC - up to 6 classes higher than at present - are located in the south-west and central parts of Scotland. Differences between 2020 and 2070 are less pronounced in the north-east and the western islands. Future climatic trends seem to have no impact on the suitability of *Pedunculate oak* trees in the whole of the north-west and centre as well as parts of the upland areas in the south of Scotland characterized by very low YC potential.

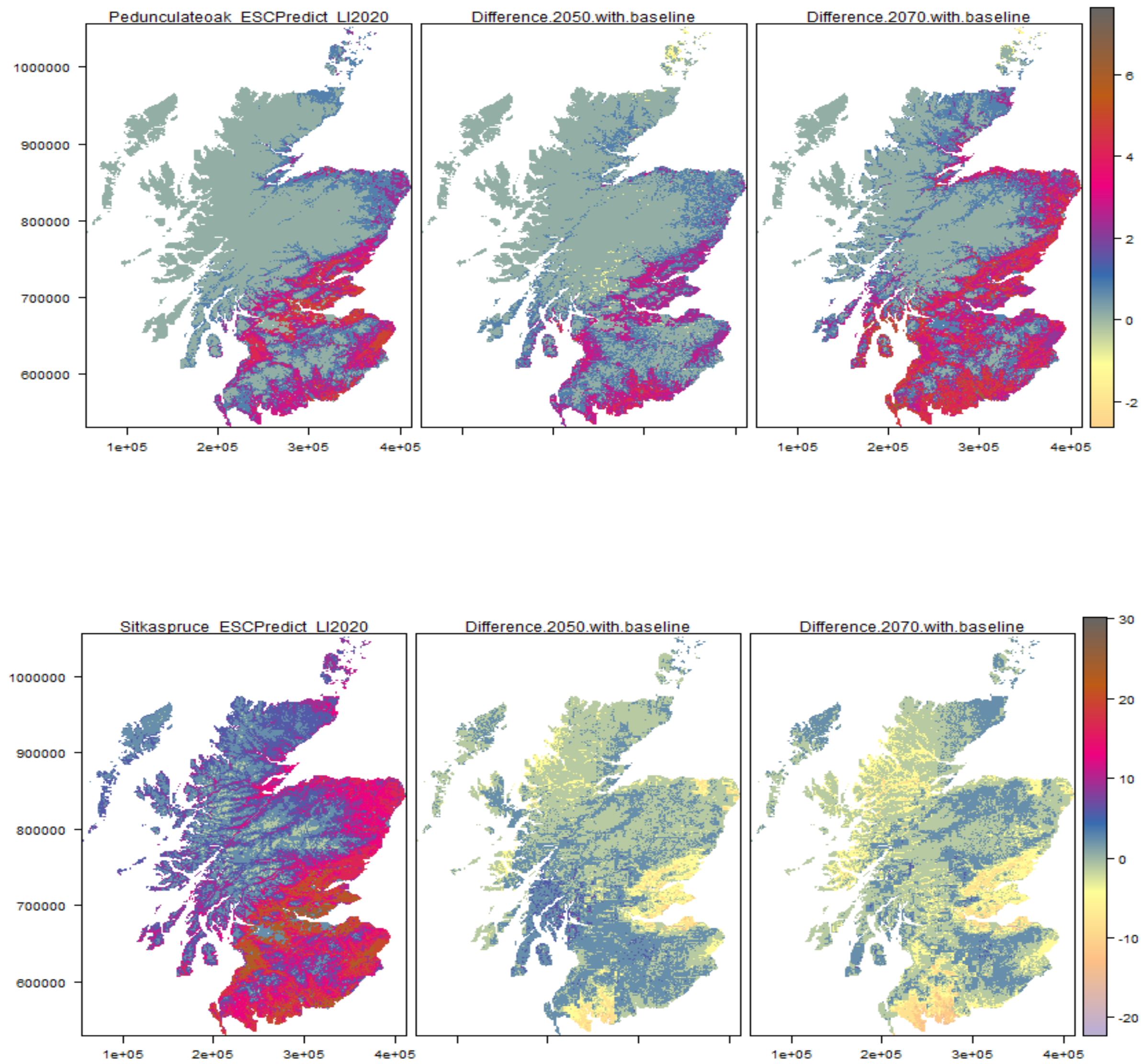
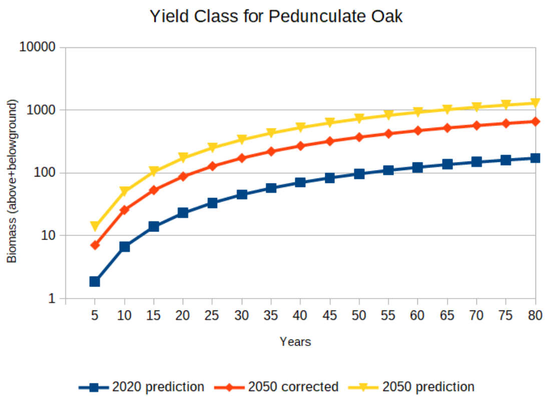


Figure 4.5. Baseline (2020, left map) and Yield Class differences from the baseline (2050 and 2070) using the lower end of the 95% confidence interval of predictions (GAMLI) for Pedunculate oak (top) and Sitka spruce (bottom). Sitka Spruce shows a varied pattern, with the highest decline in YC on the east coast but also potential increase in the southern uplands and the west coast. The X and Y map axes are OS UK-grid coordinate values.

4.4.3 Growth curve correction

The example in Fig. 6 illustrates the need for climate correction. It reproduces, for an hypothetical grid cell, the passage from YC1 (blue line) in 2020 to YC8 (yellow line) in 2050, and the proposed correction of the biomass values (red line), for *Sitka spruce* and *Pedunculate oak.* Assuming constant climate, a hypothetical forest patch planted in 2020 would follow the blue line, accumulating, after 5 years, 1.85 and 10.01 tCO_2-eq/ha and 172.46 and 227.01 tCO_2-eq/ha after 80 years, for *Sitka spruce* and *Pedunculate oak* respectively. Applying the growth curve for the present YC (in blue) leads to a different conclusion than applying the climate-corrected growth curve (red) regarding the difference in CO_2-eq between the two species at that location, as Pedunculate Oak is projected to benefit more from climate change in this case (this tendency is not widespread everywhere, as other locations might see a decline compared to the constant climate case). Using the corrected curves also potentially leads to different conclusions regarding the net carbon storage potential at that site. Notably, in the long run, *Pedunculate oak* has the potential to stock more carbon than *Sitka spruce* as hardwood usually have higher wood density than softwood, so a projected growth for 2050 (yellow) has the potential to accumulate 896.95 and 1295.20 tCO_2-eq/ha in 80 years-time for *Sitka spruce* and *Pedunculate oak* respectively, reversing the conclusion coming from the assumption of a constant climate. Therefore, to avoid bias the climate-corrected growth curve was used.



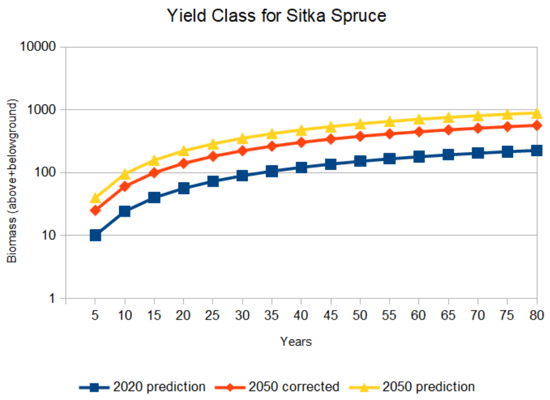


Figure 4.6. Corrected growth curve (red) of biomass (tCO2-eq/ha) for *Sitka spruce* and *Pedunculate oak*. The red curve passes through the middle point. The bottom blue curve is the one followed ignoring climate change (present climate). The top yellow curve is the one followed assuming that a new YC (reached in 2050) applies instantaneously from the present.

4.4.4 Co2-equivalent stock maps

Fig. 7 shows the aggregated future YC across the eleven modelled species obtained using the maximum of all YC layers corrected with the average growth curve technique (see 3.4). The two temporal snapshots represent the maximum potential carbon storage in the case of planting the most productive species at each location.

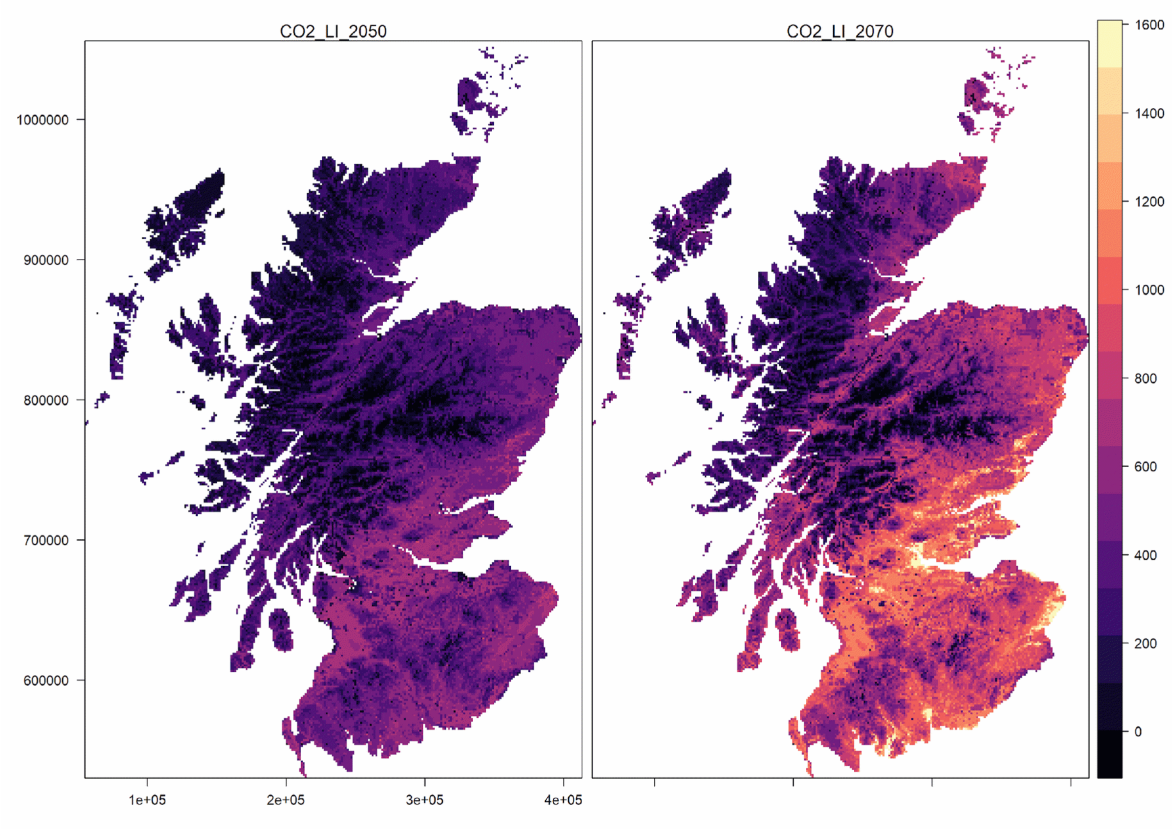


Figure 4.7. The potential maximum carbon sequestration (C02-eq/ha) by tree biomass (timber, branchwood and roots) for trees planted in 2021 in 2050 (left) and 2070 (right). Predictions using GAMLI (lower confidence interval).

The graphical representation of YC results above (Fig. 4.7) depicts, using different colours, the CO_2-eq per ha distribution for 2050 and 2070 according to the climate projections of the individual species. The biomass stock was translated into the corresponding amount of sequestered carbon dioxide through a look-up table (Supplementary material, Appendix F).

Gross Carbon Dioxide sequestration maps (i.e. excluding losses from soil) show that areas with highest potential are located in the lowlands of central Scotland and along the Borders region, while the locations at the medium-high elevation (above 600m) in the Highlands, Grampian and Strathclyde areas are characterized by the lowest potential carbon sequestration values. A very similar pattern was recorded in the comparison between 2050 and 2070 and this reflects the different levels of resilience of the eleven modelled species to future climatic pressures. However, as expected, there are substantial differences in carbon storage potentials, with the predictions for 2070 able to accumulate up to 1504.08 tCO_2/ha, while new trees cannot store more than 902.35 tCO_2/ha by 2050, which is the more pressing time horizon from a policy point of view.

4.4.5 Balance of soil carbon with tree carbon

Soil carbon release estimates considering a high intensity (HISCL) ground preparation scenario (business-as-usual) show, as expected, a lower carbon potential release from soil in the lowlands, while most of the Highlands are characterized by higher potential release values with peaks up to 1000 tCO_2/ha in the south west, north and centre-north. The net effect, considering potential tree carbon sequestration therefore suggests the unsuitability, for *carbon offsetting* purposes, of those areas (unless low soil-disturbance ground preparation techniques are applied).

Figure 8 compares, for each ground preparation intensity, the sign of the carbon budget estimated using the lower end of the confidence interval and the median (GAMLI and GAMME as said above). GAMLI and GAMME provide very similar patterns of suitability for expansion (where suitable means resulting in carbon gain). However, the ground preparation intensity makes a dramatic difference, with rather large areas of the uplands becoming suitable at low intensity (but see the discussion session on this point).

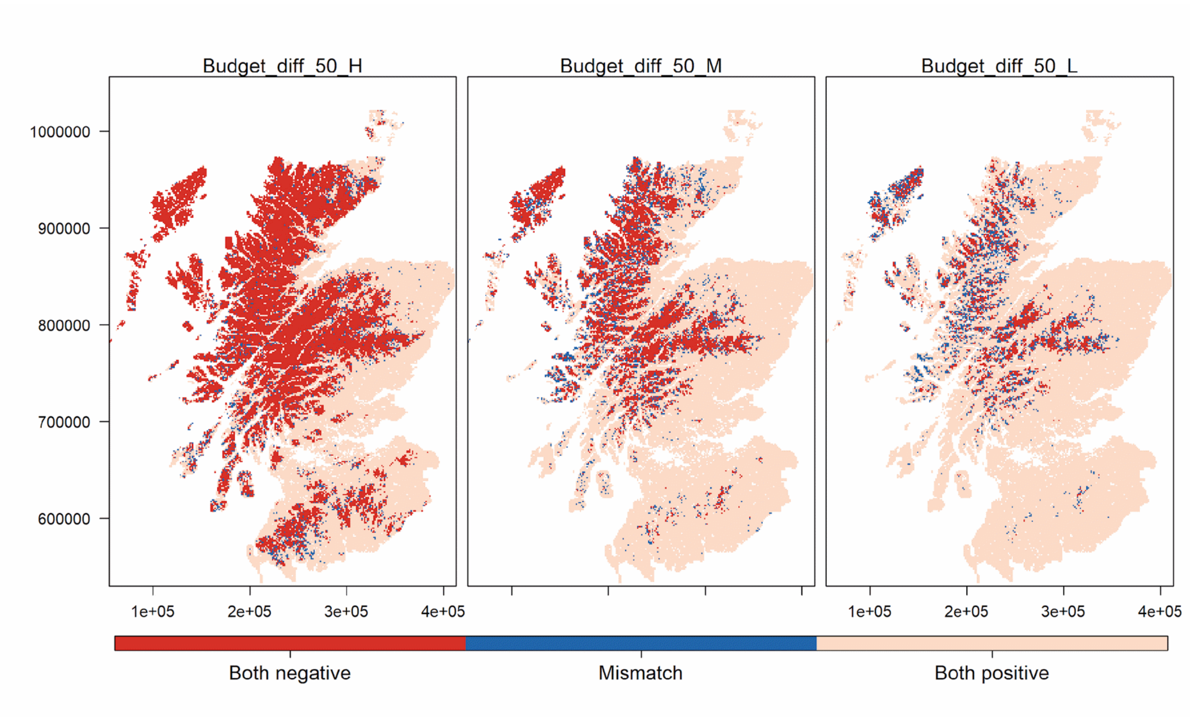


Figure 4.8. Comparison of the spatial pattern of the carbon budget calculated using GAMLI (lower end of the CI of predictions) and GAMME (median) for 2050 for the three scenarios (H=high, M=medium, L=low) of soil disturbance intensity.

4.5 Discussion

We have shown that afforestation in Scotland can contribute substantially to carbon storage in the land use sector through woodland expansion. However, our results also show that, to avoid a net release of carbon, accounting for spatial heterogeneity in the landscape, and for time evolution of species' suitability is very important. Therefore, it is crucial to be mindful of soil-based limitations, as well as their interactions with the intensity of ground preparation practices. This is relevant, for example, to ensure that any afforestation incentives don't produce undesired effects.

These results broadly agree with those by Matthews et al., 2020, and additionally provide spatially and temporally more resolved estimates of YC and carbon balance which account for the possible effects of climate change on tree growth.

The approach we used was a conservative one, to avoid under-estimating the potential to release carbon unwittingly. However, we have also shown that using more optimistic estimates for YC (the median) does not substantially alter the results in terms of which locations should be avoided. On the other hand, we might also have underestimated soil carbon losses due to climate warming which would bias the results in the opposite direction compared to the use of the low end of the confidence interval for the predicted YC.

Nonetheless it should be borne in mind that we used RCP8.5 as an upper limit to climate change, and that the use of other concentration pathways is likely to give intermediate results, with less marked differences in future YC. Future work will be extended to such intermediate scenarios.

Mineral soils were almost always suitable and organic soils often unsuitable or suitable only if planting is carried out with low-level soil disturbance (low intensity) ground preparation methods. Intensity of practice therefore makes a crucial difference, with a much larger area suitable for carbon storage when low intensity practices were used. Assuming that intensive ground preparation practices were adopted indiscriminately, afforesting large areas of the uplands could result in a net carbon loss in the next decades. Afforestation, however, does have a number of potential benefits, such as the provision of habitat, of landscape-level connectivity, the alleviation of soil erosion, mitigation of water pollution, alleviation of flooding and provision of shade for streams populated by temperature-sensitive species. Consequently, it could be practiced in sensitive areas using low-intensity methods. In any case, there is likely to be a trade-off between cost per ha and speed of expansion on the one hand, and planting method on the other hand. Having a time window for halving global emission that is drastically narrowing (Höhne et al., 2020), the speed of tree growth become an important factor for offsetting emissions, as most upland areas are also less suitable than lowland areas for rapid storage and therefore rapid mitigation of climate change, due to the lower productivity of upland trees.

While carbon offsetting is in principle possible in the lowlands and the ambition by the Scottish Government to plant some 450,000 ha or more appears theoretically feasible, there is also the potential for local conflict with farming. Also, if farming activities were to 'leak' abroad to avoid conflict, the net effect could well be that emissions are increased (see, e.g. Hoang and Kanemoto 2021). This point needs further study.

Returning to land-use conflicts, it is interesting to notice that newly planted woodlands in recent decades were established on marginal, not very productive land (which might have resulted in net carbon losses) while improved agricultural land has tended to be avoided (Brown, 2020). This agrees with what is known about farmers' attitudes to woodland planting (e.g. Slee et al., 2014) and indicates that resistance to carbon-friendly woodland expansion should be expected in the lowlands, where land opportunity costs are likely to be higher (i.e. more productive agricultural areas).

Recently, a number of studies have suggested that species' habitat suitability will be altered under the changing global climate, with winners and losers (Taylor et al., 2012; Cunze et al., 2013; Buczkowski and Bertelsmeier, 2017; Wei et al., 2017; MacKenzie et al., 2021). Our analysis corroborates such findings, and underpins actionable guidance for forestry policy by providing spatial and temporal detail for the eleven tree species considered. It suggests that, while the pattern is spatially variable, the predicted YC declines in many lowland areas for boreal species such as *Betula sp.* (Birch) are a reflection of worsening growing conditions, likely to become less favourable especially after 2050. While the increase in YC -especially in the lowlands- of more southern species such as Pedunculate Oak, Ash, or Wych Elm reflect warming conditions. However, such an increase was not sufficient to make these species a better choice for carbon storage over exotic conifers. From this point of view, Sitka Spruce was generally the dominant species in the Scottish landscape across all years, with localised exceptions where Douglas Fir or Lodgepole pine were predicted to fare better (e.g. SE lowlands). An important caveat, however, is that this conclusion implicitly assumes that the timber of the harvested conifers stores carbon for a long time, e.g. by being used to make long-lasting manufactured products.

It is also worth noting that the changes in the predicted spatial patterns of YC imply that present conditions might not reflect future success of afforestation efforts if such efforts implicitly assume no climate change by matching species to present suitability. A forward-looking policy, aiming to enhance environmental resilience, would therefore use a mixture of species likely to succeed based on their present and future projected suitability.

This study contributes to the debate regarding if and how woodland expansion should be carried out as a climate change mitigation measure. While the UK Committee on Climate Change (2020) has called for a substantial expansion of (commercial and broadleaved) forests, this paper shows that the common assumptions that these should be located in the uplands is often not tenable for *effective carbon offsetting*. On the other hand, forest expansion in the uplands with low intensity ground preparation methods appears feasible from these results but would have to be aimed mainly at other benefits, and would be likely to be achieved more gradually, eventually reaching comparable carbon storage (see e.g. Crane, 2020). This is also in broad agreement with the limited experimental results that have shown that, on organic soils, no appreciable net carbon gain and sometimes net loss can result even with low-intensity ground preparation methods (Friggens et al., 2020). We therefore concur with authors that have urged caution (e.g. Veldman et al., 2015; Seddon et al., 2019) and note that reliance on carbon sequestration by trees needs to be put into perspective. For example, the sequestration through afforestation, achievable for the UK *over 30 yrs* by the optimistic Tailwinds scenario of the UK Committee on Climate Change, (UKCCC, 2020), is ca 180 million tonnes carbon dioxide equivalent (Mt CO_2-eq) over 30 years. This is less than 1% of the total Greenhouse Gas (GHG) emissions footprint for which the UK would be responsible over the same period if the annual carbon footprint remains similar (BAU) as the footprint of 772 Mt CO_2-eq estimated for the UK in 2017(DEFRA, 2020).

As to what species to use, to maximise carbon gains, our estimates of the amount of carbon that could be stored over 30 years by afforesting the most productive 450,000 ha in Scotland using only exotic conifers vs only native species are 272 and 203 Mt CO_2-eq, respectively (see supplementary material E). Consequently, if, for the sake of argument, we assumed that only exotic conifers were used, the carbon gain over 30 years would be between 10% and 13% of the Scottish total BAU carbon footprint over that period, (i.e. of 70.7x30 Mt-CO_2-eq, as the total yearly carbon footprint of Scotland in 2017 was 70.7 Mt of CO_2-eq, Scottish Government, 2021). Although those figures represent an important potential achievement over 30 years, they are still relatively modest compared to the whole footprint. The difference between using only exotic conifers or only native species is around 3 percentage points, so the carbon-based case for exotic conifers does not appear particularly strong. A more in-depth uncertainty analysis would provide a range of a few more percentage points but not change the order of magnitude of the quantities involved. This should also be weighed against other multiple benefits that might derive from planting native species. However, the extra carbon footprint derived from sourcing timber from overseas would also need to be considered when comparing options.

In summary, while there are often cases in which it should be encouraged, it appears that expectations about tree planting as a mitigation tool have to be managed. This also supports the concerns of critics that see dangers in moving away from systemic approaches to concentrate on simple solutions (e.g. Watt, 2021). Moreover, the consequences of this analysis are bi-directional, in fact if there so much interest and attention in the use of afforestation practice to reduce a limited quantity of GHG emission, the same level of effort should be address to reduce carbon emission anyway in other sectors to avoid jeopardise the existence of native tree species in the future.

Finally, the approach we have illustrated, treating soil and trees as part of the same system, has wide applicability, uses widely available climatic data and scenarios and can therefore contribute to assessing forest-based climate mitigation policies worldwide.

Acknowledgements

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References

Boorman, D.B., Hollist, J.M. and Lilly, A. 1995. Report No. 126 Hydrology of soil types: a hydrologically-based classification of the soils of the United Kingdom. Institute of Hydrology.

Breiman, L. (2001). Random forests. Machine learning, 45(1), 5-32.

Brown, I., 2020. Challenges in delivering climate change policy through land use targets for afforestation and peatland restoration. Environmental Science & Policy, 10, 36-45

Climate Change Committee. (2020). The Sixth Carbon Budget: The UK's Path to Net Zero. Methodology Report. https://www.theccc.org.uk/wp-content/uploads/2020/12/The-Sixth-Carbon-Budget-Methodology-Report.pdf

Crane, E. 2020. Woodlands for Climate and Nature: A Review of Woodland Planting and Management Approaches in the UK for Climate Change Mitigation and Biodiversity Conservation. RSPB. http://ww2.rspb.org.uk/Images/Forestry%20and%20climate%20change%20report%20Feb%202020_tcm9-478449.pdf (accessed on 15th of March 2021).

DEFRA 2020, UK's carbon footprint. Annual greenhouse gas and carbon dioxide emissions relating to UK consumption.Available online at: https://www.gov.uk/government/statistics/uks-carbon-footprint [last accessed 12.04.2021]

Ebi, K. L., Hallegatte, S., Kram, T., Arnell, N. W., Carter, T. R., Edmonds, J., Kriegler, E., Mathur, R., O'Neill, B. C., Riahi, K., Van Vuuren, D. P., Zwickel, T., & Winkler, H. (2014). A new scenario framework for climate change research: Background, process, and future directions. Climatic Change, 122 (3), 363–372. https://doi.org/10.1007/s10584-013-0912-3

Eyring, V., Bony, S., Meehl, G. A., Senior, C. A., Stevens, B., Stouffer, R. J., and Taylor, K. E.: Overview of the Coupled Model Intercomparison Project Phase 6 (CMIP6) experimental design and organization, Geosci. Model Dev., 9, 1937-1958, doi:10.5194/gmd-9-1937-2016, 2016.

Fick, S.E. and R.J. Hijmans, 2017. WorldClim 2: new 1km spatial resolution climate surfaces for global land areas. International Journal of Climatology 37 (12): 4302-4315.

Friggens, N. L. et al. (2020) 'Tree planting in organic soils does not result in net carbon sequestration on decadal timescales', Global Change Biology, (April), pp. 1–11. doi: 10.1111/gcb.15229.

Höhne, N., den Elzen, M., Rogelj, J., Metz, B., Fransen, T., Kuramochi, T., ... & Dubash, N. K. (2020). Emissions: world has four times the work or one-third of the time.

Hollister, J.W., Tarak Shah, 2017. elevatr: Access Elevation Data from Various APIs.

IPCC, 2019: Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems [P.R. Shukla, J. Skea, E. Calvo Buendia, V. Masson-Delmotte, H.-O. Pörtner, D. C. Roberts, P. Zhai, R. Slade, S. Connors, R. van Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. Portugal Pereira, P. Vyas, E. Huntley, K. Kissick, M. Belkacemi, J. Malley, (eds.)]. https://www.ipcc.ch/srccl/

IPCC, Summary for policy-makers. Stocker T., Qin D., Plattner G.K., Tignor M., Allen S., Boschung J., Nauels A., Xia Y., Bex V., Midgley P. (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA (2013)

Keywan Riahi, Detlef P. van Vuuren, Elmar Kriegler, …. Massimo Tavoni, (2017). The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview, Global Environmental Change, 42, 153-168, ISSN 0959-3780, https://doi.org/10.1016/j.gloenvcha.2016.05.009.

LandIS database, Cranfield University 2021. The Soils Guide. Available: www.landis.org.uk. Cranfield University, UK.

Liaw, A. and Wiener, M. (2002). Classification and Regression by randomForest. R News 2(3), 18--22.

Lilly, A. (2010). A hydrological classification of UK soils based on soil morphological data. In 19th World Congress of Soil Science, Soil Solutions for a Changing World (Vol. 16). IUSS Brisbane, Australia.

Lilly, A, Baggaley, N and Donnelly, D. 2012. Map of soil organic carbon in top soils of Scotland. Map prepared for EU project GS-SOIL -Assessment and strategic development of INSPIRE compliant Geodata-Services for European Soil Data. ECP-2008-GEO-318004.

Macek, Martin, Martin Kopecký, and Jan Wild. "Maximum air temperature controlled by landscape topography affects plant species composition in temperate forests." Landscape Ecology 34.11 (2019): 2541-2556.

MacKenzie, W.H. and Mahony, C.R. 2021. An ecological approach to climate change-informed tree species selection for reforestation. Forest Ecology and Management, 481, 118705.

Marchi, Maurizio, et al. "Evaluating WorldClim Version 1 (1961–1990) as the baseline for sustainable use of forest and environmental resources in a changing climate." Sustainability 11.11 (2019): 3043.

Matthews, R. W. et al. (2016) Forest Yield. A handbook on forest growth and yield tables for British forestry. Edinburgh: Forestry Commision.

Navarro-Racines, C., Tarapues, J., Thornton, P. et al. High-resolution and bias-corrected CMIP5 projections for climate change impact assessments. Sci Data 7, 7 (2020). https://doi.org/10.1038/s41597-019-0343-8

Neteler, M., Bowman, M. H., Landa, M., & Metz, M. (2012). GRASS GIS: A multi-purpose open-source GIS. Environmental Modelling & Software, 31, 124-130.

Ovando (2020) Application of the Natural Capital Protocol at Glensaugh farm. The James Hutton Institute. Available online at: https://www.hutton.ac.uk/research/projects/integrating-natural-capital-thinking-land-based-decision-making)

Pauw, W. P., & Klein, R. J. (2020). Beyond ambition: increasing the transparency, coherence and implementability of Nationally Determined Contributions. Climate Policy 20 (4), 405-414. Taylor & Francis.

Panagos, Panos, et al. "Towards estimates of future rainfall erosivity in Europe based on REDES and WorldClim datasets." Journal of Hydrology 548 (2017): 251-262.

Poggio, L., Simonetti, E., & Gimona, A. (2018). Enhancing the WorldClim data set for national and regional applications. Science of the Total Environment, 625, 1628-1643.

Poggio, L., Gimona, A., Brown, I., & Castellazzi, M. (2010). Soil available water capacity interpolation and spatial uncertainty modelling at multiple geographical extents. Geoderma, 160(2), 175-188.

Poggio, L., Gimona, A., & Brown, I. (2012). Spatio-temporal MODIS EVI gap filling under cloud cover: An example in Scotland. ISPRS Journal of Photogrammetry and Remote Sensing, 72, 56-72.

Poggio, L., & Gimona, A. (2014). National scale 3D modelling of soil organic carbon stocks with uncertainty propagation—an example from Scotland. Geoderma, 232, 284-299.

Poggio, L., & Gimona, A. (2017). 3D mapping of soil texture in Scotland. Geoderma regional, 9, 5-16. R

Core Team (2020). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Schueler, S.; Falk, W.; Koskela, J.; Lefèvre, F.; Bozzano, M.; Hubert, J.; Kraigher, H.; Longauer, R.; Olrik, D.C. Vulnerability of dynamic genetic conservation units of forest trees in Europe to climate change. Global Change Biology 2014, 20, 1498–1511.

Schwalm, C.R., Glendon, S. and Duffy, P.B. (2020). RCP8.5 tracks cumulative CO2yesisoneofthemi emissions. PNAS, 117 (33) 19656-19657

Seddon, N.; Turner, B.; Berry, P.; Chausson, A.; Girardin, C.A.J. 2019. Grounding nature-based climate solutions in sound biodiversity science. Nature Climate Change, 9, 84–87

Scottish Government (2021). Scotland's Carbon Footprint: 1998-2017. s. Available online: https://www.gov.scot/news/scotlands-carbon-footprint-1998-2017/ [last accessed 12.04.2021].

Simola, H., Pitkänen, A. and Turunen, J. (2012). Carbon loss in drained forestry peatlands in Finland, estimated by re-sampling peatlands surveyed in the 1980s. European Journal of Soil Science, 63 (6), p.798–807

Sinka, M.E., Pironon, S. .. Willis, K.J. (2020). A new malaria vector in Africa: Predicting the expansion range of Anopheles stephensi and identifying the urban populations at risk. PNAS, 117 (40) 24900-24908;

Slee,B., Brown, I., Donnelly, D. Gordon, I.J., Matthews, K., Towers, W. 2014. The 'squeezed middle': identifying and addressing conflicting demands on intermediate quality farmland in Scotland. Land Use Policy, 41, 206-216

UK Committee on Climate Change (2020). Land use: Policies for a Net Zero UK. Available at https://www.theccc.org.uk/publication/land-use-policies-for-a-net-zero-uk/. Accessed in April 2021.

Vanguelova, Elena, Steve Chapman, Mike Perks, Sirwan Yamulki, Tim Randle, Frank Ashwood, and James Morison. "Afforestation and Restocking on Peaty Soils – New Evidence Assessment," 2018, 1–43. www.climatexchange.org.uk.

Veldman, J.W.; Overbeck, G.E.; Negreiros, D.; Mahy, G.; Le Stradic, S.; Fernandes, G.W.; Durigan, G.; Buisson, E.; Putz, F.E.; Bond, W.J. Where Tree Planting and Forest Expansion are Bad for Biodiversity and Ecosystem Services. Bioscience 2015, 65, 1011–1018

Voldoire, A., Saint-Martin, D., Sénési, S., Decharme, B., Alias, A., Chevallier, M., et al. (2019). Evaluation of CMIP6 DECK experiments with CNRM-CM6-1. Journal of Advances in Modeling Earth Systems, 11, 2177– 2213. https://doi.org/10.1029/2019MS001683

Wan, Ji-Zhong, Chun-Jing Wang, and Fei-Hai Yu. "Impacts of the spatial scale of climate data on the modeled distribution probabilities of invasive tree species throughout the world." Ecological Informatics 36 (2016): 42-49.

Wang, T., Hamann, A., Spittlehouse, D., & Carroll, C. (2016). Locally downscaled and spatially customizable climate data for historical and future periods for North America. PloS one, 11(6), e0156720.

West, V., 2018. Lookup Table Guidance Using the WCC Carbon Calculation Spreadsheet Version 2.0 1–21. Available online at: https://www.woodlandcarboncode.org.uk/news/version-2-0-of-the-wcc-launched [last accessed 12.04.2021].

Watt, R. 2021. The fantasy of carbon offsetting. Environmental Politics, published online DOI: 10.1080/09644016.2021.1877063

Wood S.N., N. Pya and B. Saefken (2016) Smoothing parameter and model selection for general smooth models (with discussion). Journal of the American Statistical Association 111:1548-1575.

Wood, S. N. (2004). Stable and efficient multiple smoothing parameter estimation for generalized additive models. Journal of the American Statistical Association, 99(467), 673-686.

Wood, S. N. (2017). Generalized additive models: an introduction with R. CRC press.

Zomer, Robert, et al. Carbon, land and water: A global analysis of the hydrologic dimensions of climate change mitigation through afforestation/reforestation. Vol. 101. IWMI, 2006.

Chapter 5 – Modelling water purification in Scotland through the use of forest expansion scenarios

(Part of an Interim report)

Baggio Compagnucci A.[ab], Castellazzi M.[a], Machala M.[d], Hamel P.[e], Poggio L.[c], Irvine J.[a], Gimona A.[a].

[a] *The James Hutton Institute, Craigiebuckler, AB158QH Aberdeen, Scotland, UK*

[b] *School of Biosciences and Veterinary Medicine, Plant Diversity and Ecosystems Management Unit, University of Camerino, 62032 Camerino, MC, Italy*

[C] *ISRIC World Soil Information, Wageningen, Netherlands*

[d] *Silva Tarouca Research Insitute (VUKOZ), Department of Forest Ecology, Lidickà 25727, 60200, Brno*

[e] *The Natural Capital Project. Woods Institute for Environment, 371 Serra Mall, CA 94305-502, Stanford, USA*

5

5.1 Introduction

In many regions of the world, the rapid development of agriculture, intensive use of fertilisers and and increased nitrogen (N) deposition have dramatically changed the natural N cycle. The impacts of this change are well documented, including the eutrophication of freshwater and coastal ecosystems, biodiversity losses, and potential contamination of drinking water. (e.g. McLauchlan et al., 2013; Cranfield, 2010). Improved practices in nutrient management have been developed to mitigate these detrimental effects, including a number of structural measures such as constructed wetlands, sediment ponds, and riparian buffers. While their efficacy has been variable in the past, recent advances in the science of nutrient transport allow their implementation to be optimised. The refined understanding of nutrient dynamics now allows the identification of “hotspots” areas of a catchment that contribute the most to nutrient export (e.g. Kovacs et al., 2012). Besides, agriculture development could have a major effect on erosion of soil and its range of key functions rising environmental concerns. These functions comprise the production of food, the storage of organic matter, water and nutrients and the provision of habitat for a wide variety of organisms. This concern has been recognized globally considering that erosion rates from ploughed agricultural fields average 1–2 orders of magnitude greater than rates of soil production, and long-term geological erosion (Montgomery, 2007). While ploughing and removal of crop residuals from land to use as burning fuel are two of the main factors that can increase soil removal, erosion rates are high on marginal and steep lands converted from forest to agricultural use to replace the already eroded, unproductive cropland (Lal and Stewart, 1990) thus deforestation can have a major impact on soil erosion. Such knowledge helps target mitigation activities and design optimal nutrient and sediment management plans.

However, solving the nutrient and sediment issues globally does not simply hinge on the science and application of best field management practices. Nutrient and sediment management plans typically encompass a broader suite of socio-economic factors such as the expected yield of agricultural land, the financial cost of mitigation activities, or the recreational use of regional water bodies. For example, while decreasing the amount of agricultural areas may be the simplest way to decrease nutrient and sediment export, trade-offs will result (e.g. Sutton et al., 2013). Similarly, the optimal solution to mitigate nutrient and sediment export may involve riparian buffers along the streams, but budget restrictions would make this solution inefficient if it were implemented in half of the catchment only. Therefore, in-depth understanding of nutrient and sediment dynamics may be only a minor component of the complex challenge faced by land planners.

To aid decision-making in such complex multi-objectives problems, approaches relying on mapping ecosystem services are increasingly used (e.g. Bastian et al., 2013; Maes et al., 2012). Ecosystem services are here simply defined as the benefits that people obtain from ecosystems (e.g. MAE, 2005). By considering the multiple benefits that land provides and quantifying their values in the local context, scientists can better understand the trade-offs inherent in a policy decision and thus present a broader view of the issue to decision-makers (e.g. Johnson 2012). Ecosystem services models have thus can help been developed to identify areas in the landscape where natural benefits are provided or lacking, and thus to plan land management accordingly. The Integrated Valuation of Environmental Services and Trade-offs (InVEST) tool (http://www.naturalcapitalproject.org/InVEST.html) provide several models. Here, the focus is on those allowing estimation of nutrient and sediment retention and as well as export (as one is the complementary side of the other). However, the use of such generic models is limited by the knowledge of ecosystem service production, and by the simplification of the biophysical processes: in fact, the multiplicity of services represented may come, in practice, at the expense of individual models sophistication.

Selecting a complex model is ultimately dictated by the available data to select, parameters and the resources available for the analyses. In particular, in projects with low data availability or where time constraints are tight, simple models such as InVEST are very attractive. Yet little is done (Verhagen et al., 2016) about the level of information provided by such models as a consequences of land use change in Scotland. This work is part of an interim report (Gimona et al., 2019) which illustrated a spatial multi-criteria approach coupled with ecosystem services models for assessing some of the consequences of scenarios of land use change. Here we focus on the description of models of nitrogen and sediment export to assess terrestrial ESs in the Cairngorms National Park (Appendices).

5.1 Data

Nutrient and sediment models requires different input datasets. While some of them are used by both models, others are specific. The following table and paragraphs summarized the data required to run the two modules of InVEST framework.

The data used were common as far as the two models allowed, given their different time scales and process representation. Table 1 summarises the input data for the two models. For the nutrient module, we used baseline climate grids from the UK Meteorological office: precipitation, temperature, sun hours, wind speed, relative humidity, air pressure, temperature to calculate evapotranspiration using the FAO56 P-M method (Allen et al., 1998). The Digital Terrain Model (10-m resolution) was hydrologically corrected using the algorithm described in Soille, 2004. Soil depth was obtained from the national NSIS soil data base (Lilly et al., 2010) and interpolated using the method described in Poggio et al., 2010. The soil available water content was derived from Gimona and Birnie, 2002. All data were resampled at 25 m resolution to match that of the land cover data.

With regard to the nutrient modules, inputs were calculated in table format to be linked with LU classes of the land cover map. First, N loads were calculated for each LU class as the sum of depositional, organic and inorganic N. Atmospheric N depositions estimates were taken from the European Monitoring and Evaluation Programme (EMEP, 2012).

The N input per land cover class, was derived from literature sources detailed in Table 2.

To better represent the spatial variability of the N input, the initial 23 land cover classes were further categorised based on the intersection between land use classes and a Integrate Administration Control System (IACS) data which represents the main building block of the management of payments from the European agricultural guarantee fund (EAGF). In addition, for each land class, a map of N input from livestock was obtained as follows. Based on the gridded 2 km June Census produced by the Scottish Government (EDINA), the density of cows (beef and dairy), sheep, pigs, poultry was calculated separately for each 25 m cell, by distributing animals only on grazing land, and their contribution to N input estimated from their per capita average input. Next, retention rates were estimated from the total input and the total amount available for leaching in each cell.

ESS model	Input variable	Source
Soil erosion and retention modelling	Rainfall erosivity factor (R factor) (MJ mm ha−1 h−1 yr−1)	European Soil Data Centre (ESDAC)[1]
	Soil erodibility factor (K factor) (Mg h ha MJ−1 mm−1 ha−1)	European Soil Data Centre (ESDAC)[2]
	Slope length and steepness factor (LS factor)	European Soil Data Centre (ESDAC)[3]
	Cover and management factor (C factor) (0–1)	European Soil Data Centre (ESDAC)[4]
	Support practice factor (P factor) (0–1)	European Soil Data Centre (ESDAC)[5]
	Digital Elevation Model (DEM) (10 m)	Ordinance Survey[6]
Shared data	Land Use Cover	Hewitt et al., 2019
	Watershed (shapefile)	SEPA
Nutrient retention and export modelling	Root restriction	Literature[11] - Expert opinion[7]
	Plant available water content (AWC) (fraction content, 0–1)	Literature[8]
	Soil depth (mm)	Lilly A., 2010
	Average annual precipitation (mm)	MetOffice[9]
	Average annual potential evapotranspiration (mm)	Derived[10]

Evapotranspiration coefficient	Derived[10]
Nutrient loading (kg ha−1 yr−1)	Literature[11]
Vegetation filtering capacity	Literature[11]

Table 5.1 Sources of dataset: 1) European Commission, Joint Research Centre (Panagos eta al., 2017); 2) European Commission, Joint Research Centre (Panagos et a al., 2012a); 3) European Commission, Joint Research Centre (Panagos et a al., 2012b); 4) European Commission, Joint Research Centre (Panagos et a al., 2012b); 5) European Commission, Joint Research Centre (Panagos et a al., 2012b); 6) OS Terrain 10 GML 3.2; 7)1 m for crops and grass, 2 m for shrubs, 3 m for trees; 8) Gimona and Birnie, 2002; 9) Met Office, 2013; 10) Calculated using method of Allen et al., 1998; 11); See Table 2.

We assumed that the N of animal origin is available for plant uptake during the growing season –up to an upper limit and available for leaching in between (Fig. 5.1). On crop lands, we accounted for the length of the growing season. The total organic N input in each cell was split into a proportion assumed to be deposited during the plant growing season, and a proportion deposited outside that period. The ratio between the two was assumed to be the same as the ratio between the duration of the growing and non-growing season. The average duration of the growing season in each cell was estimated using TIMESAT (Jonsson and Eklundh, 2004) using a time series of 10 years of biweekly MODIS EVI images, pre-processed to clear the clouds (Poggio et al., 2012). This estimate is based on changes in vegetation greenness as seen by the satellite.

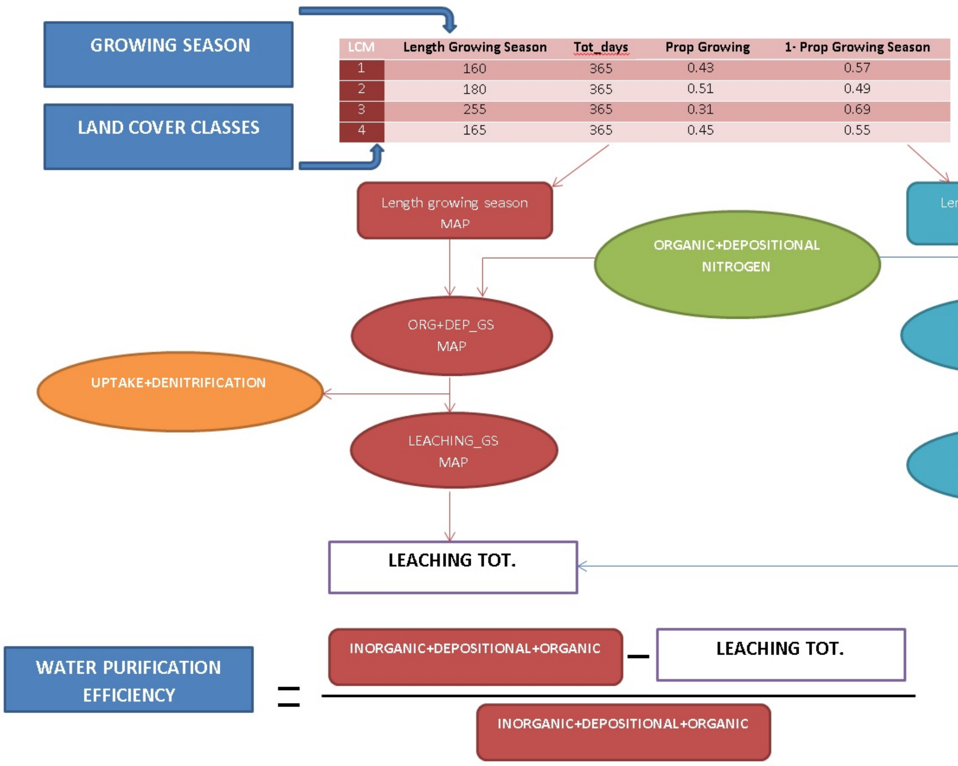


Fig. 5.1. Calculation of water purification efficiency for organic and inorganic N

5.1.1 Land use

A 25 m resolution spatial dataset was prepared by combining IACS land parcels land use aggregate dataset, agricultural land use classes only (categories ARABLE, TEMP_GRASS, IMPR_GRASS) with a data set combining LCM2007 (Morton, 2011, 25 m raster), and Scottish forest inventory data WoodR3 categories. To integrate the forest inventory data, broadleave and coniferous woodlands of a functional age (behaving as woodland cover) were extracted from the Native Woodlands Survey for Scotland 2014 (Patterson, 2014) and National Forestry Inventory for Scotland 2015 (NFIS, 2015) and used to expand the woodland extent originally in the LCM2007. Woodlands that did not fit neatly in the two LCM2007 land use categories were ignored, as well as shrubs, scrubs, and clear fell. The analysis was carried out in vector format with a 10m tolerance; the aggregated dataset was then converted to 25m raster.

Both the IACS and the LCM2007 woodland datasets were rasterized to 25x25m resolution and combined using raster map algebra (GRASS GIS), giving preference to IACS data in areas of overlap. The integration procedure is described in Hewitt et al., 2019. (Appendix G).

The dataset was made for 2015 time period and reclassified to LCM2007 categories to be used as input to InVEST models for nutrient and sediments export.

The data needed by the nutrient and sediment model of InVEST (InVEST 3.0.0, Sharp et al., 2015) are summarised in Table 1. The tool requires look-up tables for each land use class (see Table 2). Nitrogen (N) loading for each land use class in kg ha−1 yr−1 were derived from literature based on the land use classes characterisation (see Table 2 for further details).

lucode	LULC desc	Kc [5]	Root depth [7]	usle_c [2]	usle_p [2]	sedret_eff [3]	load_n	eff_n [6]	LULC_veg
3	Arable unspecified	1.15 [1]	1250	0.298	1	0.6	78 [6]	0.67	1
4	Improved Grassland	1	1000	0.220	1	0.7	90 [6]	0.27	1
5	Rough grassland	1	1000	0.312	1	0.7	44.2 [6]	0.4	1
6	Neutral Grassland	1	1000	0.263	1	0.7	98 [6]	0.36	1
7	Calcareus Grassland	1	1000	0.223	1	0.7	48.5 [6]	0.39	1
8	Acid grassland	1	1000	0.359	1	0.85	24.4 [6]	0.39	1
9	Fen, Marsh and Swamp	1	1000	0.293	1	0.8	12.3 [6]	0.4	1
10	Heather	1	500	0.391	1	0.99	19 [6]	0.39	1
11	Heather grassland	1	500	0.391	1	0.99	19 [6]	0.39	1
12	Bog	1	500	0.404	1	0.99	18.5 [6]	0.39	1
13	Montane Habitats	1	500	0.429	1	0.99	14 [6]	0.44	1
14	Inland Rock	1	500	0.448	1	0.99	14 [6]	0	1
15	Saltwater	1	0	0.372	1	1	14 [6]	0	0
16	Freshwater	0.95	0	0.423	1	1	14 [6]	0	0
17	Supra-littoral Rock	0.92	1000	0.406	1	0.05	14 [6]	0	1
18	Supra-littoral Sediment	0.92	1000	0.337	1	0.05	14 [6]	0	1
19	Littoral Rock	0.92	1000	0.373	1	0.05	14 [6]	0	1
20	Littoral Sediment	0.92	1000	0.353	1	0.05	14 [6]	0	1
21	Saltmarsh	0.92	1000	0.328	1	0.85	17.4 [6]	0.39	1
22	Urban	0.92	200	0.436	1	0.2	16.2 [6]	0.17	0

23	Suburban	0.92	200	0.391	1	0.2	16.2 [6]	0.17	0
101	Broadleaved woodland	1.5	3000	0.302	1	0.87	14.2 [6]	0.45	1
102	Coniferous woodland	1.5	3000	0.317	1	0.87	14.2 [6]	0.45	1
302	Temporary grassland	1 [5]	1000	0.221	1	0.7	111 [4]	0.27	1
303	Improved grassland	1 [5]	1000	0.221	1	0.7	79 [4]	0.27	1
3011	Oats	1.15	1250	0.298	1	0.6	91 [4]	0.67	1
3012	Potatoes	1.15	1250	0.298	1	0.6	123 [4]	0.67	1
3013	Rape	1.15	1250	0.298	1	0.6	177 [4]	0.67	1
3014	Spring barley	1.15	1250	0.298	1	0.6	97 [4]	0.67	1
3015	Winter barley	1.15	1250	0.298	1	0.6	155 [4]	0.67	1
3016	Winter wheat	1.15	1250	0.298	1	0.6	183 [4]	0.67	1

Table 5.2. List of data source: 1, Allen et al. (1998); 2, European Soil Data Centre (ESDAC); 3, InVEST Natural Capita dataset; 4, Wray (2015); 5, Dunn and Mackay (1995); 6, Estimate using growing season method. 7, Guess/estimate

5.2 Calibration and comparison

The River Dee catchment is a 2,100 km2 located in the East of Scotland. Its precipitation varies from 700 to 2000 mm, including snow over several months. Land cover is dominated by montane heath in the western areas, transitioning to heather moorland at lower altitude, and grassland, plantation forest, and agriculture in the lowlands (Fig. 5.2). Nutrient issues have been reported by the Scottish Environment Protection Agency (SEPA), with both diffuse pollution and point source pollution affecting stream condition. Understanding of the nutrient dynamics and their evolution in future climate scenarios is therefore of interest to water managers.

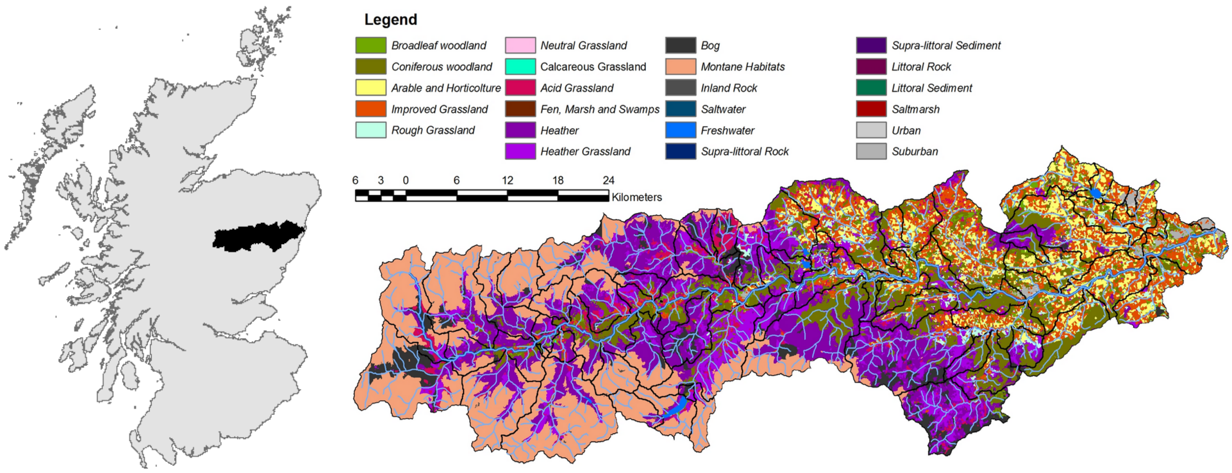
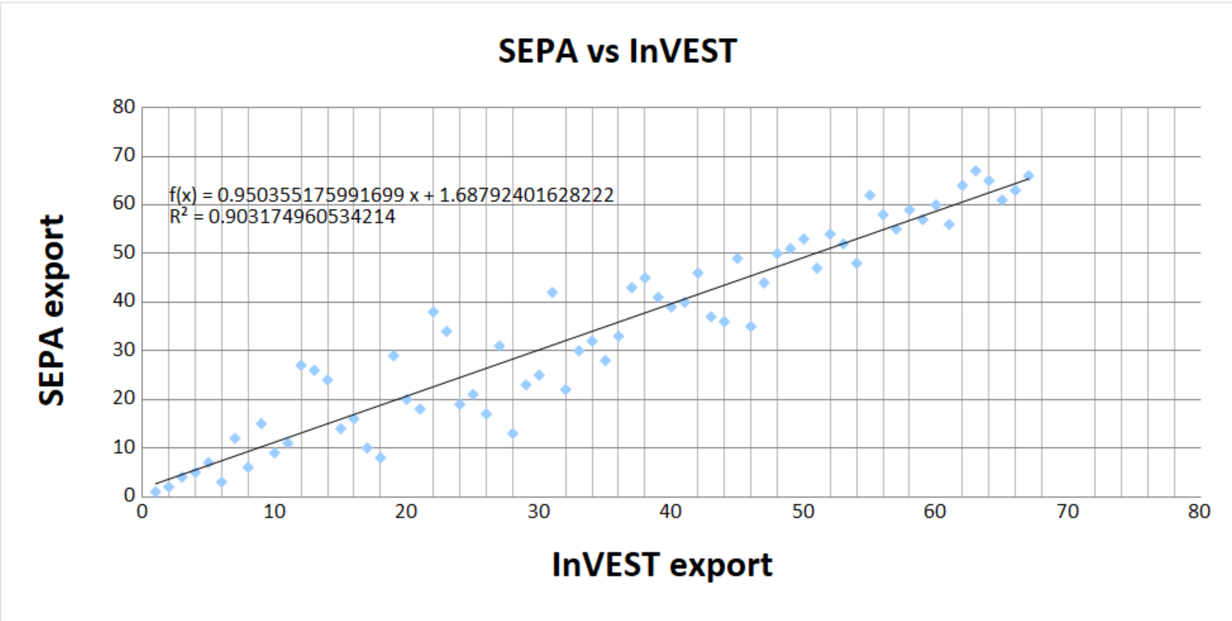


Figure 5.2. Map of the River Dee catchment showing the land uses and sub-catchment delineation (black lines).

The Dee catchment was used to run InVEST and to compare nutrient results with NIRAMS II (Dunn et al., 2004) model, while data obtained from SEPA were used for comparison with sediment module results.

NIRAMS II is a process-based, fully spatially distributed model designed to predict concentrations and fluxes of nitrate draining from agricultural land. The model is a development of the Nitrogen Risk Assessment Model for Scotland, which was created as a national scale screening tool for the EU Nitrates Directive. As such, the model has been used to simulate nitrate concentrations in agricultural areas designated under the Nitrates Directive, and it is currently being applied using climate change simulations from UKCP09 to investigate possible future changes to Scotland's water resources.

The regression analysis showed that in both cases (nutrient and sediment) the two sets of export results (ranked) were strongly correlated ($p < 0.001$), with R2 = 0.9 and 0.77. The absolute values between the two models differed, with InVEST predicting lower N export than NIRAMS and SEPA. However, a very strong correlation was found between the sub catchments ranks. Fig. 3 shows the scatter plot of ranked catchments between the two sets of export values.



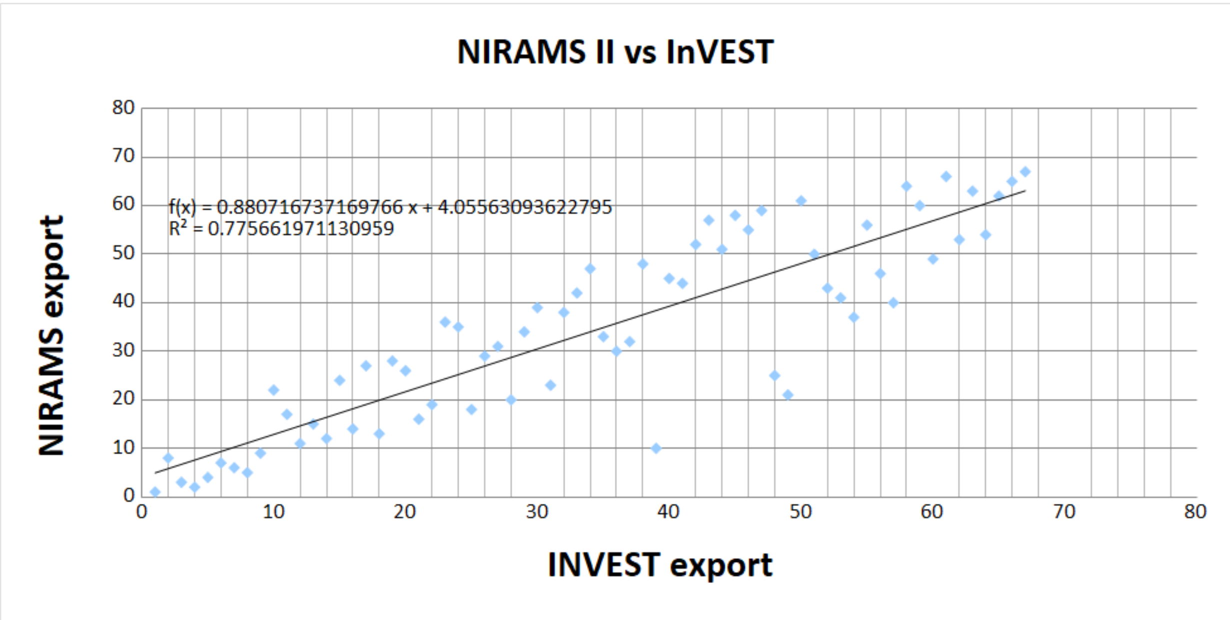


Fig. 5.3. Scatter plot of nitrogen (below) and sediment (above) export per sub-catchment (ranked) predicted by the two models against Nirams II (nitrogen) and SEPA data (sediment)

Overall, difference in export predictions is expected, given the different model structure and time resolution of the data used by the models. In InVEST, the use of annual averages for climatic variables, for example, probably results in missing some peaks in nutrient export due to intense rainfall events, especially outside the growing season. In addition, transport of leached nutrients is poorly taken into account in InVEST.

The main difference between the two models is the degree of complexity in the representation of soil dynamics, including a variety of data sources and attenuation processes. This distinction also implies a difference in the time and data required to run each model, which makes simpler models more attractive in projects where time and resources to run more complex models are lacking. Comparison showed that InVEST, the simpler model, provide the same level of information for relative comparisons of sub-catchments, such as needed in a context of spatial prioritization.

5.3 Results: Scenarios for 2050

We used three different scenarios that upscale the previous parametrization at national level and put different emphasis on potential benefits for the terrestrial environment and we further assess the benefits for aquatic ecosystems by modelling the reduction in nitrogen and sediment (soil) export to streams implied by the different planting scenarios.

The incorporation of climate change is only done through accounting for modelled changes in land capability for agriculture (based on Gimona et al., 2015).

The analysis presented here is limited to broadleaved woodlands because of their multifunctional nature. It could be easily expanded to include conifer plantations in the future. Three different scenarios were produced, starting from the baseline to analyse the consequences of woodland expansion in Scotland for 2050. An increment of 10,000 ha per year in the future 35 years was simulated.

Scenario name	**Water&Prime**	**Water&prime Future**	**Biodiv**	**Biodiv++**
Scenario name	M13	M14	B21	B22
Scenario ID	MCF23	MCF24	MCF11	MCF13
Land uses authorised to become broadleaves	Arable, all Grasslands, Heathers	Arable, all Grasslands, Heathers	Improved grassland	Improved grassland
Spatial targeting	Within 50m of lakes/rivers Slopes >= 7%	Within 50m of lakes/rivers Slopes >= 7%	Polygons touching within 100m of existing broadleaves	Polygons touching within 100m of existing broadleaves
Within 50 m from a stream	1	1	0	0
On wet mineral soil	0.5	0.5	0	0

Avoid present prime land	1	1	1	0
Avoid areas downstream of Built up area at risk of flooding	1	1	0	0
Avoid areas downstream of stretches of roads at risk of flooding	1	1	0	0
Avoid future prime land	0	1	1	
On dispersal corridors of forest spp.	0	0	1	1
In areas favourable for Biod. Action Plan species	0	0	1	1
Avoid good areas for forest plantation	0	0	0.5	0.5

Table 5.3. Spatial criteria for scenarios.

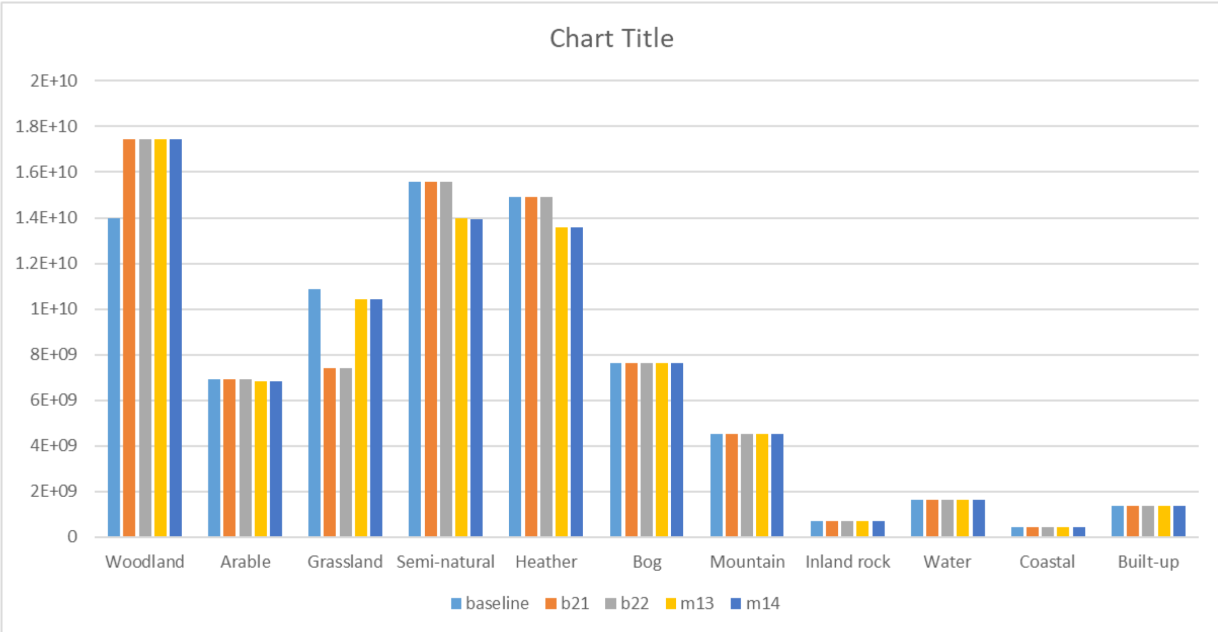
Each scenario was designed to emphasise a different balance of benefits. The final forest expansion map was produced applying a set of criteria at coarser resolution (250 m) using gridded (raster) data, while a refinement stage, was then used to weight previous results using polygon (vector) data.

The “Water&prime” land scenario uses criteria that privilege criteria related to water quality and flooding, and also protects prime agricultural land from planting.

The “Water&primeFut” scenario is as above but additionally protect agricultural land that is projected to become prime in 2050, using Metoffice climatic projections.

The “Biodiv” scenario includes criteria aimed at enhancing biodiversity, while still protecting future prime agricultural land.

The “Biodiv++” scenario includes same criteria of Biodiv scenario with no protection for future prime agricultural land.



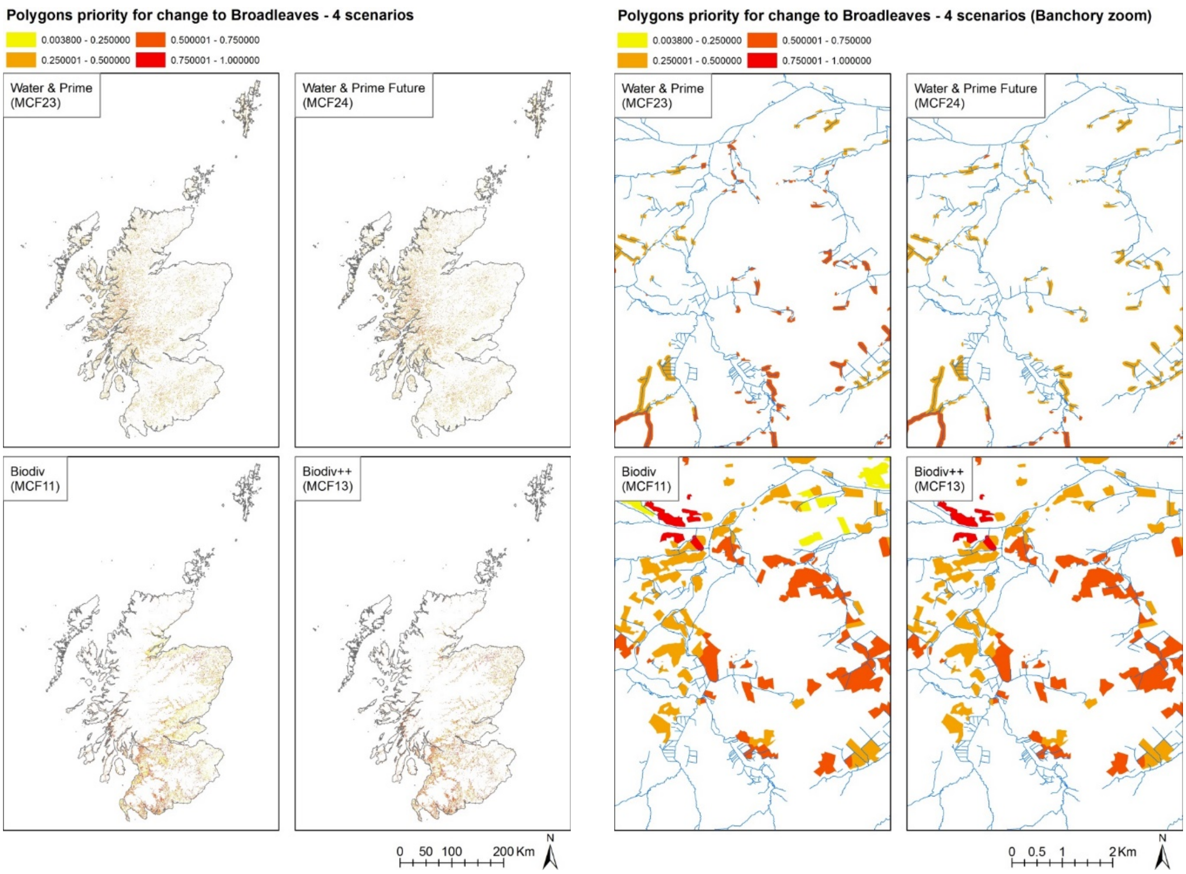


Figure 5.4. Graph of land use in each scenario (above), the opportunity map to change as defined by spatial criteria (left) and a detailed zoom in to the Banchory area of the same opportunity map (right).

The scenario/baseline export ratios are shown in Figures 5 to 12. For the uplands, there is a general reduction in export of nitrogen and sediment in all scenarios.

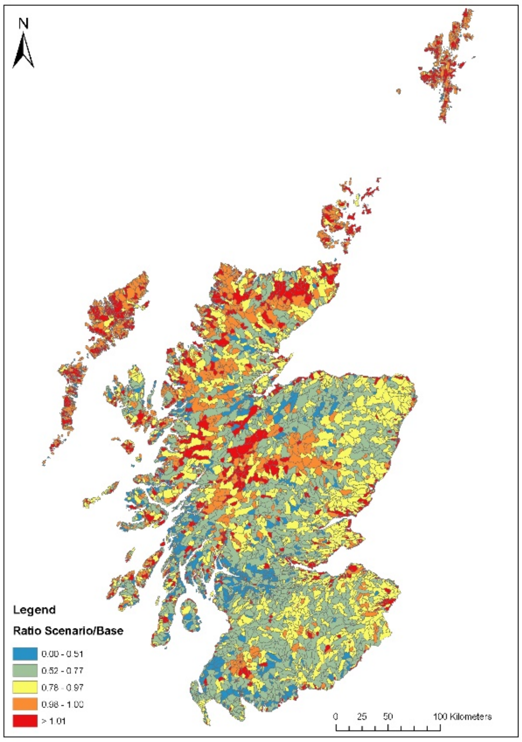


Figure 5.5. Nutrient export b21/baseline

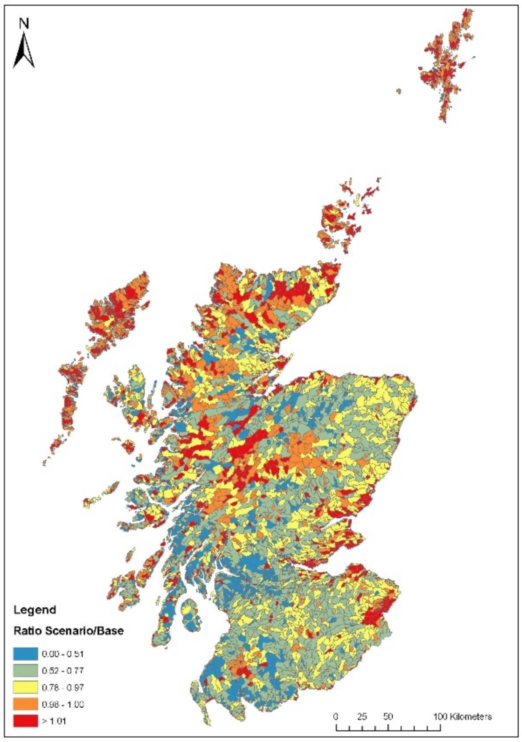


Figure 5.6. Nutrient export b22/baseline

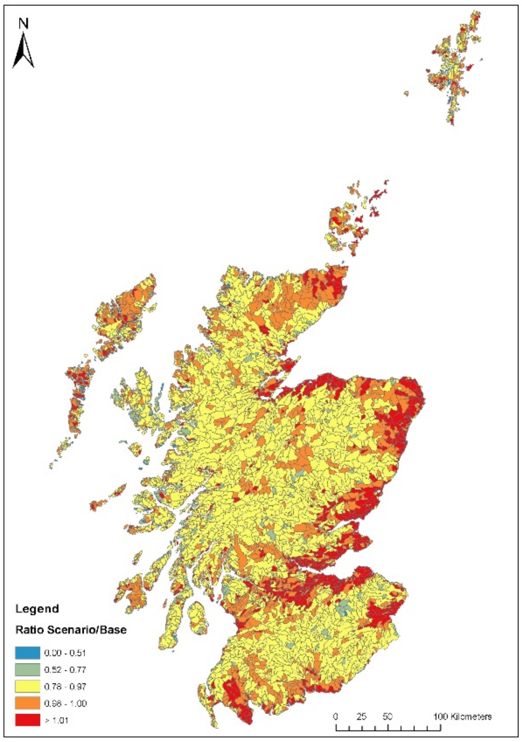


Figure 5.7. Nutrient export m13/baseline

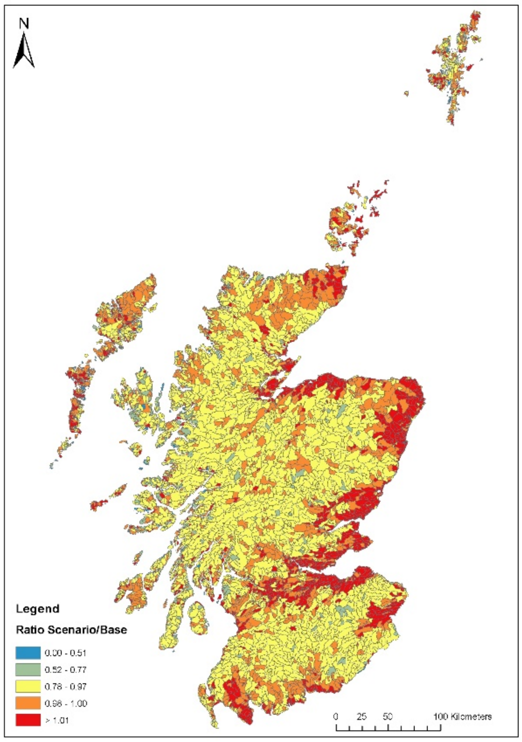


Figure 5.8. Nutrient export m14/baseline

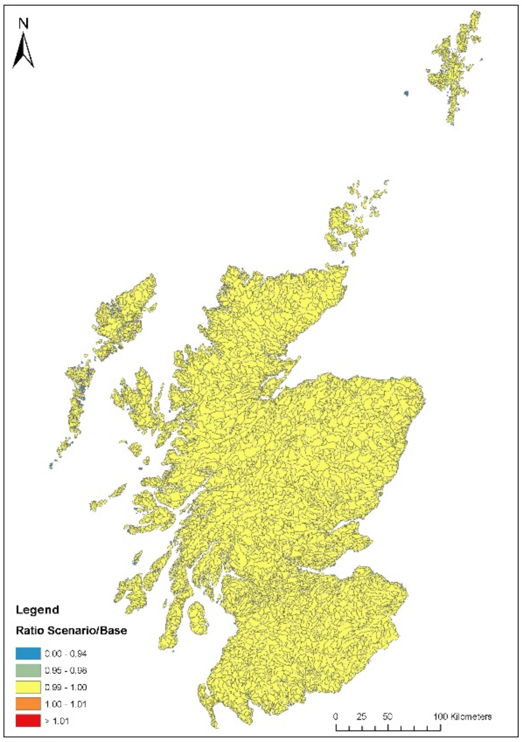


Figure 5.9. Sediment export b21/baseline

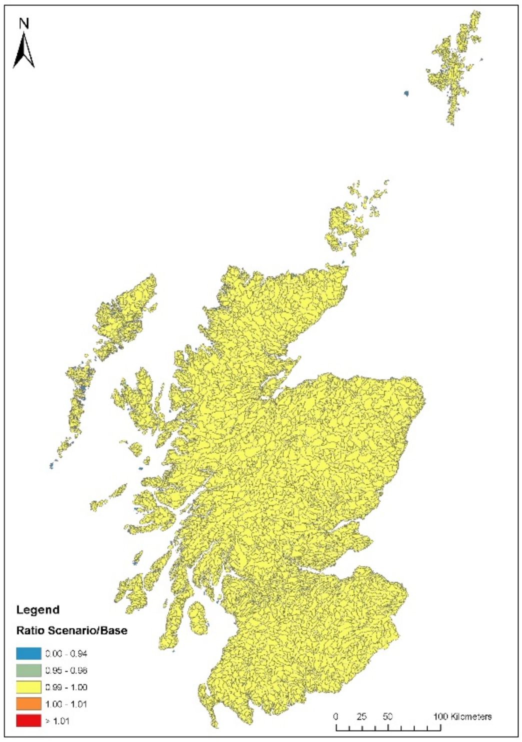


Figure 5.10. Sediment export b22/baseline

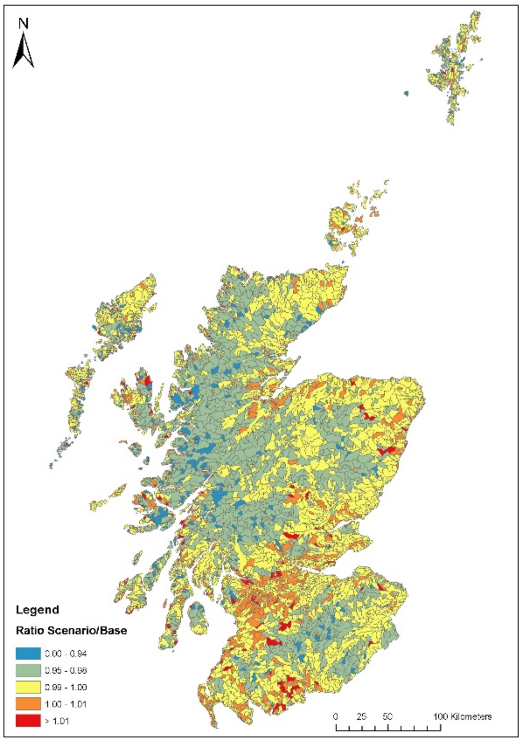


Figure 5.11. Sediment export m13/baseline

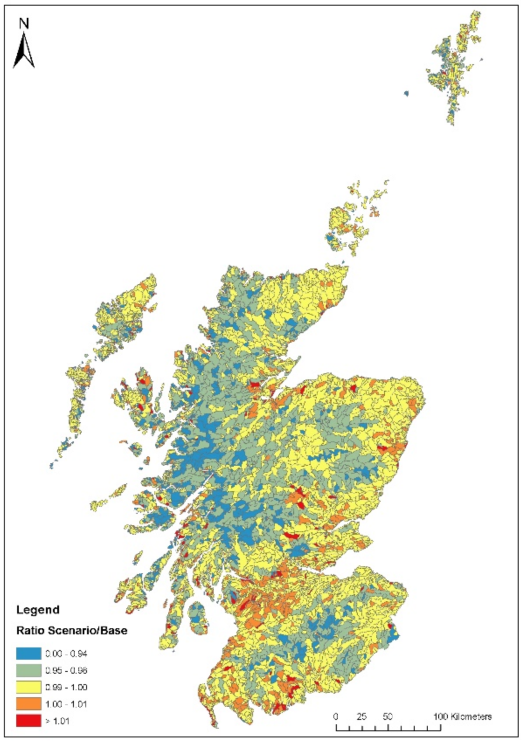


Figure 5.12. Sediment export m14/baseline

However, the spatial detail is important. It is remarkable that, in lowlands agricultural catchments on the east coast, protecting prime land implies protecting both arable and intensive grasslands. Despite planting riparian buffers, therefore nitrogen export does not decrease appreciably in many intensive agricultural catchments. In WP and WP&F, as well as some arable land, seminatural vegetation such as moorland is replaced by woodland.

The overall effect in nutrient export is modest, unlike in the Biodiv scenario, where corridors for wildlife habitat connectivity are placed on intensive grasslands. In this case there is a decrease in nitrogen export up to 20 percentage points higher than is obtained by protecting the grasslands and planting on seminatural vegetation and on riparian buffers.

For sediment export, which depends on soil erosion, there is no substantial increase in benefit in the biodiversity scenario, which targets intensive grasslands in relative flat areas, while the increase in benefits for the “Water&Prime”, and “Water&Prime Future”, scenarios are clear in the uplands, with maximum benefits on the west coast where rainfall is higher and slopes steeper than in the east of the country.

5.4 Conclusions

Although these results are specific to the landscape analysed, since they are based on the particular hydrology and nutrient dynamics of the catchment used for comparison, they give confidence in the use of InVEST for such decisions. Analysis of a decision context where absolute values of N exports were of importance suggested that the large uncertainties in InVEST outputs could dramatically impact the decision. Simple models like InVEST, initially developed for screening of the landscape and the integration of multiple objectives, are being increasingly used for ES assessments. This study provides a valuable test revealing model limitations, and this is known to be important for decision makers (e.g. Ruckelshaus et al., 2013). Further efforts in understanding these models strengths and weaknesses as well as how to calibrate them in places where more data are available are thus necessary for their application in the future.

The study illustrates how simple analyses can help identify the degree of information provided by simple models, hence the type of decisions they may inform. Because uncertainty affects most phases of most ecosystem service modelling exercises, from data gathering to parametrisation, practitioners relying on process-based models should be aware of it, and assess the consequences of under and over estimation, as well as how they may propagate to any economic analysis. Decisions that rely on estimates of impacts on ecosystems, therefore, also need to account for the degree of risk aversion of the decision makers.

Finally, scenarios illustrated above provide evidence for the potential to achieve multiple objectives through targeted land use change.

Improvement of habitat connectivity at the landscape level can be associated with a reduction in nitrogen export to streams from agricultural catchments, if stepping-stone patches of new forest are created preferentially on intensive grasslands. These same woodland stepping stones and corridors, however, provide little extra protection form soil erosion. The highest erosion avoidance through woodland planting is achieved on steep terrain in high rainfall areas.

Chapter 6 – Defining afforestation suitability targeting runoff catchment areas for Natural Flood Management

7

6.1 Introduction

This chapter analyses the opportunity to create woodland within the catchment to reduce the level of risk on sensitive receptors located downstream the Cairngorms National Park (CNP) enhancing the role of the protected area to contribute the regulating function of the system.

Climate change projections suggest that the frequency and severity of flooding is likely to increase over the next century (IPCC, 2014).

In Scotland, taking 2003 as a baseline, annual average damages for inland properties attributable to flooding is estimated up to £185 million (Werrity and Chatterton, 2004) and is likely to increase accordingly with the trend effect of climate changes. With potential costs of flooding increasing, Scotland has made great effort since the Flood Prevention (Scotland) Act 1961, when providing protection was delivered to less than 10% of the 77,191 properties estimated to be at risk from inland flooding (Werritty et al. 2002). Scotland in fact has led the rest of the UK in developing Sustainable Drainage Systems (SUDS). Together with the Water Environment and Water Services (Scotland) Act 2003 (WEWS), the plan introduced a general obligation on Scottish Ministers, Scottish Environmental Protection Agency (SEPA) and responsible authorities to promote sustainable flood management in the discharge of their relevant functions. With this renewed interest in flood risk management, "sustainable flood management" is tackled within the River Basin Management Plans required under the EU Water Framework Directive (Directive, W.F., 2000).

Traditional flood engineering methods, such as the installation of flood walls and its progressively increasing their height to contain increasingly serious events, have been found to be unsustainable over the long term, therefore an alternative approach to flood risk management is needed. An integrated catchment management approach is required to have effective benefits as wide as poverty alleviation (Warner, 2006; Kerr, 2002) sustainable development (Pollard et al., 2008;

Walmsley et al., 2001), access to energy (Falkenmark et al., 2002), healthy ecosystems (Stosch et al., 2017), thriving livelihoods and gender equality (German et al., 2006), and a strategic plan that manages both land and water through the system, recognising that activities can influence flooding elsewhere (SEPA, 2015).

Natural Flood Management (NFM) is the synthesis of this integration at catchment scale involving the work with natural features and processes to manage the sources and pathways of floodwaters (Wilkinson et al., 2019). NFM typically comprises restoring the natural capacity of a catchment to slow or store floodwater and covers a wide spectrum of measures. Those aim to reduce flood hazard, while also sustaining or enhancing other potentially significant co-benefits including enhanced ecosystem services (aquatic, riparian and terrestrial) such as greater biodiversity, improved soil and water quality, carbon sequestration, reduced soil erosion, greater agricultural productivity and improved public health and well-being (Dadson Simon et al., 2017).

As no strategy can completely eliminate flood risk, NFM measures are focussed on managing flooding within the catchment. A key component of this is combining short-term solutions, such as identification of areas prone to flood in order to decrease flood risk elsewhere (Wentworth, 2014) with mitigation plans as woodland creation in key locations for intercepting and 'soaking-up' surface run-off generated from the adjacent ground (Nisbet et al., 2006)

The desired effect of the implementation of NFM measures is to reduce the downstream flood peak (maximum height of a flood) and/or delay and elongate the flood peak downstream. NFM measures can decrease the quickflow volume of water entering the watercourse, reducing the scale and therefore impact of the flood and can increase the amount of time to prepare for a flood event (Wentworth, 2011).

Measure Group	**Measure Type**	**Main Action**
Woodland Creation	Catchment woodlands	Runoff reduction
	Floodplain woodlands	Runoff reduction/ Floodplain storage
	Riparian woodlands	Runoff reduction/ Floodplain storage
Land Management	Land and soil management practices	Runoff reduction
	Agricultural and upland drain modifications (e.g., grip blocking)	Runoff reduction
	Non-floodplain wetlands	Runoff reduction
	Overland sediment traps	Runoff reduction/ Floodplain storage
River and Floodplain Management	River bank restoration	Sediment management
	River morphology and floodplain restoration (e.g., re-meandering, floodplain reconnection)	Floodplain storage/ sediment management
	Instream structures (e.g., woody debris)	Floodplain storage
	Washlands and offline storage ponds (e.g., leaky dams)	Floodplain storage

Table 6.1: River and catchment based NFM measures (adapted from Natural Flood Management Handbook, SEPA, 2015)

General methods of spatial analysis in flooding have been considered, including mathematical, and probabilistic methods. However, at strategical level, we decided to use a practical approach leaving the analysis module developed for use within a spatial GIS-based MCA tool. In this module, the influence of the receptors on flooding can be visually explored using an indicator-based method. The indicator-based method provides a pragmatic approach to communicating areas of suitability to woodland expansion. An application example in the upper Dee catchment, north-east Scotland, is used to illustrate the capability of the spatial analysis module when applied in a decision-making context.

Traditional techniques for designing flood estimation use historical rainfall-runoff data. Such techniques have been widely applied to define flood prone areas at the sites of gauged catchment. Ungauged catchments are characterised by inadequate records, in terms of both data quantity and quality, of hydrological observations to enable computation of hydrological variables of interest for practical applications. Hydrological variables refer to evaporation, infiltration capacity, rainfall, runoff, and sub-surface flow. In this poorly condition of resource of data, the interpretation of rainfall-runoff relationships relies on one of the most important parameters to be able to predict the response of a watershed: the time of concentration (Tc).

The conceptual definition for Tc is the time it takes for a water parcel to travel from the hydraulically most distal part of the watershed to the outlet or reference point downstream. This definition has been used for many hydrologic studies and applications (Kirpich, 1940; Guermond 2008; Giandotti, 1934; Li and Chibber, 2008; Mark and Marek, 2011; Efstratiadis et al.2013).

The index Tc is generally estimated by using empirical formulas at catchment level, for example, Giannotti's formula (Giandotti, 1934) is extensively used in Italy, while Kirpich’s (Kirpich, 1940) formulas is widely adopted in the USA. In the paradox, Grimaldi et al., 2012, highlights how these formulas are very well accepted in the applied hydrology community, with limited information on their technical foundations. Whether Tc is considered to be quasi-invariant for high return period events in respect of rainfall intensity the risk is to fall into overestimation of Tc. However, strategic applications like the one we propose in this document are not affected by this risk because the aim is more to define the areas where the most of the runoff is generated rather than calibrate the index with rainfall magnitude.

The objective of the present study is in fact to identify areas which are considered to be the sources of the runoff reaching sensitive human and structural receptors. We decide to apply Geographical Information System (GIS) to carry out the analysis undertaken in 2018/2019 in consultation with Scottish Environmental Protection Agency (SEPA) and the Cairngorms National Park (CNP). These areas would be defined as targeted by woodland expansion because the trees can play an important role by slowing the flow and increasing the ‘sponge effect’ allowing water infiltration through the root ways.

6.2 Flood risk in the Cairngorms National Park

All of the rivers and watercourses within the Cairngorms National Park have the potential to flood to some degree. Most concern is generated along the Park’s main straths and glens, as when the rivers and tributaries that flow along these, namely the Spey, Dee, Don and Tay, break their banks they often result in economic and occasionally human cost. Small watercourses also represent a risk but are often poorly understood with respect to the severity of the flood hazard that can be generated on a catchment scale. A summary of the most significant flooding risks and hazards within the Cairngorms National Park is provided below. Information obtained from the Local Development Plan 2020 (LDP, 2020) provide sufficient details of flood risk and impact, along with information on historical flooding, for each local catchments identified by SEPA where significant risk exists now or is likely to occur in the future, namely Potentially Vulnerable Areas (PVAs).

6.2.1 River Spey

The River Spey rises in the high ground of the Monadhliath and Cairngorm Mountain ranges and flows in a north-easterly direction through narrow straths and scenic river valleys before discharging into the Moray Firth beyond the fertile farmlands of Morayshire. The upper part of the catchment is characterised by its mountainous areas, the highest point being the summit of Ben Macdui at 1,309 metres above sea level.

The River Spey is the seventh largest river in Britain, with a catchment area of over 3,000 km^2, and a stream network length of about 36,500 km, of which the main river comprises 157 km (Spey Catchment Steering Group, 2003).

There is a long history of flooding within the Spey catchment area, with a notable event, known as the Great Muckle Spate, destroying several bridges in 1829. The River Spey and its tributaries continue to flood regularly, with heavy rains and melting snows increasing the volumes of water in the catchment. These floods have damaged properties in Newtonmore, Aviemore and Carrbridge on a number of occasions. Most recently in 2014, Gynack Burn broke its banks in Kingussie, damaging local buildings and infrastructure (SEPA, 2015).

Due to the potential risk caused by flooding within the catchment area, five PVAs (Fig. 6.1) have been identified within the National Park, namely Aviemore and Boat of Garten, Carrbridge, Kingussie, Newtonmore and Dalwhinnie.

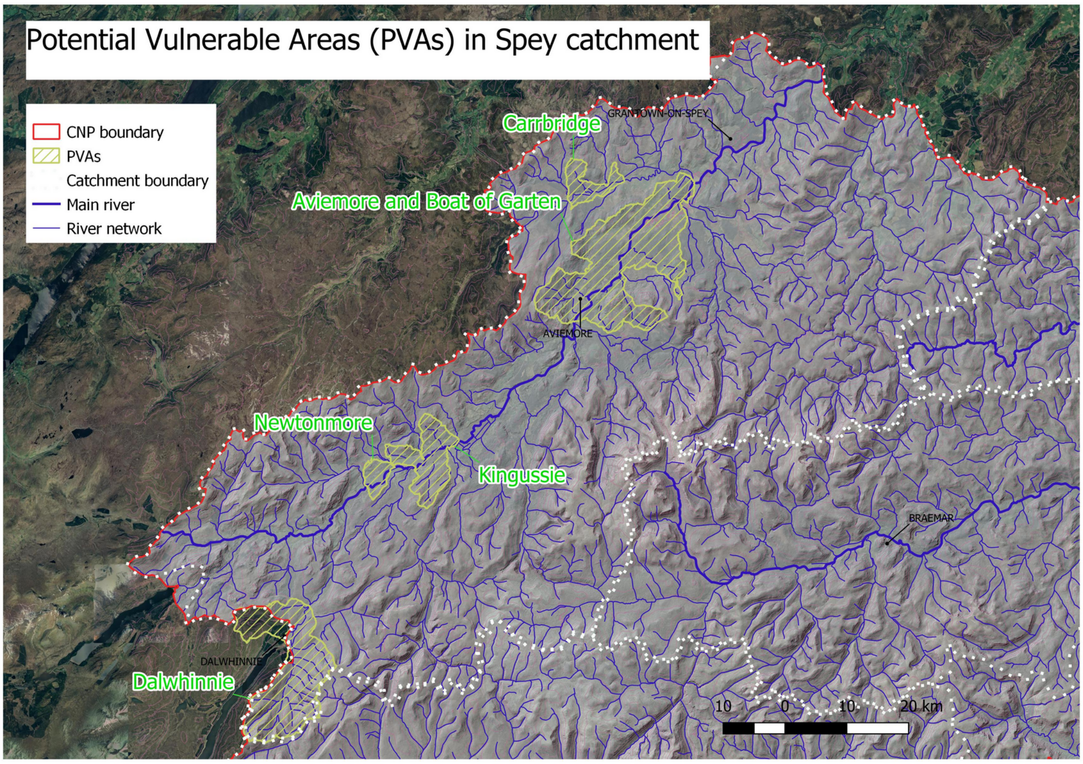


Figure 6.1 Distribution of PVAs in river Spey catchment.

6.2.2 River Dee

The River Dee rises in the Cairngorms Mountains east of Braemar on the semi-arctic Braeriach-Cairn Toul plateau. For the majority of its course, the river flows eastwards through a broadening valley, which becomes much gentler in relief as it leaves the National Park. Within the National Park, the river is fed by a number of important tributaries, namely the Lui, Clunie, Gairn, Muick and Tanar, the latter's confluence located just outwith the National Park Boundary (Dee Catchment Partnership, 2007).

The river is considered to be the best example of a natural highland river in Scotland (Maitland, 1985). The notable characteristics of the river include its great altitudinal range, its unique succession of plant communities, and its steep profile compared to other large British rivers (Dee Catchment Partnership, 2007).

Like the Spey, the Dee suffers from flooding related to heavy rain and melting snows. Major floods have been recorded in 1769, 1829 (the Great Muckle Spate), 1920 and 1956 (the Cairngorm Flood) (Dee Catchment Partnership, 2007). In 2008 surface run-off entered the Netherly Guesthouse in

Ballater and in 2014 the town's caravan park and a number of roads were closed due to flooding (SEPA, 2015). More recently, in December 2015 / January 2016, the Dee experienced widespread flooding, which caused significant damage to property and transport infrastructure.

The Dee catchment contains two PVAs (Fig. 6.2) that fall within or across the National Park boundary, namely Ballater and Aboyne.

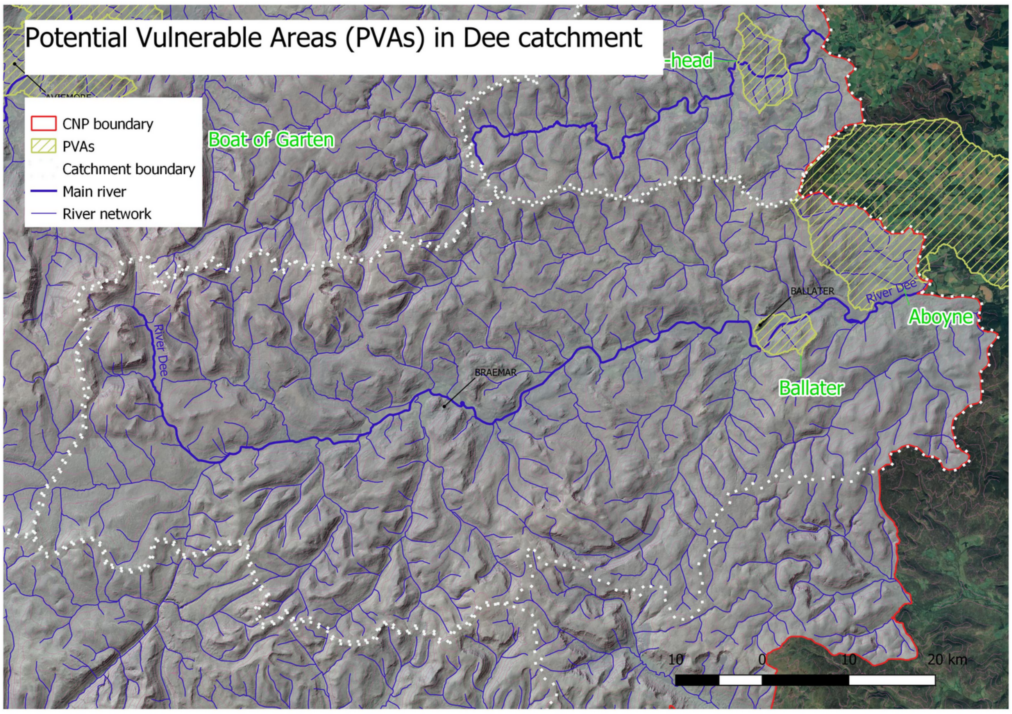


Figure 6.2 Distribution of PVAs in river Dee catchment.

6.2.3 River Don

Rising in the in the peat flat beneath Druim na Feithe, and in the shadow of Glen Avon, the River Don flows 135km east to the sea in Aberdeen. It is Scotland's 6th largest river, draining a catchment of around 1,300km^2.

The Don catchment contains one PVA (Fig. 6.3) that intersects the National Park boundary, namely Heugh-head.

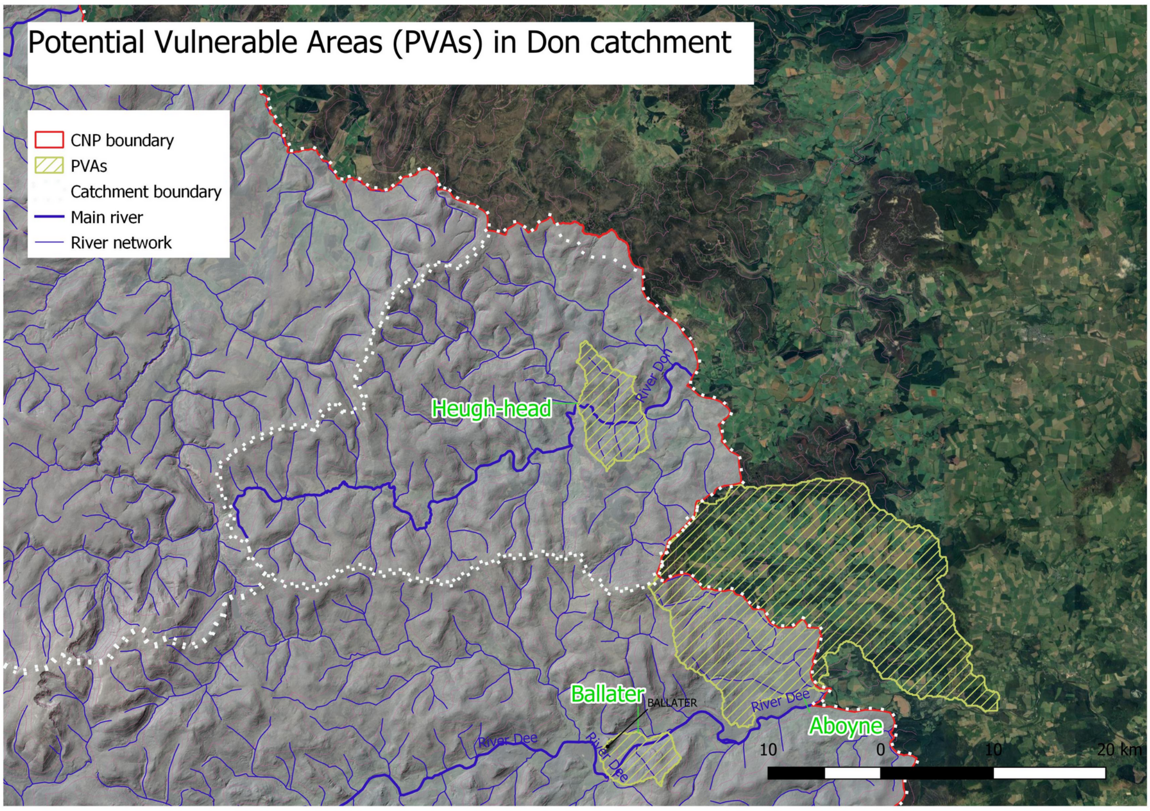


Figure 6.3 Distribution of PVAs in river Don catchment.

6.2.4 River Tay

The River Tay has the largest catchment area and is the longest river in Scotland, with many of its headwaters lying within the Cairngorms National Park. It covers an area of 5,088km^2 and is around 190km in length. More water flows through the River Tay than any other river in the United Kingdom. The main tributaries include the River Garry, River Tummel, River Lyon, River Braan, River Isla and River Almond.

The largest lochs in the River Tay catchment include Loch Ericht, Loch Rannoch and Loch Tay (SEPA, 2015). The Tay catchment contains one PVA (Fig. 6.4) that falls within the National Park boundary, namely Blair Atholl.

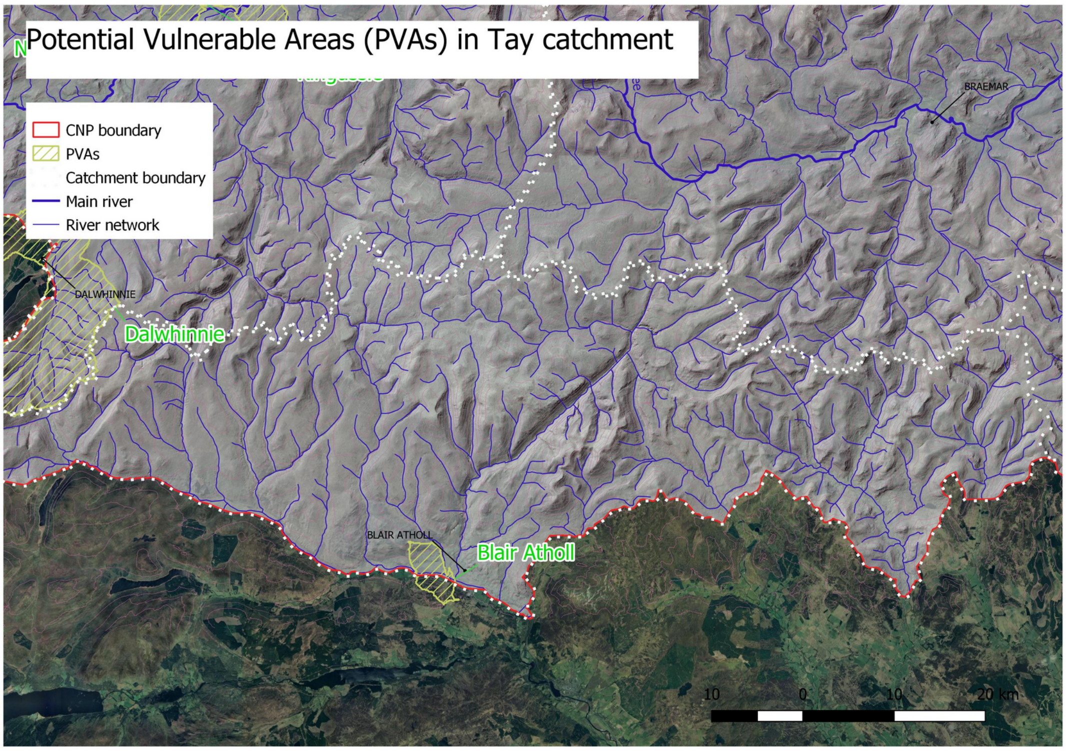


Figure 6.4 Distribution of PVAs in river Tay catchment.

6.3 Data used

The result is a complex development of multiple spatial analyses which were carried using in combination the free GIS-GRASS software under the GNU GPL license which can be obtained from http://grass.osgeo.org/ and two open-source software such as GRASS GIS and R-CRAN (R Development Core Team, 2015), an integrated suite of software facilities for data manipulation, calculation and graphical display free available from https://www.r-project.org/. The work required substantial data processing, because of the very detailed surface map used (10m digital terrain model) and over 1000 separate analyses were conducted. After consultation with relevant stakeholders, we upscaled the CNP analysis to the National level.

6.3.1 Digital Elevation Model

The Digital Elevation Model (DEM) was derived from OS Terrain® 5 (OS Terrain 5, 2018) with an original resolution of 5 m. The DEM was resampled to 10 × 10 m and the medians in each grid cell were used. The resampled digital terrain model was then clipped within the CNP borders.

Slope measure was derived with *r.slope.aspect* GRASS module based on RST perform simultaneous interpolation and computation of partial derivatives (Mitášová and Hofierka, 1993).

6.3.2 SEPA dataset

A nationally-applied methodology has been used by SEPA to produce the flood risk maps for Scotland. Those maps, which provide information on the indicative impacts of flooding at the community level, were used to identify the sensitive receptors and flagged accordingly with the flood risk correspondent (Tab. 4.2) as High, Medium, Low risk using the returning time of the similar flooding events (e.g. 10 years is very high risk).

SEPA holds the relevant information about the flood risk maps (https://www.sepa.org.uk/media/163413/impacts_of_flooding_summary.pdf) that we have received for either fluvial and pluvial risk along with the assets (residential and commercial properties, railways and roads, utilities, etc.) associated to each of the following levels of risk.

Likelihood of flooding	Fluvial and Coastal	Pluvial
High	10 year	10 year
Medium	200 year	200 year
Low	1000 year	200 year + climate change

Table 6.2 Flood hazard levels on different likelihoods of flooding. Source: SEPA.

For our goal we decided to select all receptors classified as High Risk (as the most sensitive for fluvial flooding risk (as the pluvial one originates from much smaller areas). In particular, the receptors (commercial services, roads, railways, utilities and residential properties) were used as the starting point for our analysis (Fig. 6.5). The very early-stage analysis was run just with a selection of the receptors located within the CNP area for practical reasons, however the upscaling process was run for the whole Scotland using the full receptors dataset. Results will highlight the different outcomes using a subset of this dataset.

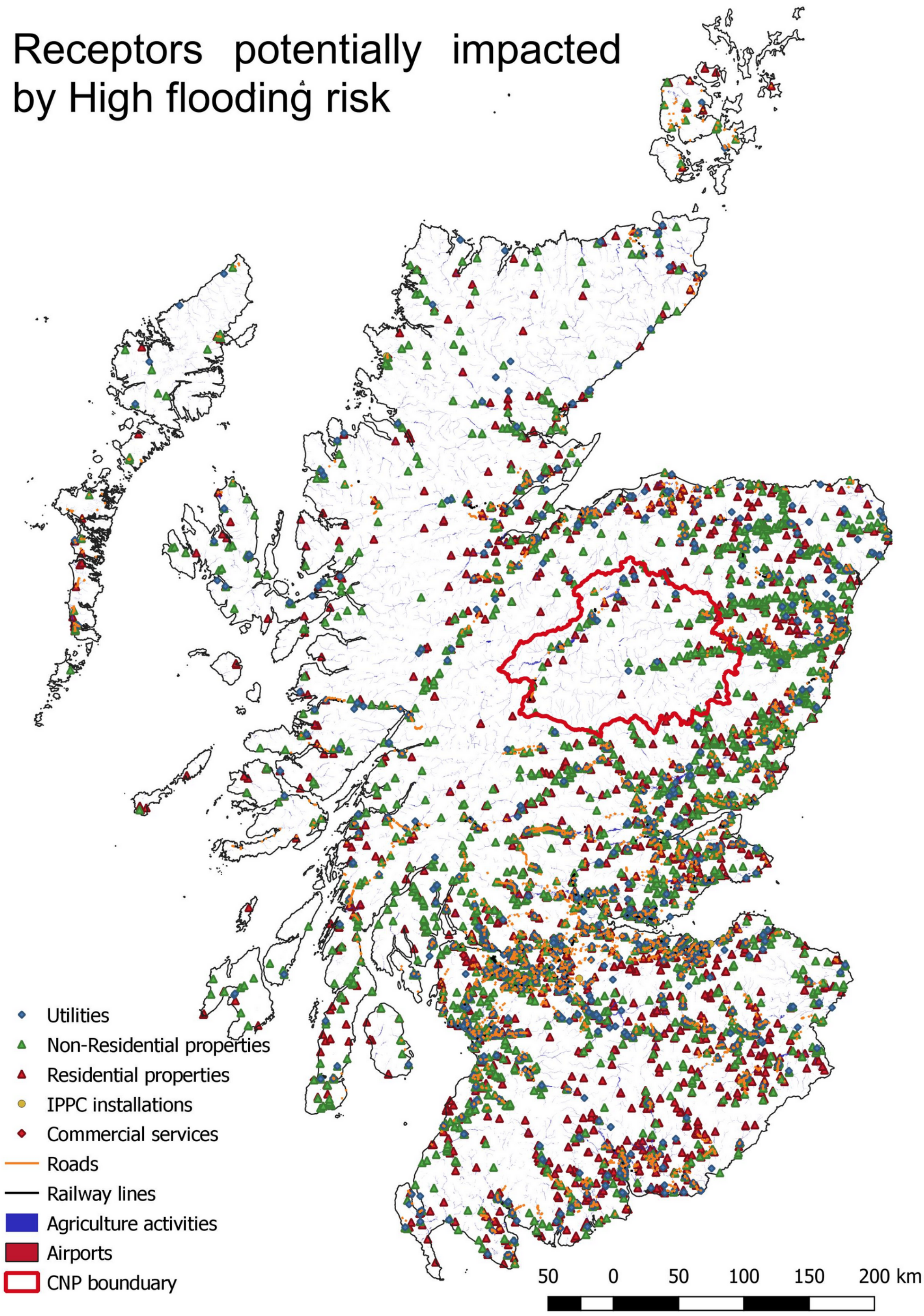


Figure 6.5 Assets identified by SEPA as potential high flooding risk receptors if located within a flood extent.

6.3.3 Soil data

The second and very important input data we used for these analyses was the (potential) Infiltration Capacity (IC) (Gagkas and Lilly, 2019).

The IC dataset was derived by horizontal and vertical distribution of soils and soil properties, and has a profound influence on catchment hydrology, in fact, soils can delay runoff, store and redistribute water and provide a supply of moisture for plant transpiration (Lilly et al., 2012).

Gagkas and Lilly, 2019 have developed a method for predicting the distribution of soil hydrological classes using spatial disaggregation of map units with the *Random Forest* algorithm as a means for downscaling the original dataset. It was determined that RF was effective at predicting the complex relationships between HOST classes and the set of environmental covariates used, so the method was applied nationally to generate maps (Fig. 6.6) of soil hydrological functions (IC) which, we used to improve environmental spatial modelling and risk assessments in flooding.

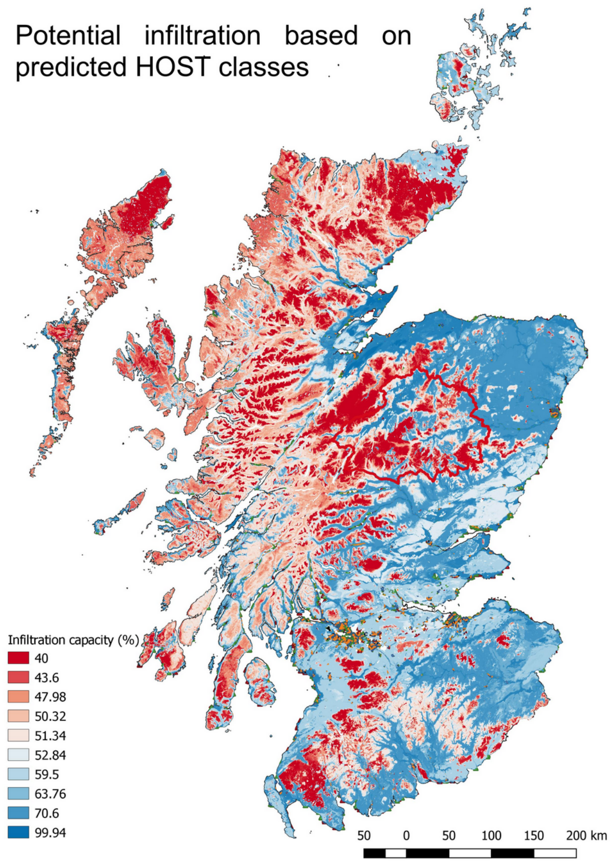


Figure 6.6 Potential infiltration capacity (modelled) based on HOST classes observations.

6.4 Methods

The following paragraph describes the methods to obtain a detailed representation of the natural flood management and where exactly prioritize attention to enhance the ecosystem services benefits through the use of long-term measures. Such restoration projects rely on the creation of new woodland within the catchment area to mitigate the effect of flooding.

Firstly, the DTM (10 m resolution) was used to derive the flow accumulation model and the drainage map which is a layer that represents the potential overland pathways due the setting of the surface considered. For the Cairngorms National Park area, after several sensitivity analyses and comparisons with the existing river network, we decided to set the threshold for the flow accumulation as 1000 cells pre-processing. This means a pixel x should have at least 1000 upstream cells that drain into that pixel before it would be considered a stream.

In the same time, the 10m DTM was used to derive the slope map for the study area.

Those data were used in an iterative process, a loop, which was run as many times as the number of the receptors present in the study areas. In particular, the iteration is defined as follows:

<u>For each receptor (point feature)</u>

- The upstream basin was reconstructed using the drainage map
- The area and the Concentration time (Tc) were calculated within the basin (Kirpich, 1940) according to the formula:

$$t_c = 0.0078\left(\frac{L^{0.77}}{S^{0.385}}\right)$$

Where L is the maximum distance from the outlet inside the basin, and S is the median slope value inside the basin.

Area values between all catchments were normalized (0-1) along with the inverse of Tc (1/Tc), the two values were then summed together to obtain a Runoff index (RI). Therefore, the RI is adding an extra factor on the barely prioritization of the watershed area extent constructed watershed upstream a receptor because they can direct a larger amount of rainfall. Within this spatial bottom down order, each watershed account for the capacity to deliver flash flooding events (1/Tc factor), which, in this work, is interpreted and associated to the theoretical definition of the Tc - the time that a drop of rainwater spends to arrive to the basin outlet section starting from the most hydraulically distant point of the basin (McCuen, 2009).

All the basins were summed together and normalized again to obtain the total RI map for all the study area. The resulting map describes the likelihood of some areas to favour the runoff and the source of potential flooding events. In addition, the total RI map was filtered using the two parameters:

- The areas where the modelled flow accumulation map is less than 1000 (pixels considered for creating surface routing).
- The areas where altitude (in m) is greater than 600

Those two spatial filters tend to prioritise the areas that are more distant from the receptor first because the creation of new woodland just above the asset would not give enough time to make a appreciable effect in the infiltration rate; in addition, we considered areas that are higher than 600m as not suitable for expansion because of the combination of low solar irradiance and high wind speeds that may reduce considerably the rate of tree growth.

The final NFM map is the results of the (rescaled 0-1) fluvial analysis using the receptors as described above filtered and combined (spatial sum) with the infiltration capacity (IC) map. This additional IC factor, was used to address new trees in areas that can both create the sponge effect and where the infiltration is currently low, so the creation of new woodland can increase this parameter through their roots.

The limited computational resources and therefore the time constraints to deliver this work for the whole Scotland, have driven the pragmatic choice to resample the fluvial source map at 25m using the median of the neighbour’s block and then clipped in the CNP area to remove potential border effect due the limitation of the receptors used.

6.5 Results and discussion

A map with the areas ranked by suitability is shown in Fig. 6.7.

The resulting map would be a useful support to the help top priorities of the political agenda such as whether woodland expansion in the Scotland could make a significant contribution to tackle the future rise in flood risk linked to climate change, as part of a whole-catchment approach to sustainable flood management. Although, there is evidence that woodland offers a number of potential opportunities for flood control from hydraulic modelling studies (Nisbet et al., 2005, Linsted et al., 2006, Calder et al., 1999), other research and experiences indicate that aspects like the increased water use of trees and the forest sponge effect are largely restricted to the headwater or to small catchment level (Nisbet & Thomas. 2006) while the cumulative effect doesn't propagate far downstream and to the larger watershed scale. For this reason, our resultsindicate greater scope for flood reduction and may reduce small floods but, generally, not extreme events.

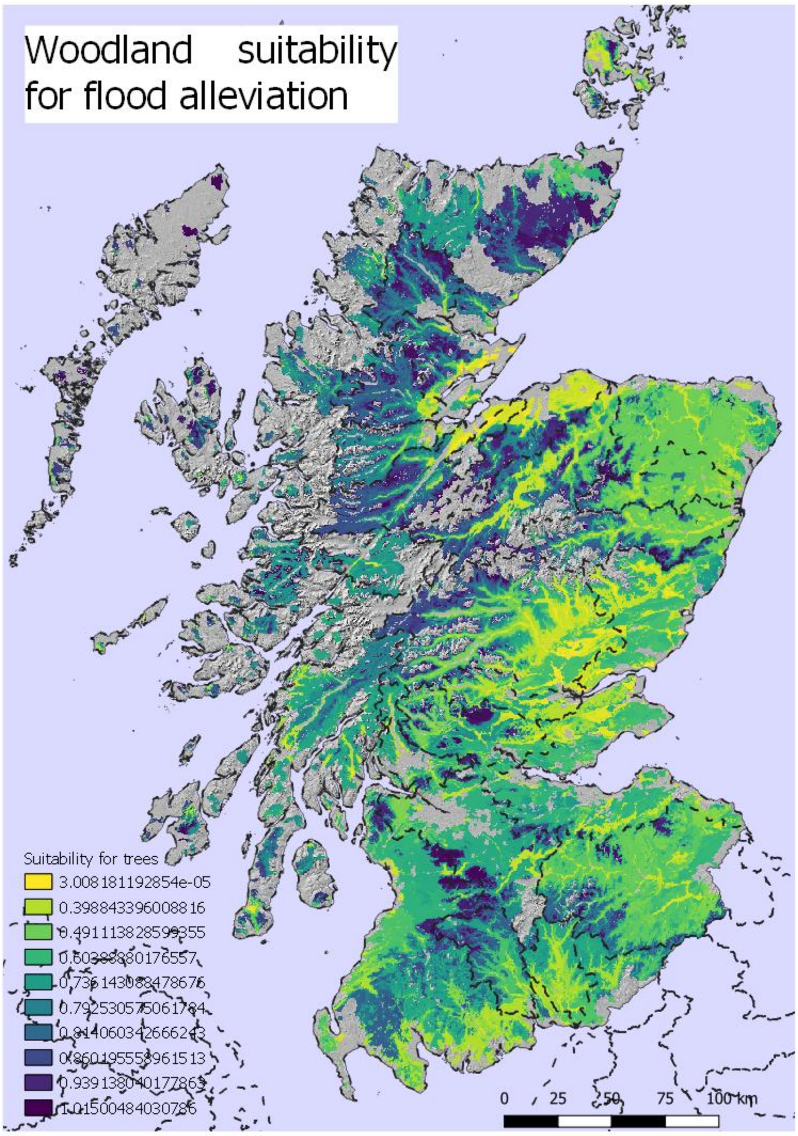


Figure 6.7 Suitability of woodland expansion for flood alleviation

At a very local scale the analysis conducted nationally acquires greater significance when it comes to preventing the high frequency events which identify the high-risk receptors. The CNP suitability map (Fig. 6.8) does not show where planting will have the greatest impact on flood management. However, the map does show where the opportunity for reducing run-off rates is the greatest.

Furthermore, having the catchment upstream the single receptors been treated as a uniform body, darker blue areas (higher scores) are clustered around the middle slope and in areas upstream of the PVAs, therefore likely to have a positive effect on reducing flooding downstream through implementation of new woodlands planting schemes.

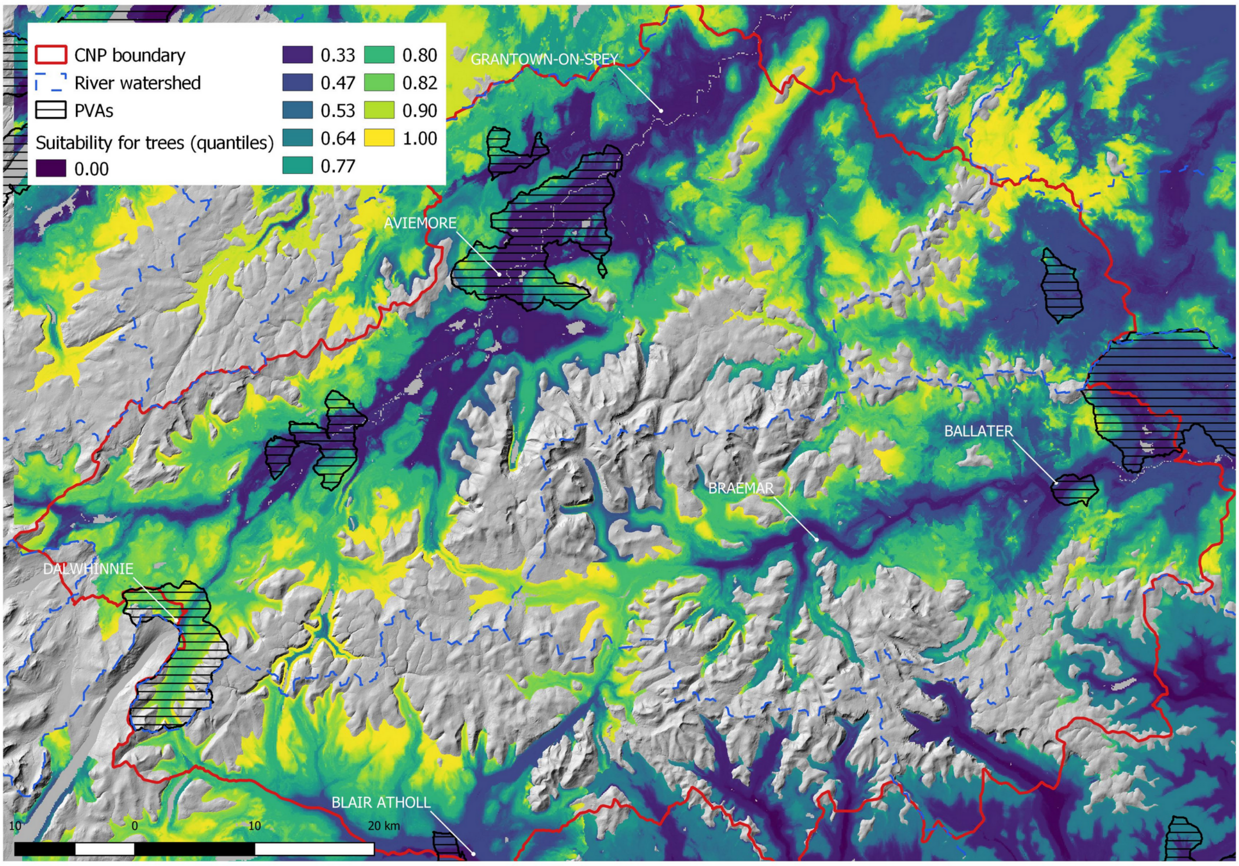


Figure 6.8 Suitability maps for woodland creation within the Cairngorms National Park area.

In addition, the results of our analysis identify in detail the priority areas for woodland expansion to alleviate flooding risk as the intermediate area between the headwaters and the upper part of the alluvium (flooding terrace), i.e., somewhere in the middle-lower part of the catchment slope. Recent studies (Hou et al., 2018), on modelled small catchments accounting for different rainfall events and different configurations of land use (scenarios of new woodland), seems to support our findings.

The NFM map obtained will be used as\ input data for the sMCA analysis in the natural protected area and will be represent one of the criteria to address the multifunctionality and the potential of the new forest.

Using this analysis, we are quite confident that we can identify:

- The areas where most of the runoff has been produced;
- The basins to prioritize for NFM;
- How the woodland expansion target in the CNP can help to reduce the small flooding event and increase the resilience of the potential vulnerable areas (PVAs).

A caveat is tree planting is unlikely to mitigate large and extreme rainfall events.

Chapter 7 – Modelling native broadleaved and conifer dispersal avenues through the use of circuit theory model

7.1 Introduction

Habitat fragmentation is considered one of the most severe threats to global biodiversity (Sala et al., 2000; Foley et al., 2005), and birds and mammals are recognized to be seriously affected (Andrén, 1994; Recher, 1999), along with amphibians (Stuart et al., 2004), plants (Hobbs & Yates, 2003) and invertebrates (Didham et al., 1996). The mitigation of ecosystem fragmentation is important in new targets of the European Biodiversity strategy to 2020 (Estreguil at al., 2013). Understanding how this mitigation affects the habitat availability and connectivity of species at the landscape scale is important for effective conservation planning (Uezu et al. 2005; Laita et al. 2011; Helm 2015). While the paradigm of the species distribution known as "landscape connectivity", assumed as the movement of organism among habitat patches, can be well interpreted by everybody, the functional relationship of organisms to landscape structure might not be so straightforward to understand. There are examples in nature, where connected habitat patches still may not be functionally connected for some organisms and even non-contiguous habitat patches may be functionally connected for others (With et al., 1997). In the forest context, if two woodland habitat in a landscape are connected by a corridor well structured which do not successfully deliver the functional response for the species results will be insufficient for species movement (Tischendorf, et al., 2000). Therefore is important, rather than focus on a single organism, use a multispecies approach (Adriansen et al., 2003) when considering connectivity. Dispersal movement of organism and terrestrial species relied for long time entirely on the use of least-cost method (Knaapen et al., 1992), its evolutions (Yu, 1996; Myers et al., 2000; Calabrese et al., 2004; Rothley et al., 2005) and multiple applications (Gustafson et al., 1996; Adriansen et al., 2003; Bunn et al., 2000). Even thougth parametrization of the cost surface is a difficult process, least-cost analysis are increasingly being coupled with graph-theoretic techniques thanks to the expanding accessibility of GIS software that can compute rapidly the least-cost routes. Recent studies (Urban et al., 2009; Urban and Keitt, 2001) successfully emphasised analyses based on graph theory for modelling the functional response of a target species (or group of species) to landscape pattern (patch size, shape, location). Graph theory and graph-based models increasingly become appealing because their spatial representation that can be visualized and further studied in relation to land use changes, and can solve important issue related to population stability and resilience (Urban et al., 2009). Ecological literature generally considers the nearest neighbouring patch, or patches within a limited neighborhood of the focal patch (a buffer) as the measures to return significant effects (Kindlmann, 2008), or, alternatively graph networks (Dale at al., 2010) even though the limitations of such approaches has been recognized (Moilanen, 2010). In these models (Fig. 7.1), patches of habitat (nodes) are distinguished from the matrix. The connections among nodes, called links (or edges) suggest the potential for movement or dispersal of a focal species. Finally, graph-based metrics of connectivity can also be used when simulations are impossible or impractical (D'Eon et al., 2002) and do not require the services of a skilled programmer.

Examination of species distribution patterns using modern methods of spatial analysis can help us to better understand the effects of ecological, environmental and anthropogenic pressures. Those are usually the trigger to significant landscape changes which represent the challenges to interpret relative importance among habitat removal and increased barriers to movement that can reduce gene flow (Cushman et al., 2006) and promotes species invasions (Real and Biek, 2007).

In recent years, a new approach to understanding habitat connectivity has been borrowed from studies of gene flow in plants (McRae et al., 2007) to envisage the landscape as an electrical circuit, with each cell in a raster grid presenting a given ‘‘resistance’’ to movement of the modelled organisms (McRae et al., 2008).

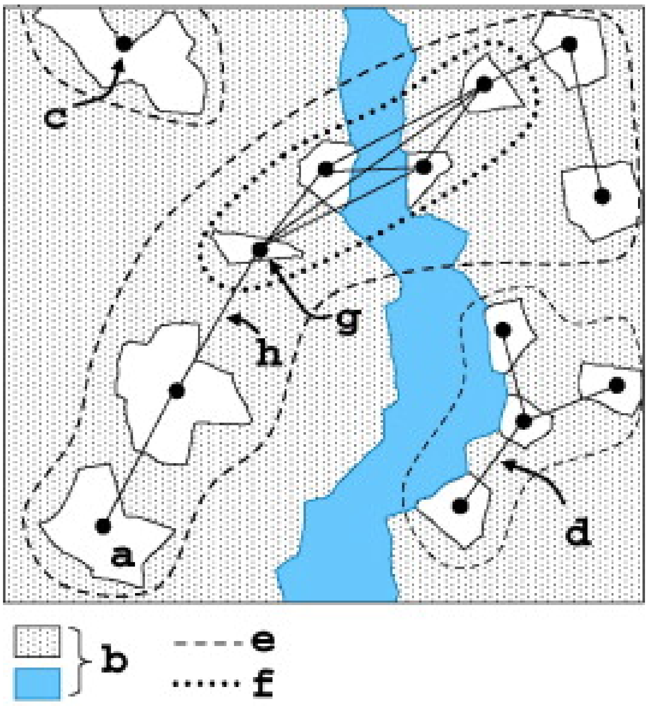


Fig. 7.1. From Galpern et al., 2011. Illustration of key terms in patch-based graphs. (a) Patch – the focal habitat on the landscape. (b) Matrix – the landscape excluding the patches, shown here as a stippled surface and a solid-coloured river. (c) Node – the graph element used to represent the patch. (d) Link – the graph element used to represent the connectivity relationship between patches. (e) Components – groups of nodes connected by links. (f) Compartment – a group of nodes identified according to some criterion; this compartment has been identified based on the density of links among nodes, and is part of a larger component. (g) Cut-node – a node which, if removed, would disconnect a component. (h) Cut-link – a link which, if removed, would disconnect a component.

Here we compare simple connectivity measures in their ability to predict colonization events in two large and good-quality empirical data sets using Circuitscape software (McRae & Shah, 2009). Circuitscape has been applied with success in modelling the effects of animal movement at large scale for strategy planning and at finer time scales for management, where species connectivity represents a primary concern (Scoble et al., 2010, Spear et al., 2010). Circuit theory progresses further the graph theory-based methods through the inclusion of two main ecological concepts: the least cost path, which is often used to extract ecological corridors (Adriansen el al., 2003) and random walk of species (Chandra et al., 1997) which helps to interpret the key nodes of the corridors. Beside, Circuitscape, which was used in the last decade in ecology (Carroll et al., 2017; Dilts et al., 2016. Proctor et al., 2015) and in defining priority areas (Koen et al., 2014; Breckheimer et al., 2014, Gimona et al., 2012), accounts for multiple dispersal pathways which allows the user to evaluate the degree of redundancy (Gimona et al., 2012).

Within the Cairngorms National Park Forestry Strategy 2018 (CNPA, 2018) the third objective is to restore lost or vulnerable forest ecosystems through creating forest habitat networks. Woodlands in the Cairngorms National Park are vitally important for wildlife and support some of the most charismatic creatures such as red squirrels, ospreys, capercaillie, and the Scottish crossbill, the

UK's only endemic bird. Prior to widespread forest clearance by humans in the medieval period Pinewood Caledonian forest covered most of the Scotland area offering food and shelter to many of these species and is now shrink to 1% of its original size, many fragments of which are now included within the CNP area where is still fragmented between the sub catchments (Futurescapes). Here Circuitscape model has been applied to create opportunity maps that can illustrate dispersal flow paths and variations in the difficulty for "walkers" across the study area. To address specific species that are recognised pinewood specialists we distinguish the analysis for broadleaved and conifer for connecting isolated fragments of forest with same composition that is essential for both habitat and wildlife resilience.

7.2 Methods and materials

Analysis was applied to the whole of CNP. The landscape where forest is embedded is quite diversified with relatively small broadleaved woodlands on the margins of the CNP, which are quite fragmented and dispersed in a farmland matrix, while on the foothills of the upland deciduous forest tends to be surrounded by conifer plantations and/or heather moorlands. At a broader scale the river Spey is dominated by native and non native conifer while the river Dee is more broadleaved oriented, however the valley are of course separated by upland areas often devoid of trees (Fig. 7.2). Circuit theory (McRae and Beier, 2007; McRae et al., 2008) was used in this to identify the ecological corridors in heterogeneous landscapes. In the circuit model, landscapes has been interpreted as a conductive surfaces while woodland patches represent source current zones. The system overtakes the restrictive assumption of the least-cost path theory (Larkin et al., 2004, Kautz et al., 2006, Watts et al., 2010) using an electrical current, flow and voltages analogy, related to ecological processes.

7.2.1 Land cover data

The land cover data were derived from the Land Cover Map 2007 (LCM2007), produced by the UK Centre for Ecology and Hydrology (CEH, Morton et al., 2011).

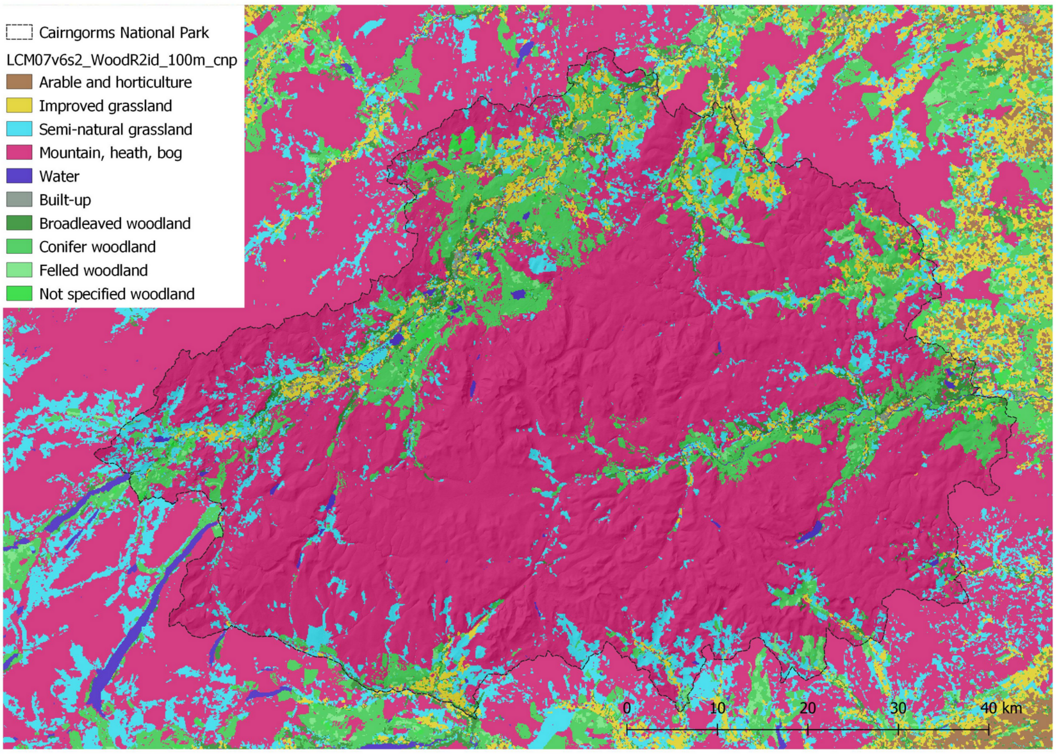


Figure 7.2. Land cover classes (aggregated) form LCM2007.

This dataset, compared with the digital land cover classification for Scotland (LCS88) (MLURI, 1988) dramatically improves spatial and thematic accuracy and better represents landscape objects. The GB framework is based on Ordnance Survey Master Map topography layer (hereafter referred to as OSMM) and the NI framework is based on cartographic data from Ordnance Survey Northern Ireland (now part of Land & Property Services). The spatial framework has been further refined by supplementing the generalised national cartography with agricultural census data boundaries and image segments. LCM2007 is the first land cover map to provide continuous vector coverage of UK Broad Habitats derived from satellite data (Morton et al., 2011). An integrated LCM2007 with Forestry Commission woodland inventory (namely LCM07v6s2_WoodR2n3 see Technical report Hewitt et al., 2018 in appendix G) were created in January 2017 by Marie Castellazzi at the JHI, with the aim of improving the representation of woodland in the LCM2007 land cover map. The integrated dataset incorporated the latest version (as of January 2017) of 3 Forestry Commission datasets:

- Native Woodlands Survey for Scotland 2014 (NWSS),

- National Forestry Inventory for Scotland 2015 (NFIS),
- National Forest Estate Legal Boundary for Scotland 2016.

Map is shown in Figure 7.2.

7.2.2 Definition of matrix resistance for broadleaved

The matrix resistance parameters for species dispersal were derived from Gimona et al., 2012, who derived the parameters from Watt et al, 2010. The Delphi analysis (Crance, 1987) used a group of experts from UK conservation agencies, universities and consultancies, who were asked to rank landscape resistance to dispersal for a ‘generic focal species’ (GFS). Resistance values were match with the corresponding land cover classes in LCM 2007 (see Table 1). GFS is defined by the ecological context and requirements reflecting the most probable needs of real species. This generalization is useful when single species data are not available. However, Gimona et al., 2012 modified the original parameters accounting for the degree of modification and vertical structure of each land cover.

7.2.3 Definition of matrix resistance for conifer

Matrix resistance values for conifers were obtained from CaperMap which is a spatial modelling results made in-house at the James Hutton Institute (JHI) under Objective 1.3.4 of the RD1.4.1 RESAS project 2015-2021 of the Sottish Governament (RESAS, 2015). Caper Maps (Fig. 7.3) are the level of disturbance in woodland for protected species “Capercallie” (Tetrao urogallus) habitats considering core paths, towns and roads.

The maps shows the probability of occurrence of Capercallie (suitability) and the core areas (zones) of living and nesting. Both maps were developed by Jim McLead at JHI using MELODIC (Gimona et al., 2016) tool.

The CaperMap model was run for the extent of CNP (+10km buffer zone) at 25m cell size for 4 scenarios. For purpose of this analysis we uses Scenario 3 (resampled to 100m resolution) that represents the highest possible level of disturbance (Fig. 7.3) on the basis of the Caledonian Pinewood Inventory (CPI, Jones, 1999) dataset.

- Baseline (all conifer woodlands, core paths, towns, roads, etc.)
- Scenario 1 Baseline + All Paths network
- Scenario 2 Baseline + CPI Regeneration Zones all converted to Caledonian Pinewood to enhance existing cover
- Scenario 3 Baseline + All Paths + CPI Regeneration Zones

Zonation of the caper maps has integer values categorised with suitability into 0=Unsuitable (value < 0.5), 1=Suitable (0.5 <= value < 0.8), 2=Excellent (0.8 <= value).

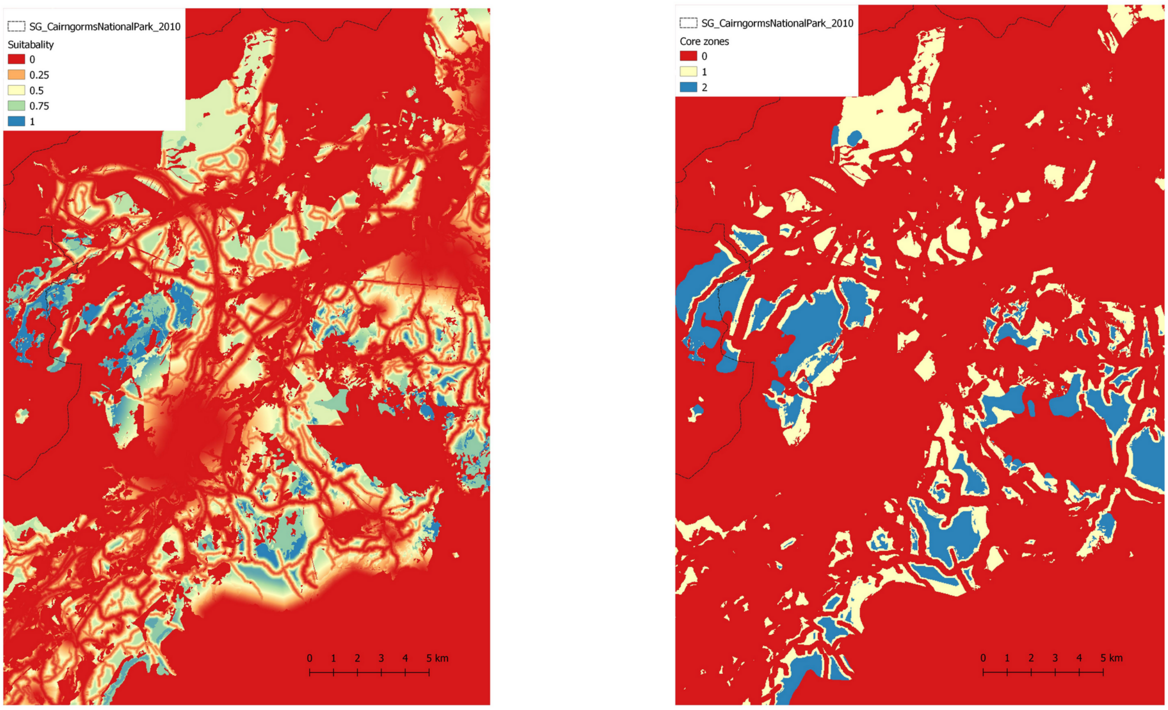


Figure 7.3. Caper maps results of Scenario3 run for CNP (zoom) that shows the suitability maps (left) and the core zone area for Capercallie.

To define better the core habitats of the bird a double check spatial process was carried out comparing the CaperMap zones with the LCM07v6s2_WoodR2n3 land cover map. The zone was considered a patch of valid conifer woodland if was simultaneously a native woodland conifer and a core zone as defined by CaperMap.

On the other hand the suitability maps were used to define the resistance matrix as an inverse function stretched between 0-100 values applying the formula:

$$Res_i = (1 - Suitability_i) \cdot 100$$

where *i* indicates a particular raster cell.

Therefore the resistance value (converted in resistance maps – Fig. 7.4) was assigned based on the inverse of habitat suitability because more suitable areas means higher levels of biodiversity, and consequently low resistance to the species. Broadleaved (selected from land cover map) and conifer (cross selection with CaperMap) woodland patches >0.5 ha, were defined as "sources" (origin of current/random walkers), while patches <0.5 ha were excluded because are more likely to suffer edge effects, and to have a low population of dispersers (Gimona et al.. 2012). Sources are defined as a key ecological patch that sustains ecological processes and from where ecosystem services are provided.

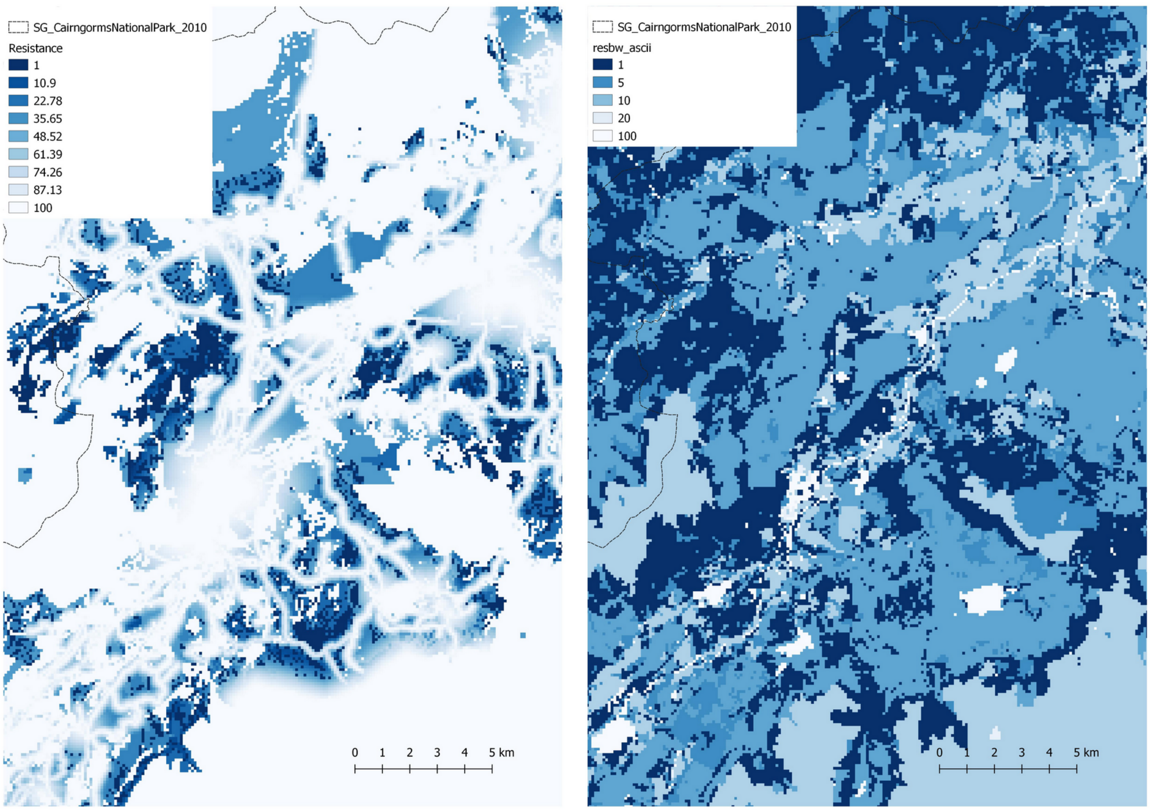


Figure 7.4. Maps of the resistance matrix (same zoom location) for conifer (left) and broadleaved (right).

7.3 Results

Circuitscape creates ASCII current and voltage maps for each pair of core areas, plus a cumulative current map that sums up current between all pairs. This cumulative value, defined as the density of random walkers are migrating from patch to patch (Fig. 7.5), depends on the resistance of the conductive surface which account for the distance from broadleaved source patches (Gimona et el., 2012), and from conifer patches, where CaperMap consider very little range movement of Capercallie from core zone (e.g. 5-11km, Moss at al., 2006).

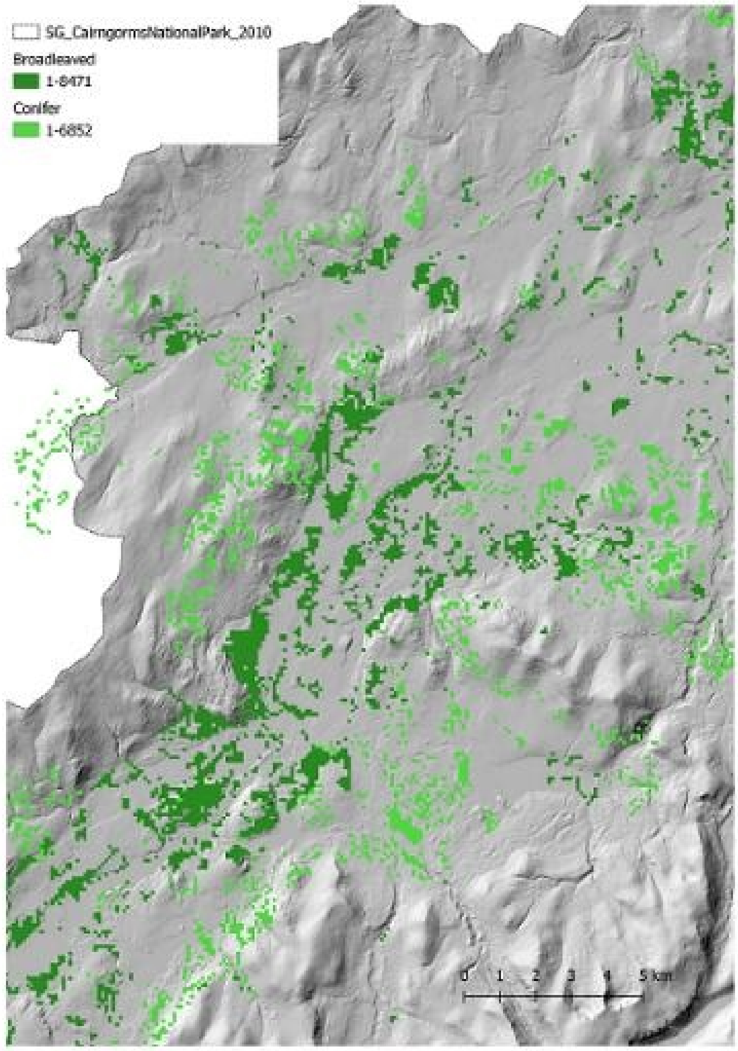


Figure 7.5 Broadleaved and conifer woodland patches

7.3.1 Current maps and potential dispersal avenues

The cumulative current density classes, in Fig. 7.6, is classified in quantiles and indicate areas with different probability of being traversed by random walkers. The blues are those with the highest probability of being traversed, hence are the areas where with connectivity is higher among forest habitat patches.

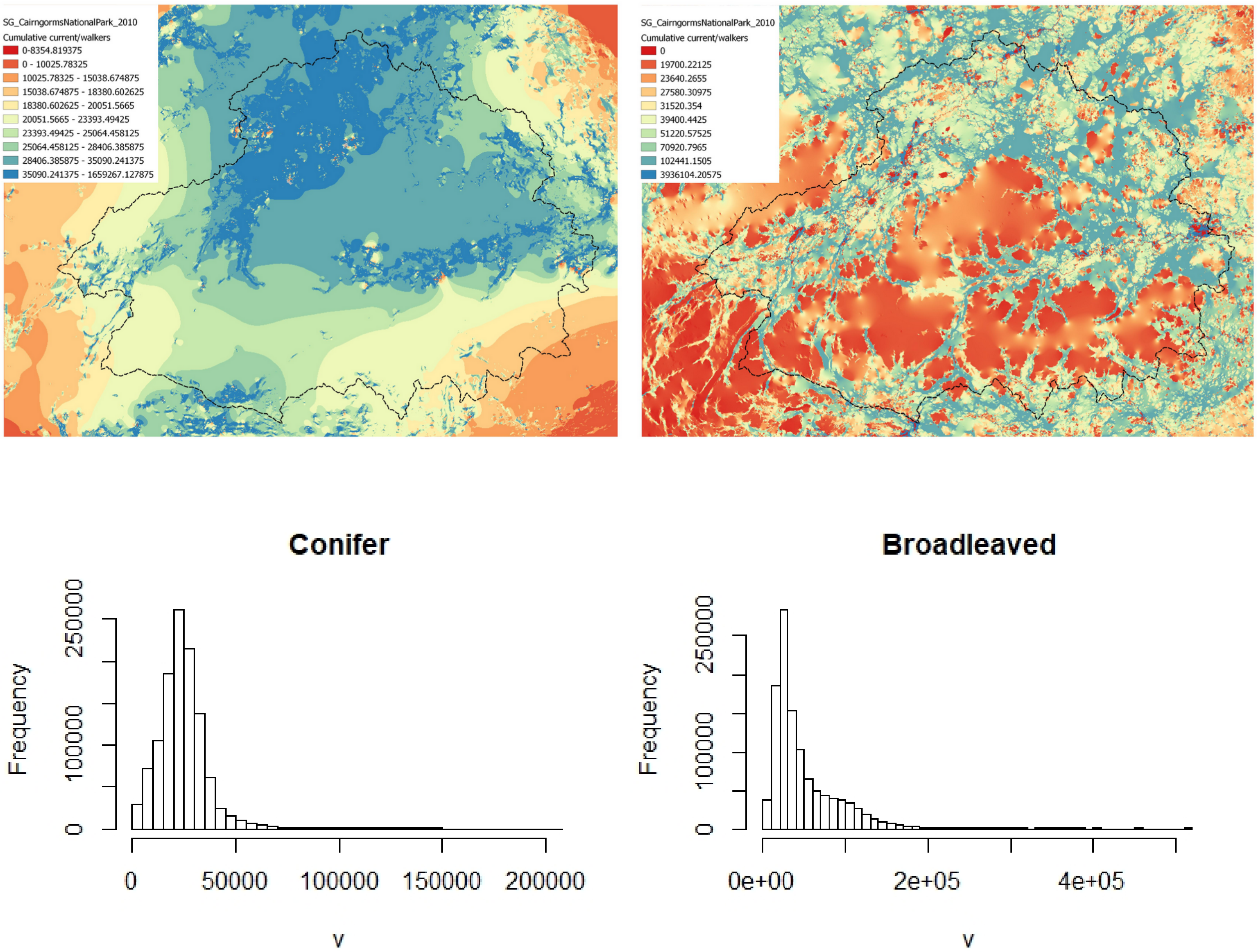


Figure 7.6. Conifer and broadleaved cumulative current modelled and classified with quantiles (above). Woodland landscape connectivity is more discrete for broadleaved because the strong link between land use and resistance. Histograms of the statistical distribution of cumulative current values (below).

7.4 Discussions and conclusions

The two types of target woodland have shown substantial differences related to the use of different data input, however the connectivity assessment is likely to vary geographically inside the main valleys of the river Spey and river Dee, where ecological sources are located. The resistance value was defined as the inverse of habitat suitability and was calculated to be in the range of 1–100 for both type of woodlands. Generally, the ecological resistance of coniferous species seems to be quite limited in the neighbourhood of the sources with a values that decay progressively over distance, while broadleaved species seems to be more distributed where available land cover classes occur.

Although the term "corridor" is recognized by ecologists, it is often used without explicit definition. The results of this analysis does not identify corridors of continuous habitat, but potential dispersal pathways (Fig. 7.7) and so the opportunity to introduce new stepping-stone (Beier & Noss 1998, Hess & Fischer 2001, Schmucki & de Blois 2009) woodland patches. Defining clearly a corridor's roles is only the beginning of the design process, developing design criteria for each of a corridor's functions remains an unfinished and formidable task and during this process, functional conflicts might be uncovered by the proposed land use change (Hess et al., 2001).

Parametrization of Circuitscape is heavily affected by the scarcity of the data specifically produces for target species. Despite that, data from Gimona et al., 2012 (elaborated version of Watt et al., 2010) were useful to complete this exercise. Other data such as CaperMap produced by Jim McLeod offered lower confidence in the ranking of the resistance values, compared to the absolute values from literature used for broadleaved, but represents the only available option to introduce a proxy data rather than a specific expert based opinion.

In this study we were unable to provide firm proof that such woodland expansion in the potential dispersal pathways leads to increase of species populations (or decrease of mortality), but it is likely to create the opportunity for further investigation of land use change consequences. Having said that when there is no better alternative given the time and resources required to collect detailed life history data on several species (e.g. Etienne et al., 2003) this approach is useful in determining planning opportunities.

Chapter 8 – Spatial multi criteria analysis (sMCA) – background, methods and supporting data

8.

8.1. Introduction

This chapter aims to describe the method used, the additional criteria and areas of no-go for trees (hard constraints) and the negative criteria intended as the areas advised against for tree planting (soft constraints). These extra criteria can add further information to the sMCA designed to help support decisions about land use change. Effective support for land use change means identifying areas that are appropriate for the proposed change (e.g. woodland expansion) but where this land use change could also either provide other benefits (such as recreation opportunities) or reduce problems (such as poor water quality).

Decision-making in forest planning projects requires consideration of landscape across its between functional components and this exercise is often complicated due to the wide range of criteria involved. In order to be effective forest managers must wear the conservationist hat to target high-priority lands by focusing on the integration of scientific criteria with considerations from local residents and land owners. The best way to arrive at a conservation program decision that is democratic requires consensus (Innes et al., 1999; Sayer et al., 2013) therefore tools that help maximize agreement and minimize conflict among different stakeholders groups are always preferred.

Over the last twenty years, multi-criteria decision making (MCDM) is an approach applied for solving forest resource management disputes (Pereira and Duckstein, 1993, Keisler and Sundell, 1997). It was amply demonstrated that the monetary valuation of ecosystem services is not straightforward (Schröter et al., 2016), sometimes even detrimental (Temel et al., 2018) and the non-monetary valuation is not necessarily easier, but it is often key to environmental MCDM (Martinez-Alier et al., 1999, Carbone et al., 2000, Munda, 2000). Multi-criteria decision analysis (MCDA) form the basis of MCDM. This is a tool that can centre the importance of consider multiple information in strategic planning, and can be used of helping decision making for multiobjectivity. In addition, it has been used as an effective ,means in directing forest owners to comply with the public (governmental) forest policy (Kangas et al., 2001).

Under the umbrella of MCDA can be found a wide variety of different ways to elicit inputs approaches, representations, combination algorithms, and processes for interpretation of results. Recently, (Ananda et al., 2009, Diaz-Balteiro et al., 2006) have produced an exhaustive survey of a larger number of journal articles and text books published on MCDM applications in forest management, concluding that computation of elements of the methods (algorithms) aids and innovative advancements in MCDM informatics available can accelerate the use of MCDM in forest management problems. According to this trend, selection of specific methods seems to be driven by availability of specific expertise and software tools (Huang et al., 2011).

Simple methods where multiple criteria are base on cut-off values, namely non-compensatory approaches, outlined theories which increases in the value of one criterion cannot be offset in the value of another one (Hwang and Yoon, 1981). The simplicity makes methods easier to understand and apply, but they require including or excluding alternatives based on hard cut-offs. On the other hand, compensatory approaches use multi-attribute utility theory (Keeney and Raiffa 1976), multiplying weights by normalised criteria values (converted to a continuous 0–1 scale) and summing these to derive a score or rating for each alternative. In this modelling exercise, criteria can be traded off against each other on a continuous scale, so that a weight decrease in one criterion can be compensated for by an increase in another.

The last 20 years or so have evidenced a technological progression and a remarkable quantity of researches on spatial multi-criteria analysis (sMCA) and integrating MCDA into GIS (Pereira and Duckstein 1993; Jankowski 1995; Laaribi et al. 1996; Malczewski 1996, 1999; Thill 1999; Chakhar and Mousseau 2008, Perpiña et al., 2013, Nguyen et al., 2015, Van Hoang eta al., 2020).

The combination of GIS and MCA added allows decision makers to deal with the problem of handling large mounts of complex information and to divide the main issue into smaller problems and then integrate the assessments in a logical way. Further, sMDA have the flexibility to integrate additional information, that do not directly represents the opportunities and constraints, but could have an indirect impact (population, GDP, etc.) in the identification of priorities.

There are many classifications in place for the formal MCDM methods that have been implemented in the GIS environment including compensatory weighted linear combination (WLC) (e.g. Malczewski 2000), reference point methods (e.g. Tkach and Simonovic 1997) and analytical hierarchy process (AHP) (e.g. Rinner and Taranu 2006) .developed by Saaty (1980). Among these, the WLC method is the more realistic and subtle in their modelling, and most widely-used method (Eastman et al. 1993; Malczewski 2000). WLC methods can calculate the total value score for an alternative as a linear weighted sum of its scores across several criteria,

$$V=\Sigma_i w_i \cdot x_i, \text{ where } \Sigma_i w_i = 1, \qquad [1]$$

however, in WLC methods is also possible to broke down dimension i into several subdimensions j, where x_{ij} is the alternative's score on the jth subdimension of dimension i,

$$vi = \Sigma_i w_{ij} \cdot x_{ij}, \text{ and } V=\Sigma_i w_i \cdot v_i. \qquad [2]$$

Among the fundamental principles of MCA stands out the concept that the criterion weights represents the relative importance of their criterion value (Massam, 1988). In this study we applied this principle to develop a compensatory explicitly spatial GIS-based WLC (GIS-WLC) model for defining woodland suitability in the Cairngorms National Park. This suitability, oriented to define a multifunctional woodland expansion, was realized combining the (dis-)benefits maps produced and described in chapter 4, 5, 6, 7 with additional spatial information described in chapter 8.

8.2. Methodology

The work makes use of previous described methods and ecosystem services (criteria) to create woodland value maps for water quality, natural flood management and biodiversity (see chapters 5, 6 and 7). These value maps are used as input in the analysis, focusing on the case study of Cairngorms National Park (see chapter 3). Therefore, the benefit maps produced at national scale (e.g. nutrient and sediment export) were clipped to the CNP extent. These three raster maps were selected to investigate the dynamic of woodland, while others were included in the analysis with no change in weighting.

Because within CNP there is so much concern about native pinewood forest, due the sensitivity of the species they host, we additionally divided the analysis in two branches to address different spatial suitability for broadleaved and conifer woodland. Therefore, the two parallel, but distinct analyses, refers to different spatial attribute maps, specifically produced for the analysis they refer to (e.g. broadleaved analysis consider dispersal avenues for broadleaved species).

Although nutrient and sediment export maps represent a single opportunity to expand woodland here we adopted the GIS-WLC to hierarchically consider the compound of these two criteria as different aspects of the same service, specifically as a subdimension of the water quality. The reason for this is located in the forest function itself, in fact it is widely recognized that afforested areas may be a substantial sink for nutrients (agriculture in-organic origin) that would be otherwise susceptible to leaching and transported to the stream network (Burt and Pinay, 2005, Lohse et al., 2009, Nieminen et al., 2017). On the other hand, forest in buffer zone may be efficient in reducing mostly total suspended solids (TSS) and phosphorous (P) to water courses which is a serious water quality problem (Jordan et al., 1997, Pärn et al., 2012). Hence, if nutrient and sediment export affects the movement of N and P, we can consider these process as the main component of the water quality attribute.

The scientific debate on reduction and adaptation to climate change, has recently driven the attention to the hot topic of how countries can reach net-zero emission and the capacity of forests to partially offset this release. Although this argument is going on for a while there is still a level of confusion in determining whether afforestation is a good or a bad way of meeting future carbon emission targets (Matthews et al., 2020, Brown, 2020, Harper et al., 2018) especially when peatland and carbon rich soil are considered for woodland expansion schemes (Smith, 2012, Minkinnen et al., 2008). A number of public bodies have characterised peatland sites but this effort was largely uncoordinated (Vanguelova et al., 2018). Carbon and Peatland Mapping (NHS, 2016) derived from 1:250000 soil mapping validated against habitat data (Scotland's Soils, 2018) is one of the most commonly used dataset to identify organic soil and potential sensitivities when afforestation planning is considered. However, in Baggio Compagnucci et al., 2021 (submitted) we have demonstrated that carbon emissions and the potential of afforestation scheme to offset part of the GHG can be manoeuvred by the intensity of ground preparation. Therefore, in this work, was applied a intergovernment policy that defines soil carbon sensitivity. To achieve this, the budget for 2050 obtained in Chapter 4, was used as a hard constraint to narrow down woodland expansion analysis where a net positive is due under climate change variations.

The function of this chapter is therefore to introduce the methodology applied for the study of new woodland opportunity within the Cairngorms National Park accounting for the multifunctionality of the broadleaved and conifer afforestation sites. On the other hand this chapter outlines all the additional data that, together with the four criteria presented in the previous chapters (Carbon sequestration, Water quality, Flood mitigation and Functional connectivity), will be used to investigate the potential woodland suitability in the protected area.

8.3. Additional criteria and soft constraints

The sMCA is created by integrating the results from previous chapters (from 4 to 7) while another five baseline criteria are added. These additional layers are included in the sMCA model, but not in the weighting process previously described. The purpose of these layers is to create more variability in the final quantitative results, therefore they will not be altered to influence the woodland suitability map by the weighting process but their emphasis and the reasons for being included in the analysis are illustrated here.

8.3.1. Productive lands

The association of farmland with forest and mountain is a key ingredient of the Cairngorms National Park landscapes, in some places the dominant ingredient. Change to these landscapes as a consequence of declining farm management could in some places have a significant effect on the special landscape qualities of the Park.

High-nature-value, low intensity agriculture including wood pasture and agro-forestry contribute significantly to landscape character of the National Park. The combination of wetlands, wet grasslands and low intensity mixed farming hosts one of the most important UK mainland sites for breeding wading birds. Combined with careful seasonal grazing rare habitats depend on farming and crofting, eg species-rich grasslands and aspen woodlands containing the dark bordered beauty moth (a key species in the Cairngorms Nature Action Plan, CNPA, 2019).

Following the departure of the UK from the European Union's Common Agricultural Policy (CAP), the future of agriculture is uncertain; it is important to ensure that the Cairngorms National Park Forest Strategy helps to reduce conflicting objectives of securing the future of farming in the National Park alongside new woodland creation. The culture of crofting and farming in the

National Park needs to be maintained, while retaining the potential to market local, fresh, healthy and environmentally sustainable farm produce.

8.3.1.1 Methods

All arable and improved grasslands were extracted form the land use baseline (See paragraph 8.4).

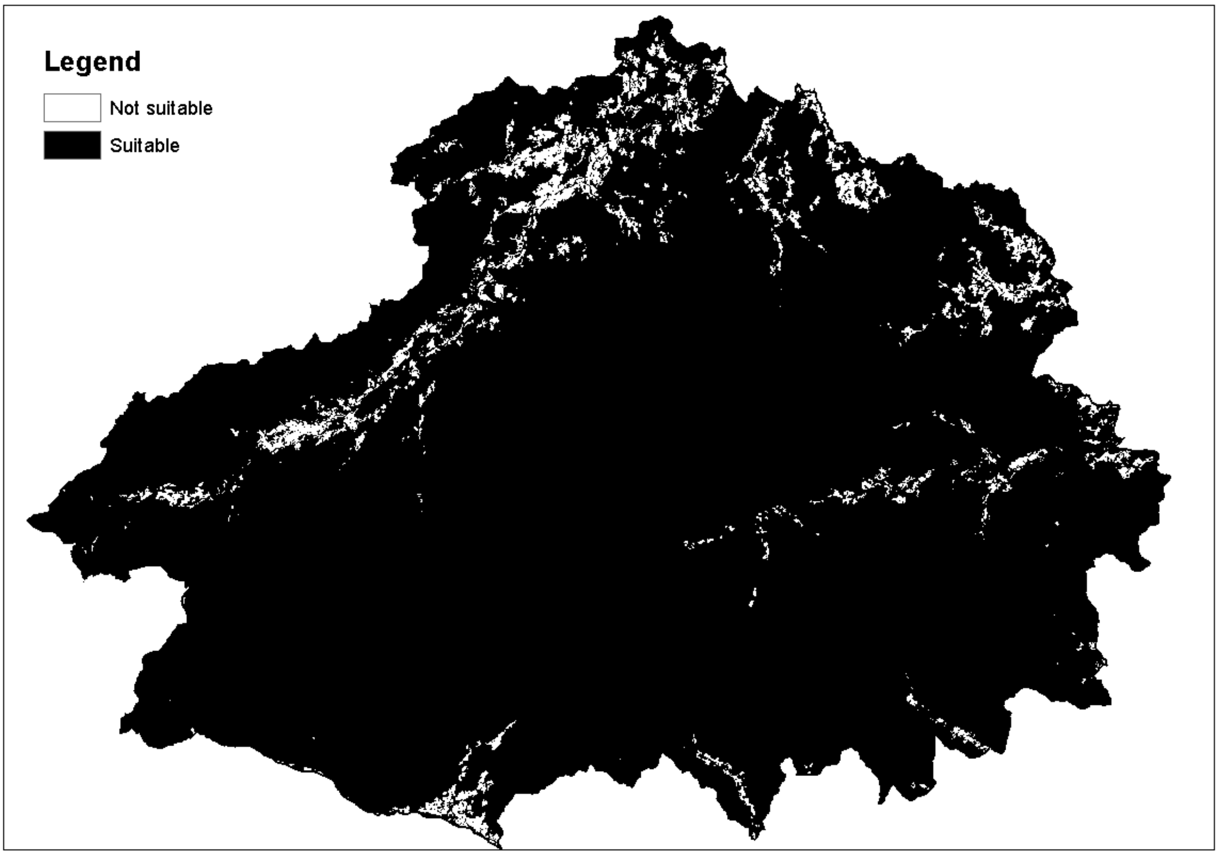


Figure 8.1. Additional criterion (map) of productive lands in CNP.

8.3.2. Non priority areas for waders

8 In this work, the Apparent Occupancy (AppOcc) value for five waders species was used, obtained from Debbie Fielding (JHI), and referenced to the JHI technical report (Newey et al., 2016). In the report, authors map the distribution of five species of wader; curlew, lapwing,

oystercatcher, redshank, and snipe to illustrate how their numbers have changed between 1990 and 2010.

Maps from report, obtained in vector format, illustrating spatial variation in relative abundance, and abundance change, were based on the outputs of predictive models. These models described the relationship between environmental variables and either recent abundance and change in occupancy (a measure of relative abundance).

8.3.2.1 Methods

Following method was applied to the data obtained from Newey et al., 2016 in order to give more priority to the species that is well represented (Fig. 8.2). In particular,

• Soft Constraint layer (Fig. 8.2) was obtained as $Wders\_soft_i = SUM(AOs_i * Ws\_r_i)$

Where *AOs* is the Apparent Occupancy for each species *i,* and *Ws_r* is the weight for each species calculated as $Ws\_r_i = SUM(Rcl(1/NumC_i)$

where Rcl is the rescaling (from 0 to 1) function using formula *(value – min) / (max – min)* and $NumC_i$ is the number of cells having Apparent Occupancy more than 0.

• Also, using the same data input, an Hard Constraint layer was defined if one or more species are present in the Special Areas of Conservation (SAC) and Sites of Special Scientific Interest (SSSI).

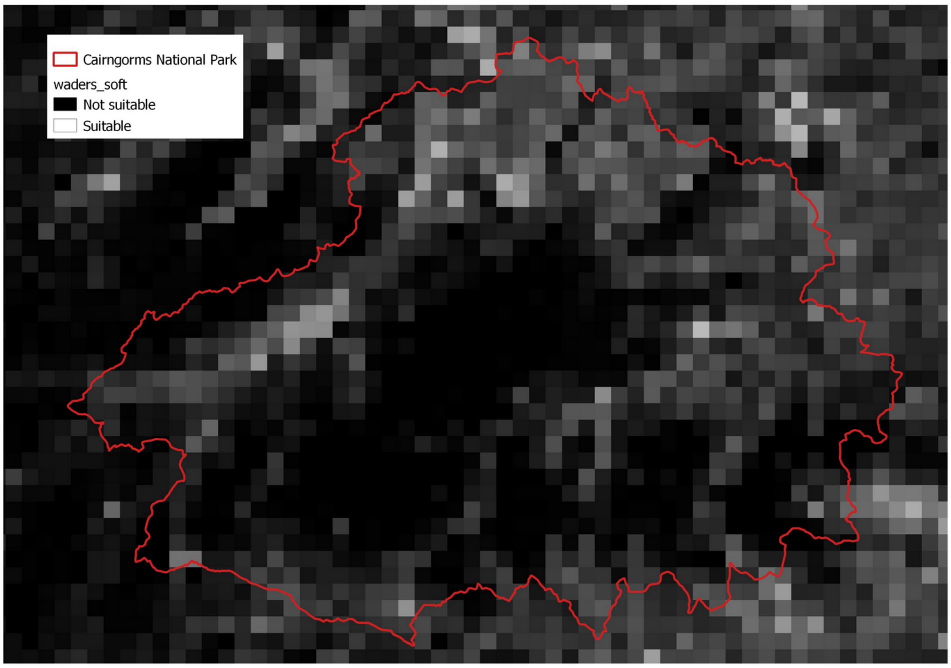


Figure 8.2. Definition of the priority habitats for wader birds.

8.3.3. Speacial Areas designed for woodland

CNP Forestry Strategy 2018 (CNPA, 2018) highlights an extra caution when considering woodland creation in any designated site, including: SAC and SSSI.

An assessment of the suitability for woodland creation on each SAC and SSSI in the National Park has been carried out by Scottish Natural Heritage (SNH) and a summary of the suitable areas for woodland is listed in appendix B.

8.3.3.1 Method

This two dataset are considered a suitable area for woodland if the name of the designed site match with the SNH assessment (appendix B), contrary the designed site is considered as an hard constraint. The resulting layer in Fig. 8.3 shows the area of the first conditional overlap.

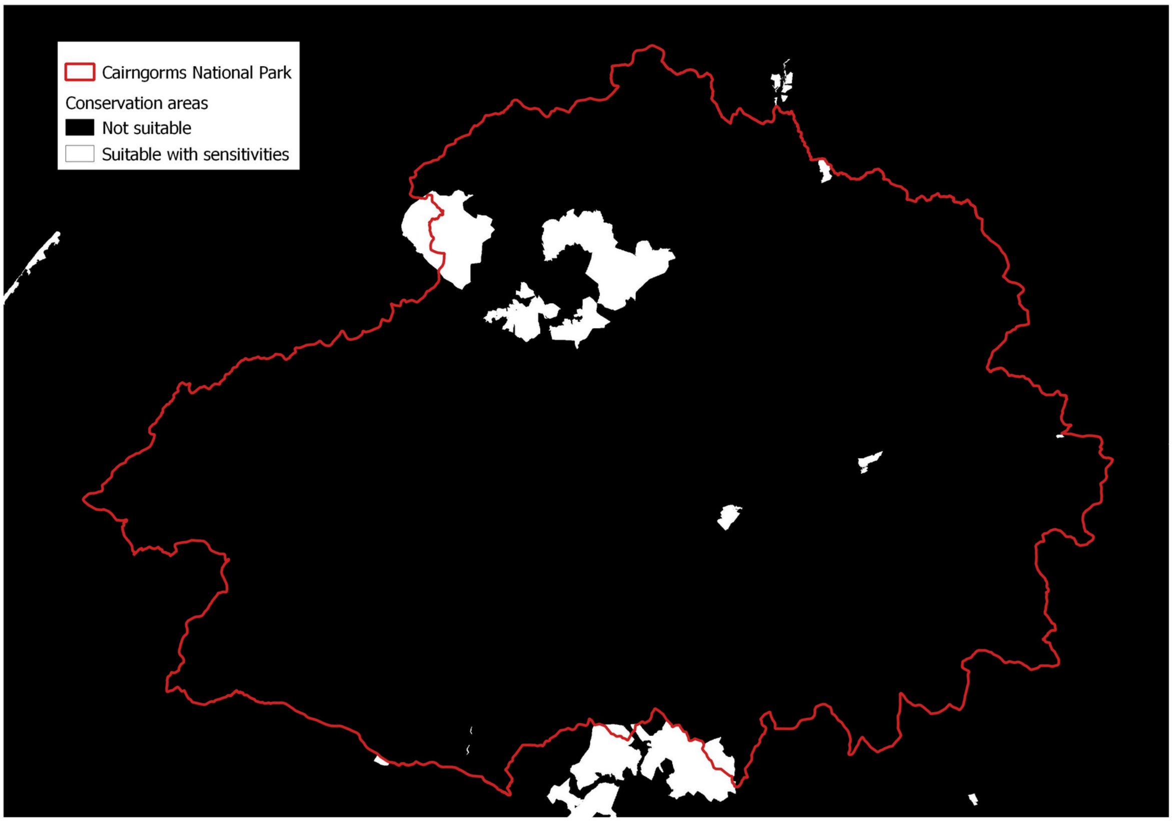


Figure 8.3. Soft constraints areas defined using NSH assessment.

8.3.4. Distance from woodlands

Outwith the Cairngorms national park the landscape is very fragmented and the proximity between similar habitats has been lost. Within the protected area the connection between similar habitats is still relatively achievable due to their closer spatial distribution. Prioritizing areas, starting from the closest suitable areas, and making links between fragmented sites, should greatly increase the value (biological and economical) of existing sites avoiding to (re)create large new areas of additional habitat. Therefore, this criterion, targets isolated patches making them functional habitat units once again. Notably, increasing the size and connectivity of larger patches could have some positive effect in the resilience of the woodland systems.

8.3.4.1 Method

Cells defined as forest were clumped together to define the patches of broadleaved and conifer woodland. Form those patches, then Euclidean distance (Fig. 8.4) was calculated using the r.grow.distance function of GRASS-GIS (R Core team, 2020). A further elaboration of this analysis was introduced to exponentially decreases to 0 the distance value at the point of 2000m (cutoff) from each patch.

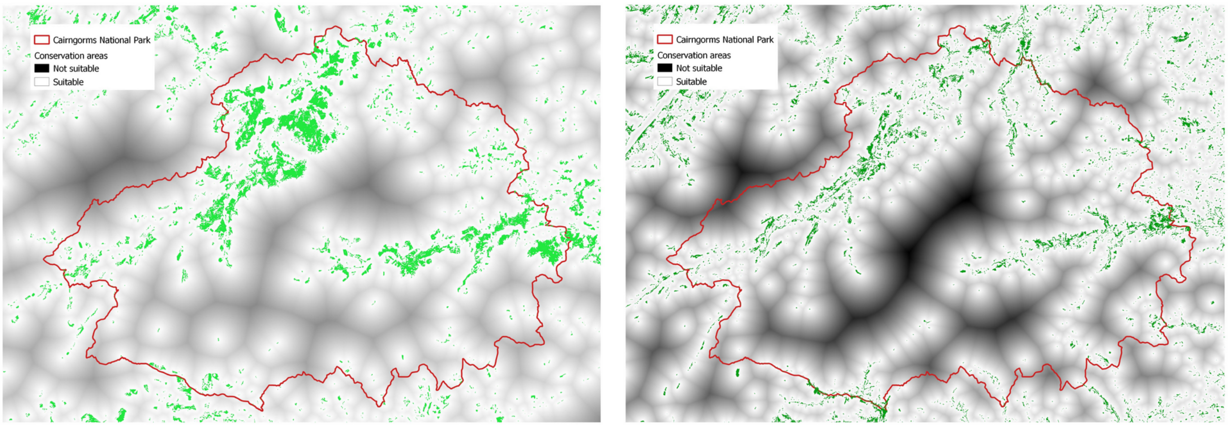


Figure 8.4. Maps of distance form relevant patch of native conifer (left) and broadleaved (right).

8.3.5. River buffer

The riparian zone is defined as any area adjoining the edge of a watercourse or waterbody. The main benefits of riparian woodland are improvements to water quality, shading to reduce summer temperatures for salmon and freshwater pearl mussel, bank stabilisation and an increase in habitat diversity and connectivity.

Rivers and burns are natural corridors along which riparian woodland can create woodland habitat linkages within and between river catchments. Gullies formed by upland burns can be refuges for woodland remnants, also containing associated understory species.

Eurasian beaver is currently established close to the National Park. It is possible that beaver may return to the National Park in the future. A significant increase in riparian woodland is needed to ensure sufficient habitat to minimise potential impacts of future beaver populations.

8.3.5.1 Method

This layer was created by applying a 50m distance buffer (Fig. 8.5) to the whole the river network inside GRASS-GIS (R Core team, 2020) software.

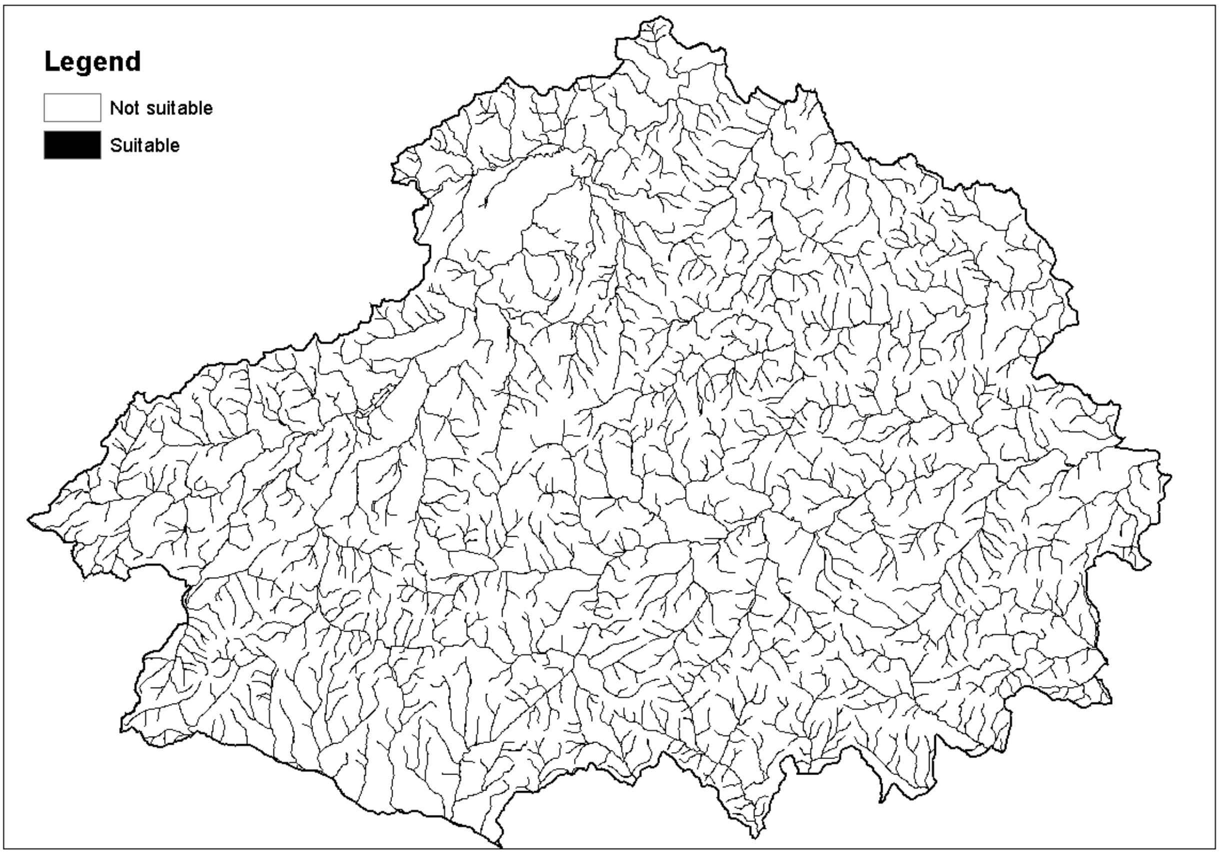


Figure 8.5. Map of the river 50 m river buffer.

8.4. No target areas – hard constraints

Besides these additional criteria, "no target" areas were carefully excluded by the analysis because of their bio-physical constraints (e.g. wet areas or already afforested). The existing Forestry Strategy 2018 (CNPA, 2018), aims to strongly encourage landowners to consider more woodland creation and natural regeneration using Scottish Forestry Grant Scheme limiting funds to where they are needed most by mapping a 'target area'. However, those locations are missing all the important information that we produced in this work, such as the impact of the type of ground preparation and the related level of carbon gain/loss due to the soil disturbance (See chapter 4).

8.4.1. Existing woodlands

The exclusionary layer was applied to mask out areas that don't need to assess suitability for forest, since forest is already there.

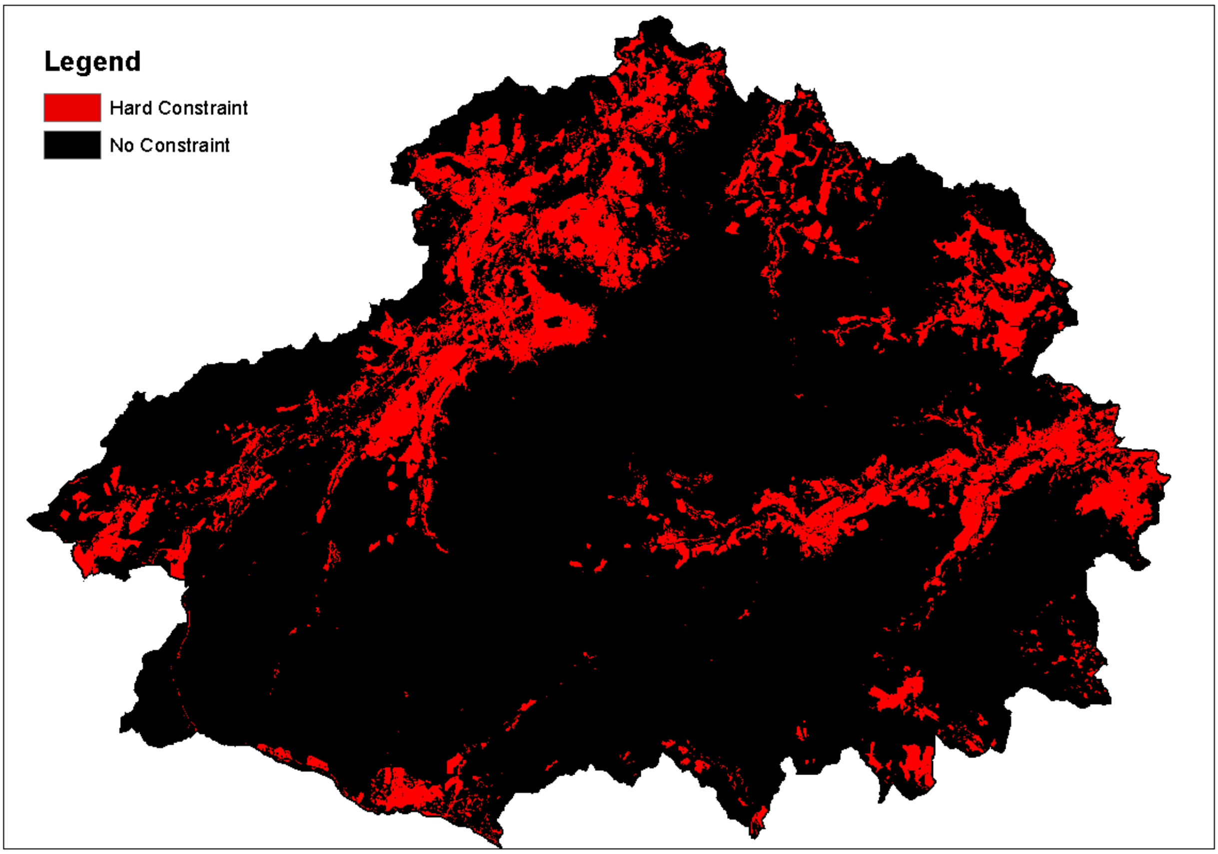


Figure 8.6. Map of the hard constraint of existing woodland.

8.4.2. Wind limitation (Detailed Aspect Method of Scoring)

Recurring peak of winds can damage trees that are standing in high wind exposure areas, both resulting from natural setting or due to harvesting and thinning. This is one of the most commons natural disturbance associated with forest that can produce relevant ecological and economic impacts (Mitchell et al., 1995).

8.4.2.1 Method

A DAMS (Detailed Aspect Method of Scoring) score equal or more than 24 (expert opinion) was used to define the suitability for woodland. This represents the biophysical limit for tree growth without being heavily affected by wind-throw events correlated with high frequency storm conditions.

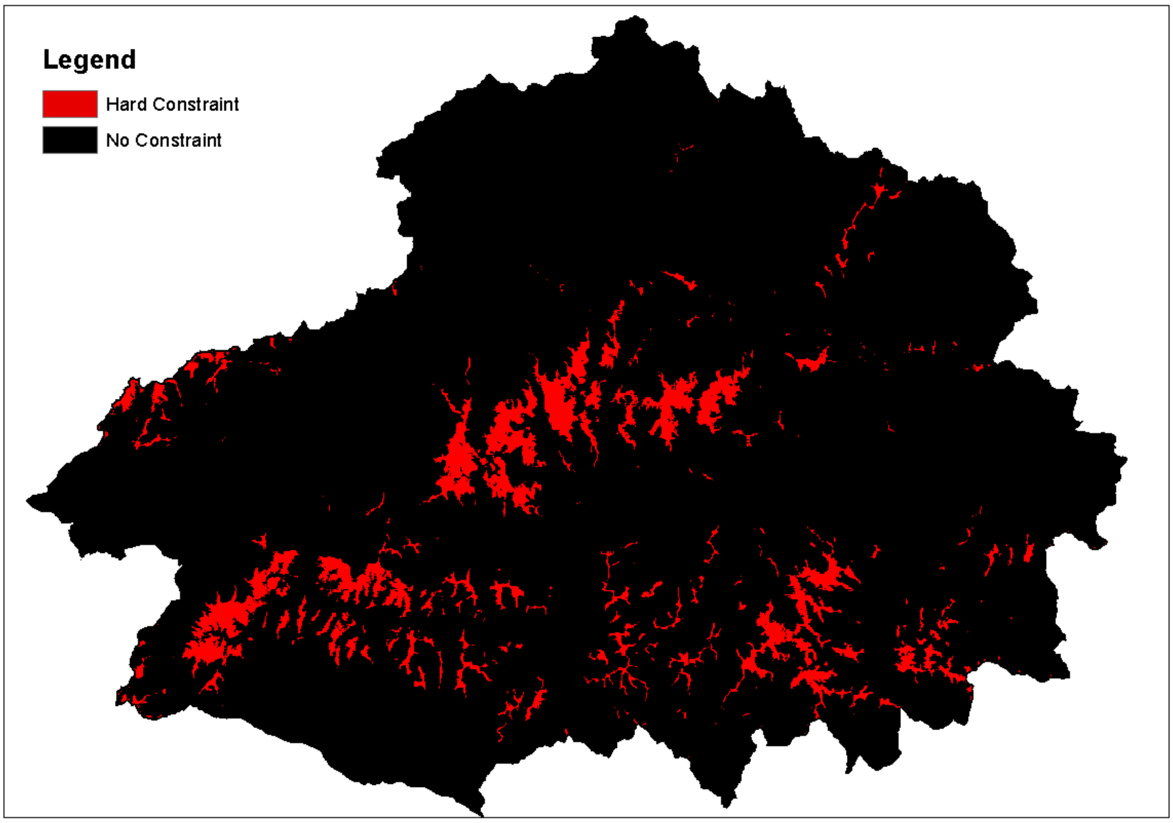


Figure 8.7. Map of wind exposure threshold for limiting tree growth.

8.5. Land Cover baseline

The land cover map used in this work was obtained from the technical report (Appendix G) developed at The James Hutton Institute as an update on work in the Strategic Research Programme (Work package 1.4.3) (Hewitt et al., 2018). The integrated land use datasets described in the first part of the report, have some main limitations relate to the use of mixed data from multiple sources or data not originally intended for that purpose. Datasets such as Integrated Adminstration and Control System (IACS) in fact are not ideally suited for land use time series analysis providing poor information outside the agricultural classification. However, joining land claims information to different spatial land cover datasets, even if some of them are out of date. Is possible to obtain a highly detailed map of agricultural land uses to use as input for the InVEST nutrient and sediment model. The extended classification, in which key crop types with known nutrient loads are disaggregated, was more useful than the simple classification (LCM2007), since it allows different arable cropping regimes with correspondingly different nutrient loads to be separately modelled. In this study, the extended land use classification produced in Hewitt et al., 2019 (number 2 of Table 8.1) was used for modelling exercise the water quality while the LCM2007 integrated with Forestry Inventory 2015 (number 3 of Tab. 8.1) produced at JHI by Marie Castellazzi was used as a baseline (Fig. 8.8) for land use change scenarios.

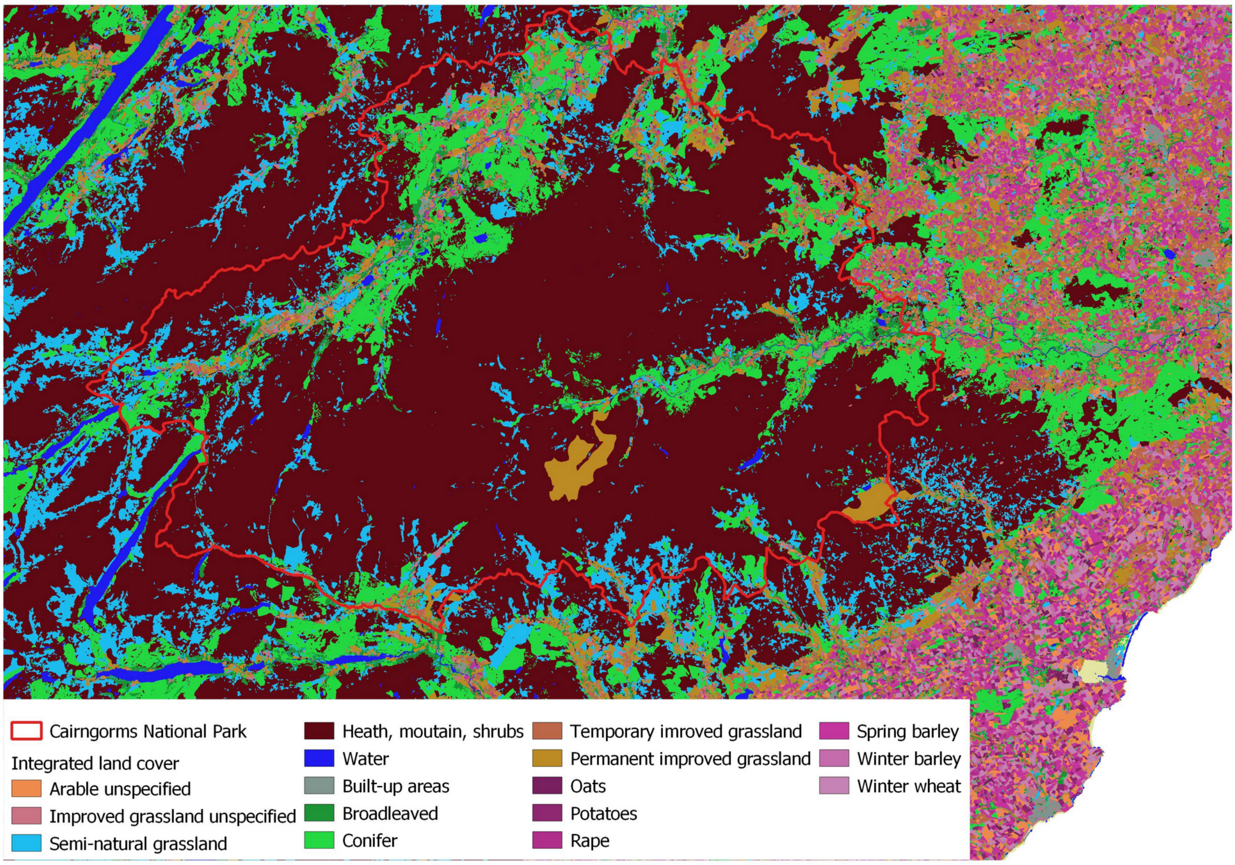


Figure 8.8. New classification obtained by the integrated land cover / land use map.

Table 8.1. Dataset obtained and produced in house by JHI.

No.	Name	Scale/resolution	Time periods available	Accessible	Description/sources	Type	Format	Created by
1	**IACS predominant land use 2008-15**	From 1:5000 (lowlands) to 1:50000 (uplands)	2008-15	Restricted access, contact creator	IACS surveyed land parcels with area claimed under CAP payments system, with predominant land uses assigned according to the extended classification (see documentation).	Land use information (spatial)	ESRI Shape file	Richard Hewitt Richard.hewitt@hutton.ac.uk
2	**IACS predominant land use, extended crops classification**	From 1:5000 (lowlands) to 1:50000 (uplands). Minimum mapped unit c. 0.2ha	2010, 2015 (can create any other date between 2008 and 2015 as required)	Restricted access, contact creator	IACS surveyed land parcels with area claimed under CAP payments system, with predominant land uses assigned according to the extended classification (see documentation).	Land use information (spatial)	ESRI Shape file	Richard Hewitt
3	**LCM2007 integrated with Forestry Commission woodland inventory data (LCM2007w2 and LCM2007w3)**	LCM states minimum mappable unit 0.5ha, though some woodland parcels may be smaller	2007 with 2015 woodland	Restricted access, contact creator	LCM2007 (produced by CEH), merged with Native Woodlands Survey for Scotland 2014, National Forestry Inventory for Scotland 2015, National Forest Estate Legal Boundary for Scotland 2016.	Land use information (spatial)	ESRI Shape file, 25m raster	Marie Castellazzi Marie.castellazzi@hutton.ac.uk
4	**IACS_LCM07w_raster**	As LCM2007	2007/20 10, 2007/20 15	Restricted access, contact creator	IACS and LCM2007w3 merged using the ArcGIS MOSAIC tool, giving overlay priority to IACS.	Land use information (spatial)	25m raster	Richard Hewitt
5	**IACSextended_LCM07w_raster**	As LCM2007	2007/20 10, 2007/20 15	Restricted access, contact creator	IACS and LCM2007w3 merged using the ArcGIS MOSAIC tool, giving overlay priority to IACS.	Land use information (spatial)	25m raster	Richard Hewitt

Chapter 9 – Spatial multi criteria analysis (sMCA), land use scenarios and results

9

9.1 Introduction

The following chapter describes the undertaken sensitivity analysis using the method (WLC) and data (criteria) described previously and outlines specific land use scenarios linked with modelled preference of hypothetical group of stakeholders. The original design of this thesis was to include the real orientation of a selected group of stakeholders that are directly involved in forestry activities within the CNP (e.g. CNPA, SNH, RSPB, local communities, Cairngorms Parteneriships, Dee Catchmebnt Partnership, Cairngorms Connect, Aberdeenshire Council, Perth&Kinross Council, etc.) however this part was constrained by resources and time. Artificial simulations of the preference were therefore used to drive the choices of the probable policy makers and, consequently, the decisions for addressing afforestation and future land use changes.

The weights of importance are alternatively doubled using the weighted linear combination (WLC) in a methodology that is simple, but reliable and valuable for a conflicting decision-making environment.

9.2 The attribute maps (criteria)

The attribute maps (Tab 9.1) represent the (dis-)benefit score of each cell in the landscape.

Our attempt has the need to integrate both qualitative (binary) and quantitative (continuous) data, however is important to make a distinction between spatial multi-objective problems (or models) and the algorithms to solve the problem. This can be categorized into discrete or continuous (Goicoechea et al., 1982). A discrete variable is limited to a fixed or countable set of values, while a continuous variable can take on any value in a specific interval, otherwise it is a mixed model. If all variables are discrete, the mode is a pure integer one and results are limited to few possible options, while if the values of all decision variables are continuous the model will involve a higher number of alternatives in a more variable While discrete measures has been tested useful for demonstrating a certain level of efficacy in decision-making (Ram et al., 2011; Uhde et al., 2015) , others studies showed that the loss of information when changing from the underlying continuous scale to the binary outcome measure results in a loss of power to detect differences in the analyses (Schmitz et al., 2012) because in the construction of the surrogate over continuous variables, there is the assumption of continuity: as a continuous variable varies by a small amount, the response is assumed to vary smoothly (Swiler et al., 2014).

In this work we addressed the discrepancy between the scale of process variables and the scale of decision alternative by adopting a rescaling technique to increase the cohesion and efficiency of managing data, where map attributes (criteria) with different measurement units share a common scale in the interval (0-1).

The maps such 50m river buffer (BUF), dispersal avenues (DSA), production (PRO), waders habitat (WAD), and conservation (CON) were classified as binary maps (0 or 100) because they describe the presence or the absence of the service/opportunity and cannot have a continuous value, so these maps are substantially add information whether an area is suitable or not. Priorities from maps as the two components of water quality (WQ) (nutrient (NUT) and sediment (SED)) were defined with the top 75th percentile (high confidence interval) of the result from the specific analysis. Contrary, natural flood management (NFM), carbon sequestration (CBS) and distance from woodland (DIST) which have a continuous value were standardised (see 9.4) between 0 and 100.

9.3 Sensitivity analysis

Preference weights measured for different land management alternatives or multi-services can vary significantly across individuals and across groups engaged (Gimona et al., 2007). Although the original design of this thesis was to consider the involvement of decision makers and stakeholders, this exercise was not possible due to the limited amount of resources and time available. Therefore, a sensitivity analysis (double weighted) was used to review the various risks and changes in model inputs. This analysis helps check five different priorities that could potentially reflect the different emphasis of different policy makers or implementers (e.g. conservationist might favour ecological connectivity – Priority 3). The results obtained from the sensitivity analysis verify the robustness of the solution as a negotiation between different stakeholder interests. Priorities were defined using the five selected criteria (NFM, carbon, nutrient export, sediment export, and connectivity). To simulate this, 5 different sets of weights (Tab. 9.1) were chosen. One equally weighted base model was applied to all criteria, while in the other three scenarios, each criterion was double weighted versus the other attributes.

To investigate the multi-functionality of the two types of forest attribute maps were combined with the GIS-WLC. A normalisation was firstly applied to all criteria to rescale value from 0 to 100 using the formula:

$$K=((x-minR1)\times(maxR2-minR2)/(maxR1-minR1))+minR2 \qquad [3]$$

Where K is the new attribute map, x is a generic cell value, $R1$ is the map to standardise and $R2$ is the map to match with.

Each combination (split for broadleaved (bw) and conifer(cw)) was then obtained applying the general formula:

$$Cj = (BUF*w1) + (DSA*w2) + (CBS*w3) + ((NUT*w4_1)+(SED*w4_2))*w4$$

$$+ (NFM*w5) + (DIST*w6) + (PROD*w7)$$

$$+ (WAD*w8) + (CON*w9) \qquad [4]$$

where Cj is the set of combination and $w1...w9$ are the weights. Note that $((NUT*w4_1)+(SED*w4_2))$ represents WQ.

Forest type	Criterion	Code		Weights				
				EQW	Priority 1 natural flood management	Priority 2 water quality	Priority 3 ecological connectivity	Priority 4 climate change
Broadleaved	River buffer	BUF		0.125	0.100	0.100	0.100	0.100
	Natural flood management	NFM		0.125	0.200	0.100	0.100	0.100
	Nutrient	NUT	WQ	0.125	0.100	0.200	0.100	0.100
	Sediment	SED						
	Dispersal avenues for broadleaved	DSA_BW		0.125	0.100	0.100	0.200	0.100

	Carbon sequestration	CBS		0.125	0.100	0.100	0.100	0.200
	Distance from broadleaved	DIST_BW		0.125	0.100	0.100	0.100	0.100
	Productive lands	PROD		0.125	0.100	0.100	0.100	0.100
	Habitats for waders	WAD		0.125	0.100	0.100	0.100	0.100
	Conservation zone designed for woodland	CON		0.125	0.100	0.100	0.100	0.100
Conifer	River buffer	BUF		0.125	0.100	0.100	0.100	0.100
	Natrural flood management	NFM		0.125	0.200	0.100	0.100	0.100
	Nutrient	NUT	WQ	0.125	0.100	0.200	0.100	0.100
	Sediment	SED						
	Dispersal avenues for conifer	DSA_CW		0.125	0.100	0.100	0.200	0.100
	Carbon sequestration	CBS		0.125	0.100	0.100	0.100	0.200
	Distance from conifer	DIST_CW		0.125	0.100	0.100	0.100	0.100
	Productive lands	PROD		0.125	0.100	0.100	0.100	0.100
	Habitats for waders	WAD		0.125	0.100	0.100	0.100	0.100
	Conservation zone designed for woodland	CON		0.125	0.100	0.100	0.100	0.100
Hard Constraints	Existing woodland	EXW		NA	NA	NA	NA	NA
	Wind limitation (DAMS)	DAMS		NA	NA	NA	NA	NA
	Priority areas for Waders	WADna		NA	NA	NA	NA	NA
	Special protected areas not designated for woodland	SPA		NA	NA	NA	NA	NA

Table 9.1. Selected weights for combination. NUT and SED has a subdimensional weight of 0.5 each. The names of the five sets of combinations refers to equal weighted (EQW) and to the double weighted layer.

Each resulting maps from different sets was then masked out by the hard constraint layer. The spatial distribution of different benefits areas, for each map, was highlighted with the use of the final target following Scottish Government (SG) goals selecting the number of hectares that proportionally the CNP can allocate. Those resulting maps represent where high score benefits are located to meet the target for each combination of criteria.

Within this selection were also identified some ecosystem service bundles (Raudsepp-Hearne, 2010), which are consistently higher than the top quartile. Those represent areas with the highest probability to achieve win-win outcomes and where potential synergies between ES are located and solve the conflict between the potential stakeholders channelling the consensus.

Looking at the national target for woodland expansion, 450k hectares, to be reached in 2050, we proportionally calculate the area that will be allocate to the CNP, which is ca. 43k hectares. This expansion will project the protected area to have a woodland coverage of 161k ha, so the passage would be from the current 15.5% to over the 21%, which fully meets the national policy targets for woodland expansion (Scottish Government 2009; Reid, 2018, Scottish Government, 2021).

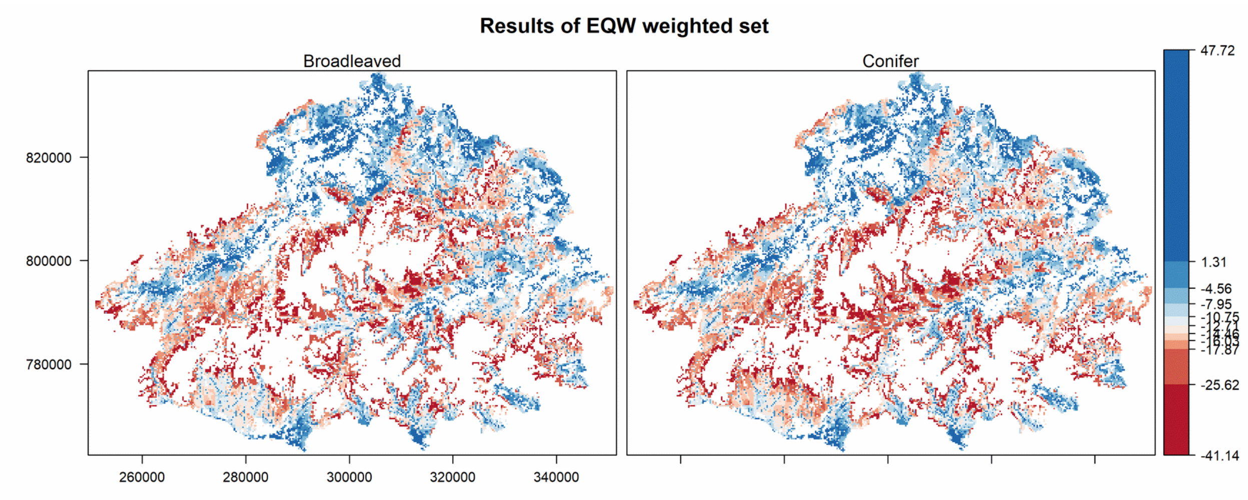
Those 43k hectares of woodland were inserted in the current land use map to simulate land use change scenarios in the protected areas and finally the amount of carbon that can be sequestered by 2050 was estimated to compare benefits for each option.

9.4 Results of Weighted Linear Combination (WLC)

The use of the WLC in the Cairngorms National Park was carried cell by cell through raster analysis tools for an area of 757860 hectares (total CNP surface) at 25 m horizontal resolution. From the overall analysis, using the hard constraints layers produced previously, the pixels were masked out, removing 513567 hectares from the combination results. Therefore, a remaining area of 24292 hectares was left available revealing that the target to expand forest of 43000 hectares in CNP is amply and physically possible. The application of the hierarchical weights combination method allowed obtaining two set of suitability maps (broadleaved and conifer) of afforestation to enhance water quality, flooding mitigation, biodiversity, and carbon stock.

9.4.1 Equally weighted set (EQW)

In particular, the results of the equal weighted map (EQW) (Fig. 9.1) revealed a general setting of the high scores located in the upper part of the study area along the valley of Spey river, with few high scoring zones in the extreme southern part and a scatter pattern of medium-high values along the valley of River Dee. Conversely, the low scores are located in central part of the study area, mostly on the ridges between the valleys and along the slopes of the glens that connect the main valley with the CNP plateau. The analysis of the histograms allowed to define the median (-12.709 and -13.118 for broadleaved and conifer respectively) and the lower limit (from where to start to select the quantity of hectares to meet the 43k target). The further classification (quartiles) of the 43k hectares selected for each type of forest (Fig.9.2) however showed that excellent to high suitability classes in EQW combination are located in the same areas both for broadleaved and conifer, while classes from good to low migrate to separate locations.



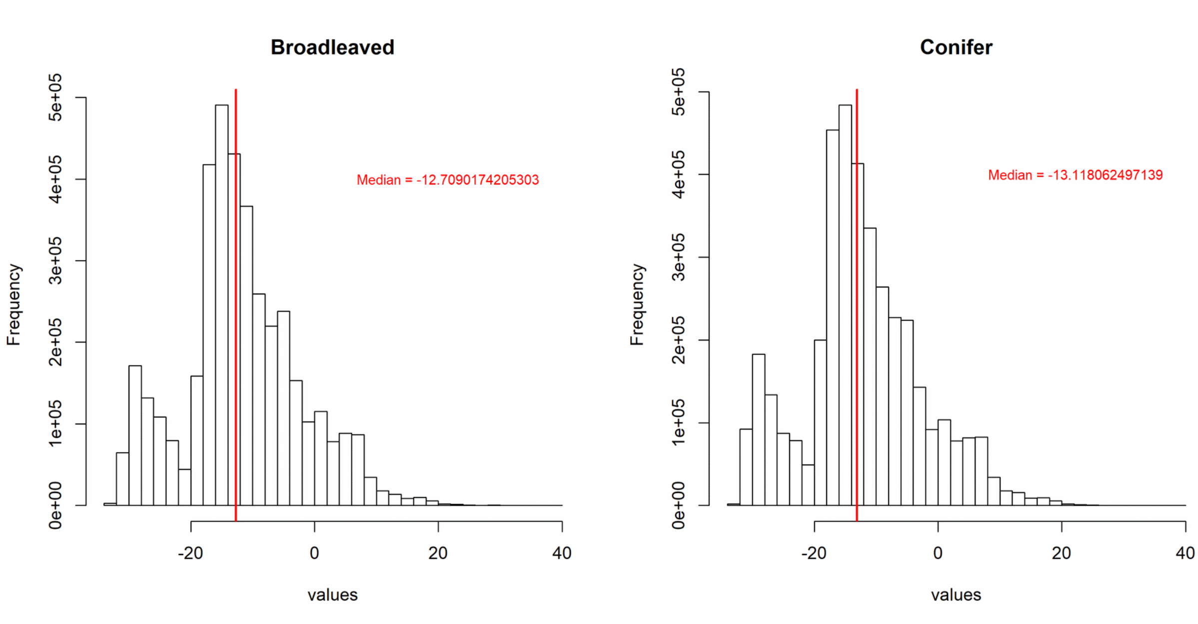


Figure 9.1. The heat maps for EQW combination for broadleaved and conifer and the respective histograms.

Main differences in values between the two combination (broadleaved and conifer) are more evident in the upper part of the River Dee and along the border of the lower part of the River Spey. In total, the categorization of the target identified 45011 ha and 44943 ha for conifer and for broadleaved, respectively, just above the limit fixed to be in line with the national goal.

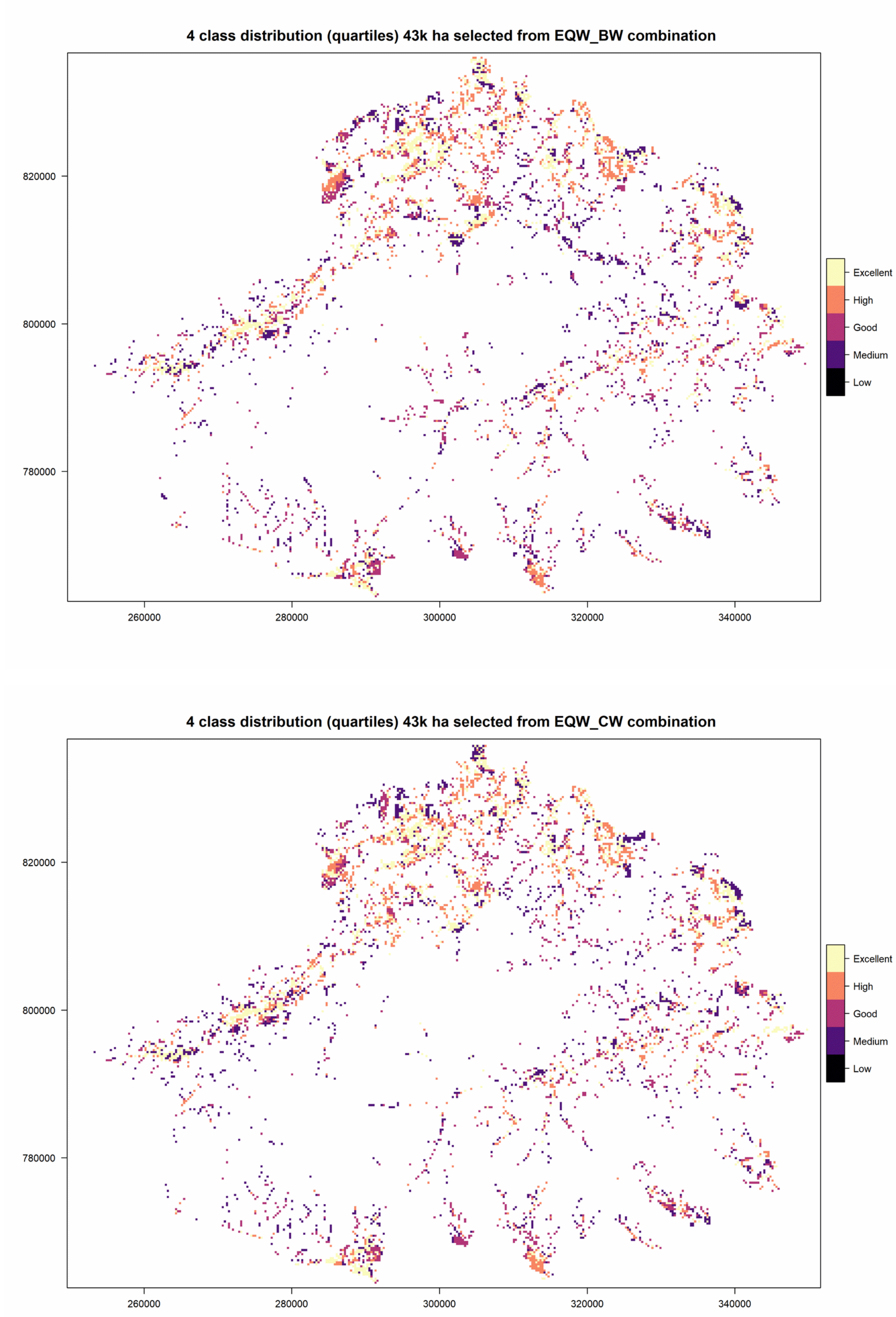


Figure 9.2. The categorization (quartiles) of EQW results of the selected 43k ha for broadleaved (above) and conifer (below).

9.4.2 Priority 1 – Natural flood management

The results of the double weighted NFM set (Fig. 9.3) revealed a substantial and generalised increase of the scores compared with EQW maps. In this combination, most of the areas that in EQW maps recorded low scores, are compensated with the double weights applied to the NFM criteria. In fact, the layer map produced in Chapter 6 prioritised areas along the slope and just above the head waters of the catchment start to become streams, hence, superficial runoff. This score increment is also appreciable in the histograms with a marked migration of the median closed to the 0 for both broadleaved and conifer, respectively calculated as -1.496 and -1.859.

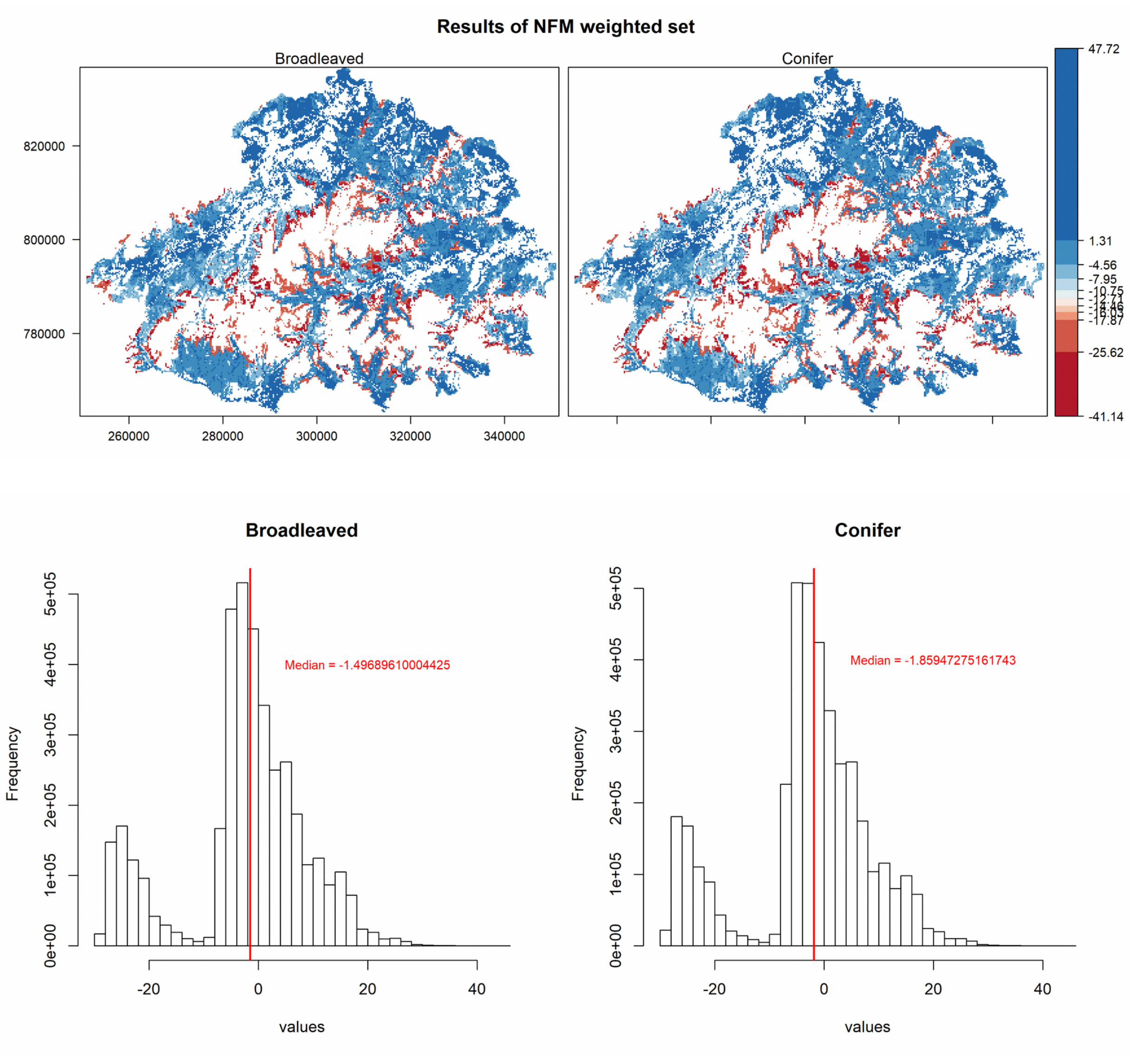
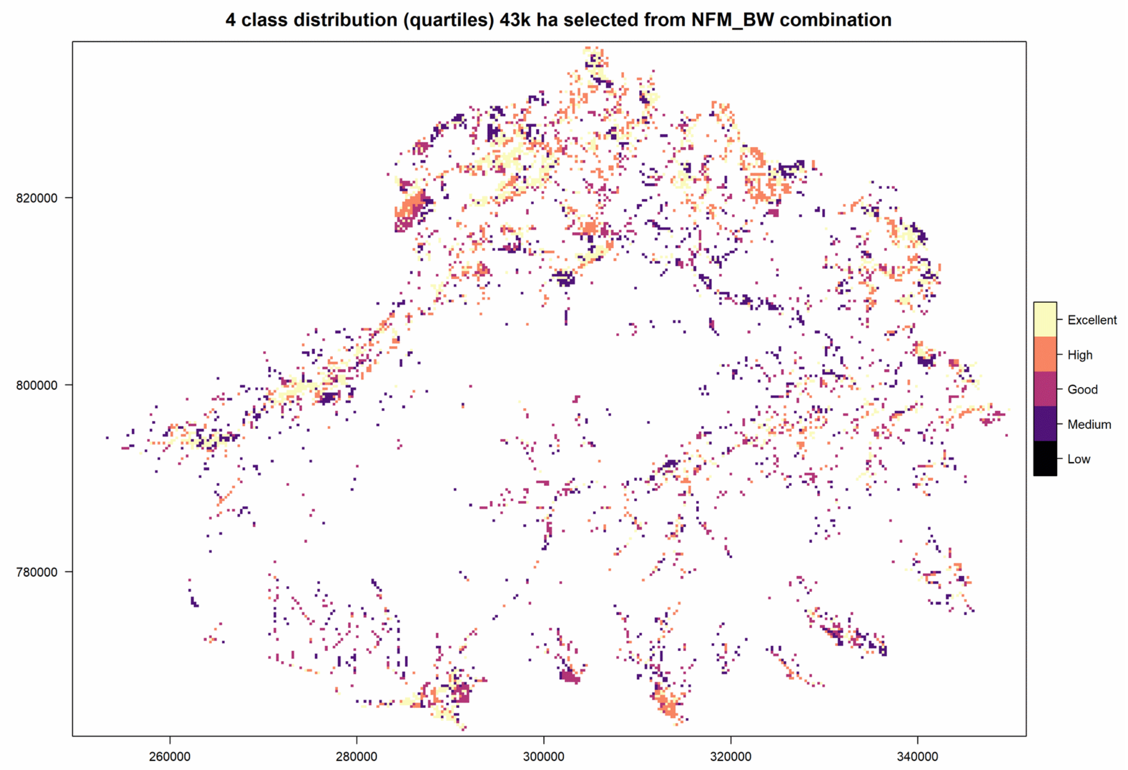


Figure 9.3. The heat maps for Priority 1 combination for broadleaved and conifer and the respective histograms.

Despite this general positive trend in scores the categorization still identify within the 43k hectares selected for each type of forest (Fig.9.4) a quite similar pattern compared to the EQW confirming that the values increase comprised the overall range of values.



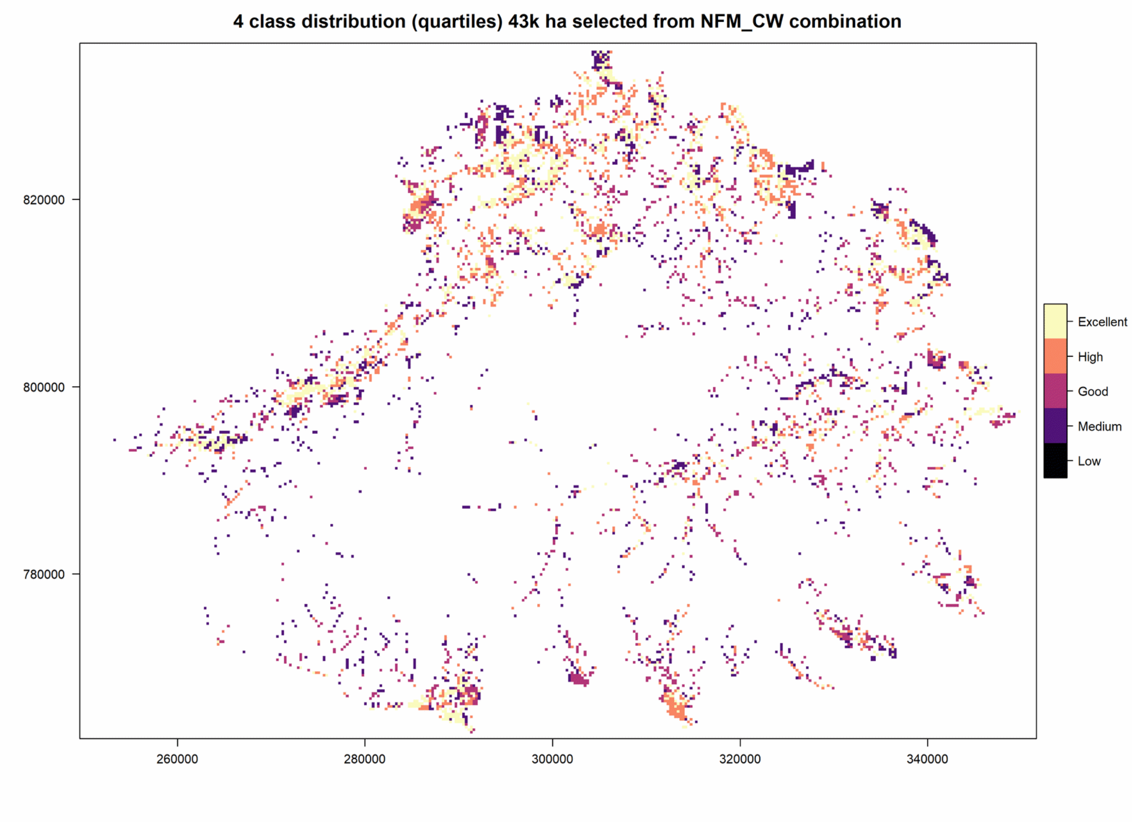


Figure 9.4. The categorization (quartiles) of Priority 1 results of the selected 43k ha for broadleaved (above) and conifer (below).

9.4.3 Priority 2 – Water quality

The results of the double weighted Priority 2 set (Fig. 9.5) as expected, identified, zonation of increase values that matched with the water quality layer produced for this work. The value increment here is localised along all the Spey valley, the lower valley of Dee and the extreme north-east of the study area. Scattered increments in values associated with the sediment layer are mostly located in the very central part of the study area, but being masked by hard constraints, do not contribute to the final results. In addition, on the relative histograms, scores identified a median of -11.201 and -11664, respectively for broadleaved and conifer.

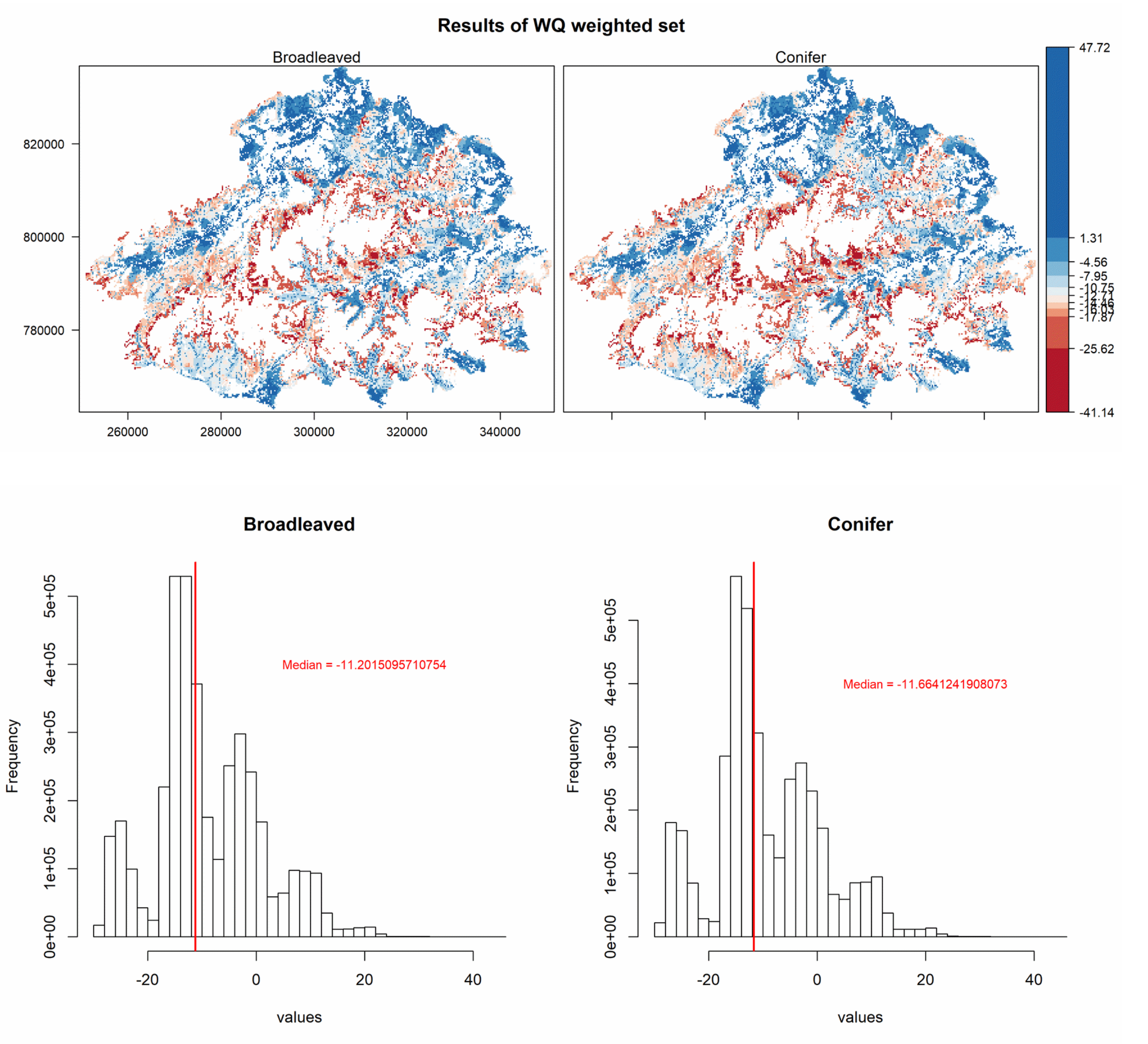
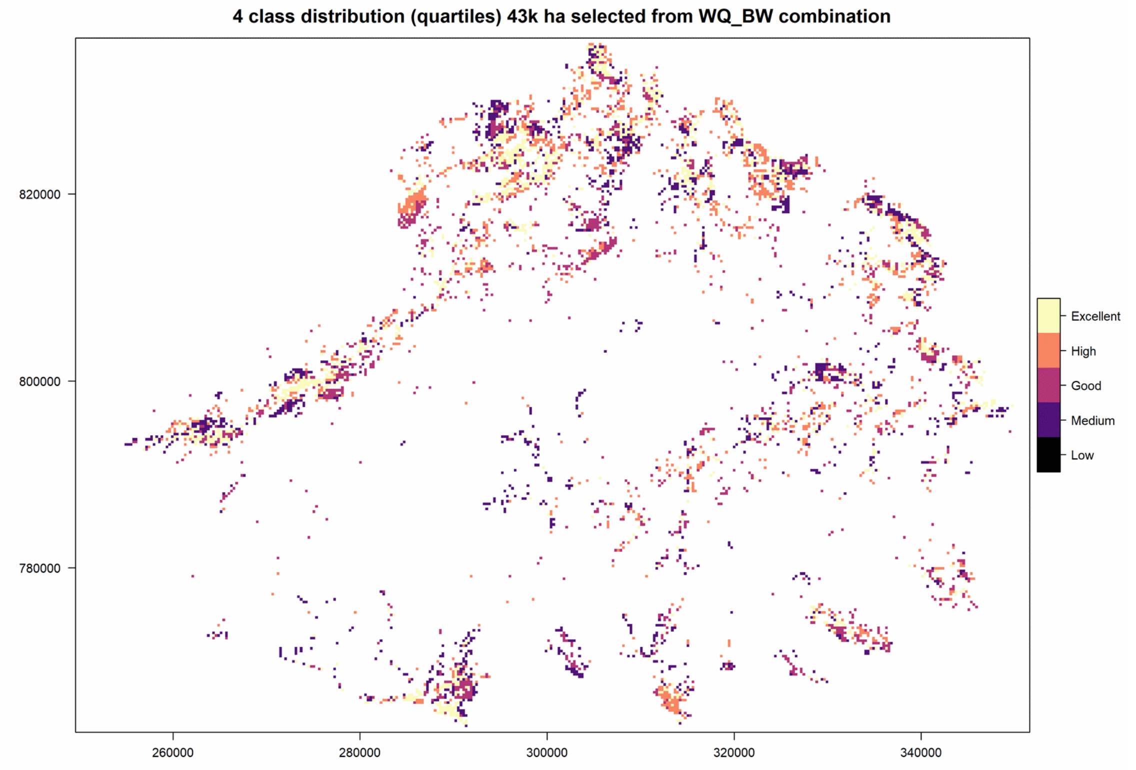


Figure 9.5. The heat maps for Priority2 combination for broadleaved and conifer and the respective histograms.

Most of the excellent and high suitability areas are still placed in the northern part of the study area, with a total selection of 44729 and 44718 ha, respectively for broadleaved and conifer.


4 class distribution (quartiles) 43k ha selected from WQ_BW combination
820000
800000
780000
260000
280000
300000
320000
340000
Excellent
High
Good
Medium
Low

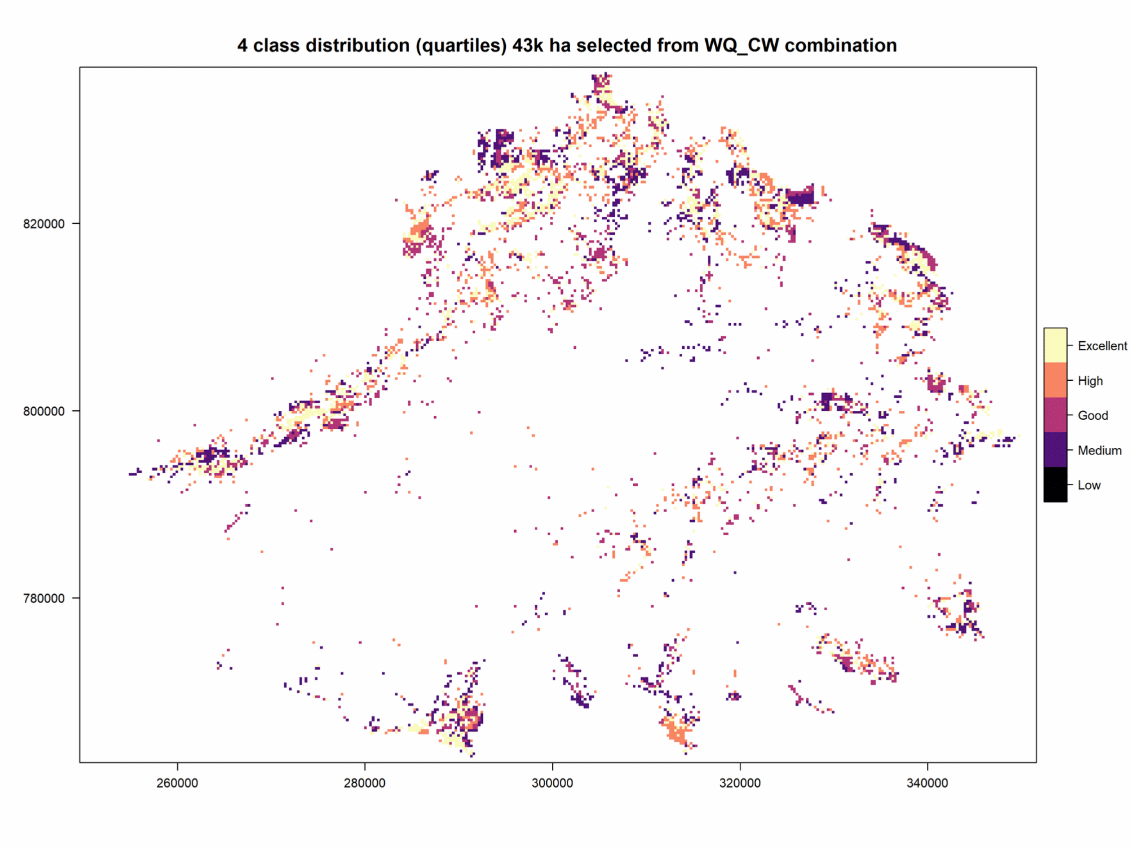


Figure 9.6. The categorization (quartiles) of WQ results of the selected 43k ha for broadleaved (above) and conifer (below).

9.4.4 Priority 3 – Ecological connectivity

The double weighted Priority 3 set (Fig. 9.7) showed, a similar zonation to Priority 2, with median of -11.442 and -11.817 identified in the histograms of broadleaved and conifer. Using alternatively and exclusively two different layers (dispersal for conifer and broadleaved) in the double weighted exercise, resulted the increment in values to be more localised along the Spey valley for conifer, while the increase affects more the Dee side and the connecting valleys between the two main streams for broadleaved.

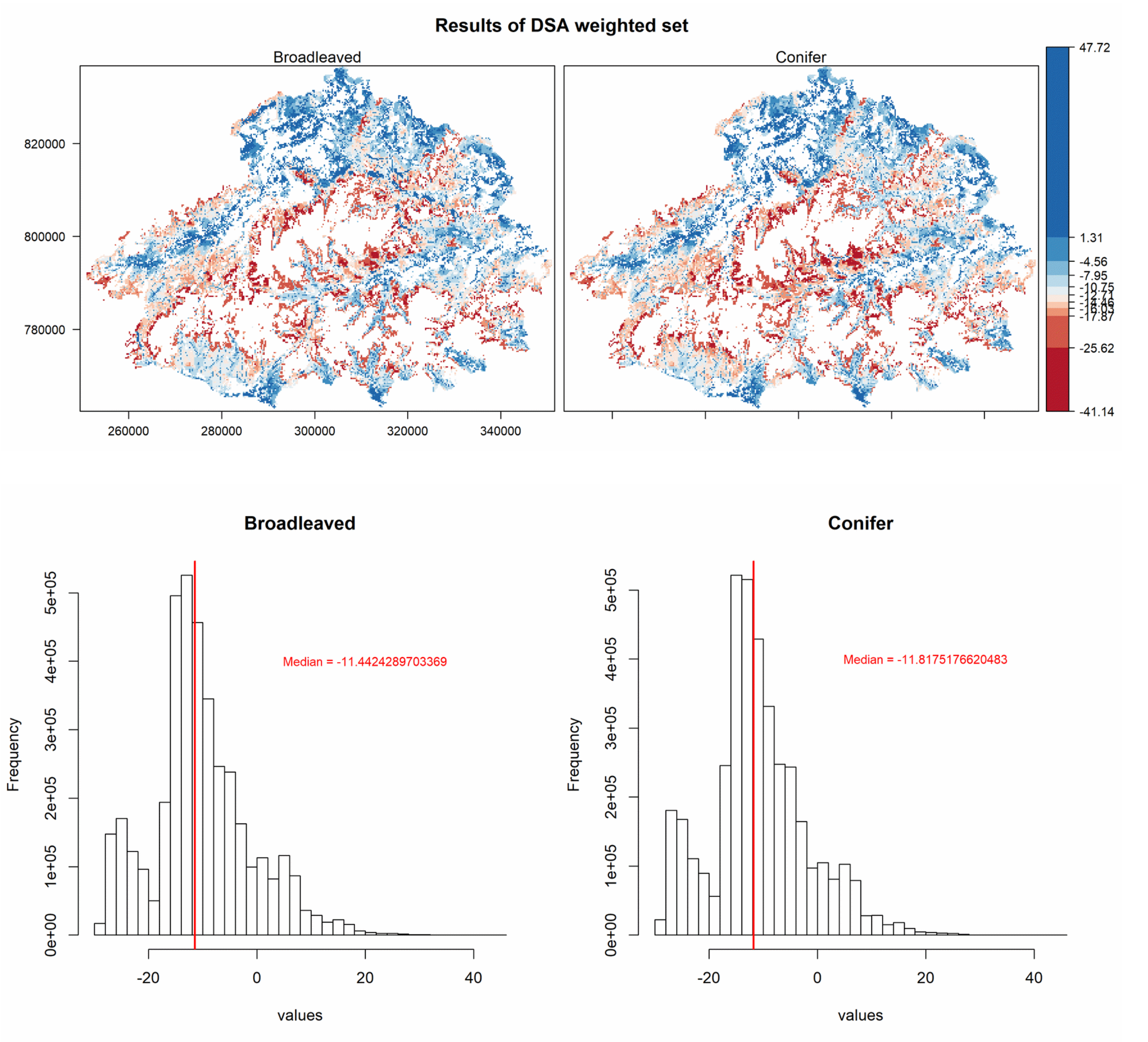
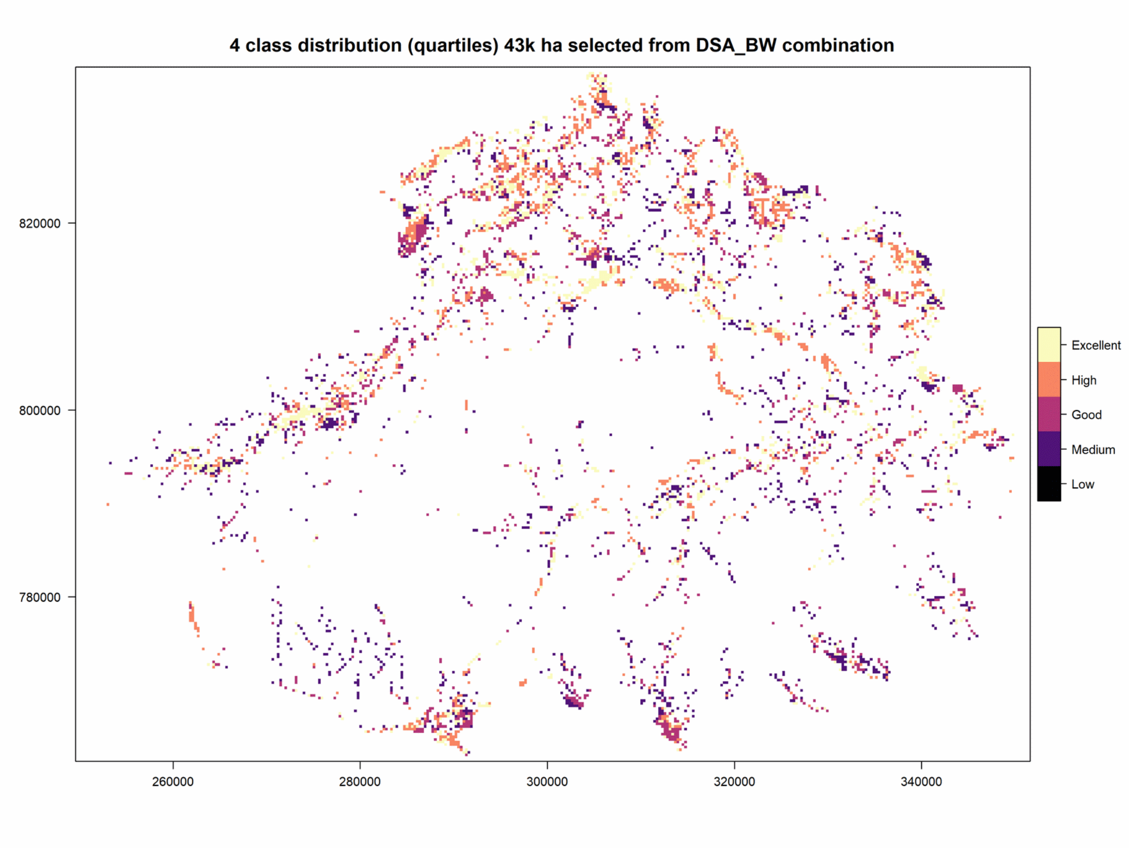


Figure 9.7. The heat maps for Priority 3 combination for broadleaved and conifer and the respective histograms.

Comparing the two categorisation here, highlights how some excellent and good category are migrating from north to south for the two type of forest because of the different setting of the dispersal avenues layers. A selection of 44688 and 44067 ha, respectively for broadleaved and conifer, was identified.


4 class distribution (quartiles) 43k ha selected from DSA_BW combination
820000
800000
780000
260000
280000
300000
320000
340000
Excellent
High
Good
Medium
Low

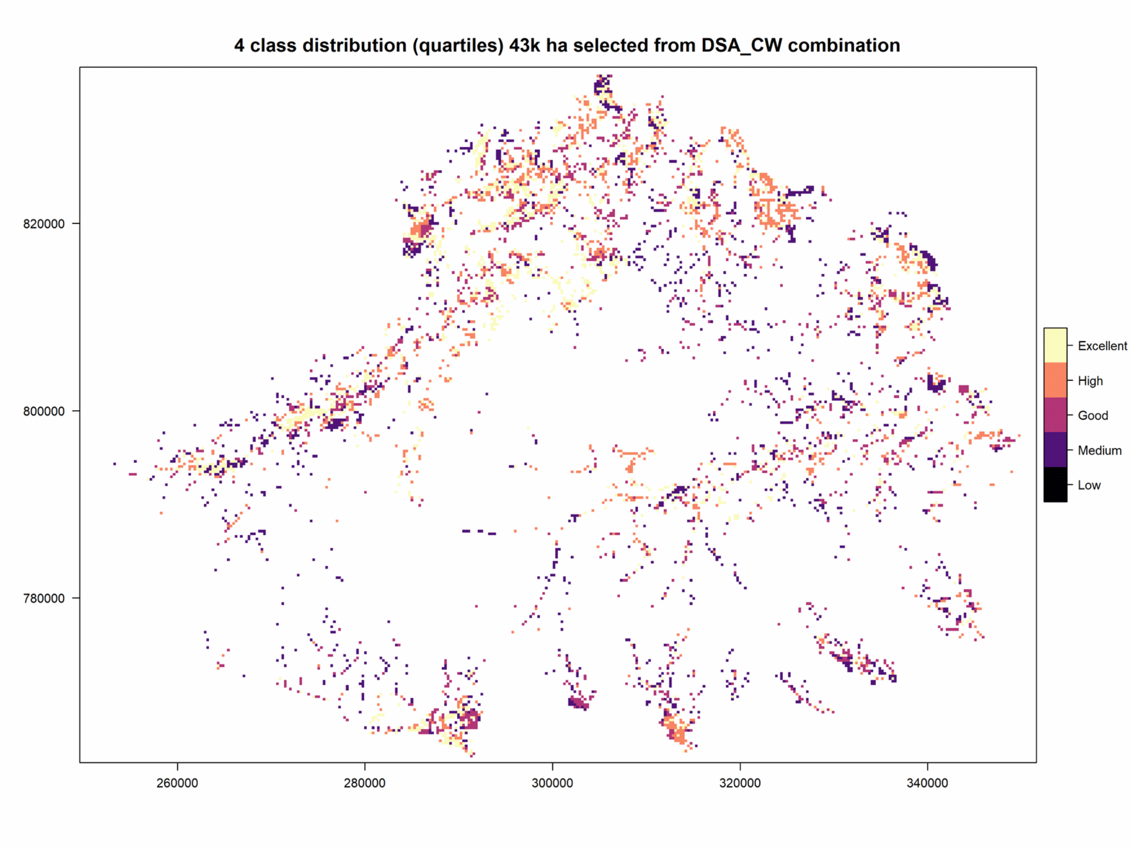


Figure 9.8. The categorization (quartiles) of Priority 3 results of the selected 43k ha for broadleaved (above) and conifer (below).

9.4.5 Priority 4 – Climate change

Finally, double weighted Priority 4 set (Fig. 9.9) showed, a general positive trend in all the zones, however the importance of this increase is not as much as the one recognised in the Priority 1 double weighted set. Besides, the application of the double weight to the carbon sequestration layer, seems to leave unaltered the central zone of the study area with some very low values. Medians between broadleaved and conifer from histograms hang within the range of -8.453 and -8.829.

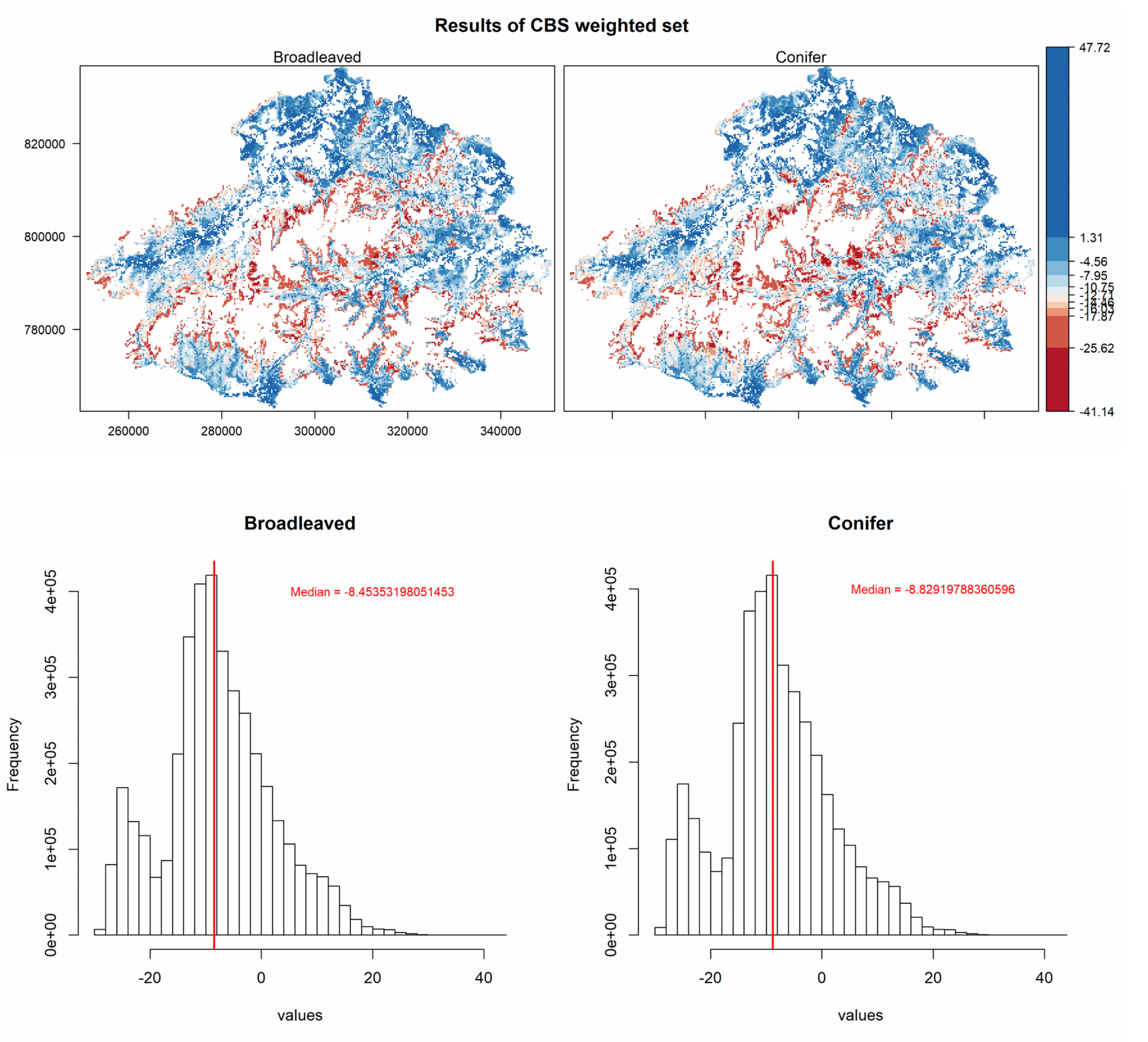
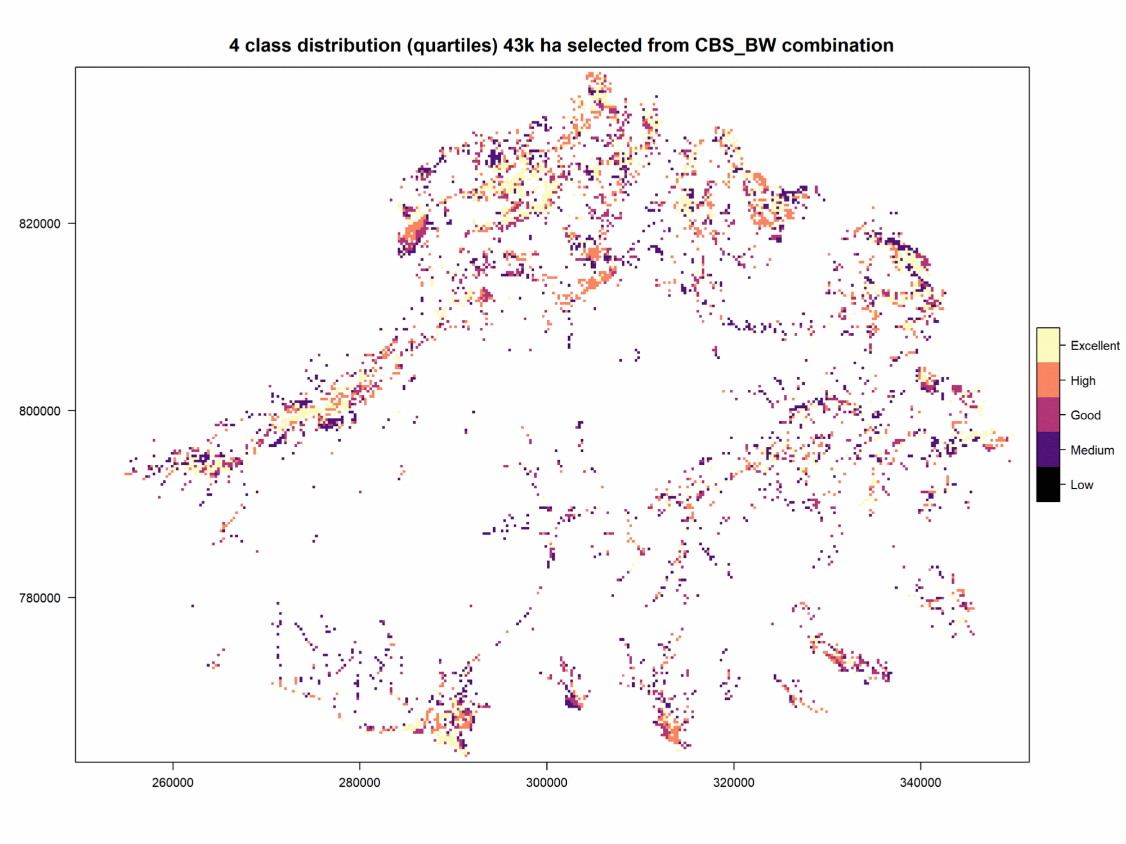


Figure 9.9. The heat maps for Priority 4 combination for broadleaved and conifer and the respective histograms.

Excellent and good categories are still placed in the upper and the lower part of Speyside, however some extreme south-west zones can pass from medium and low to good in comparison with the EQW categorisation. Priority 4 set have selected of 43920 and 44489 ha, respectively for broadleaved and conifer, was identified.


4 class distribution (quartiles) 43k ha selected from CBS_BW combination
820000
800000
780000
260000
280000
300000
320000
340000
Excellent
High
Good
Medium
Low

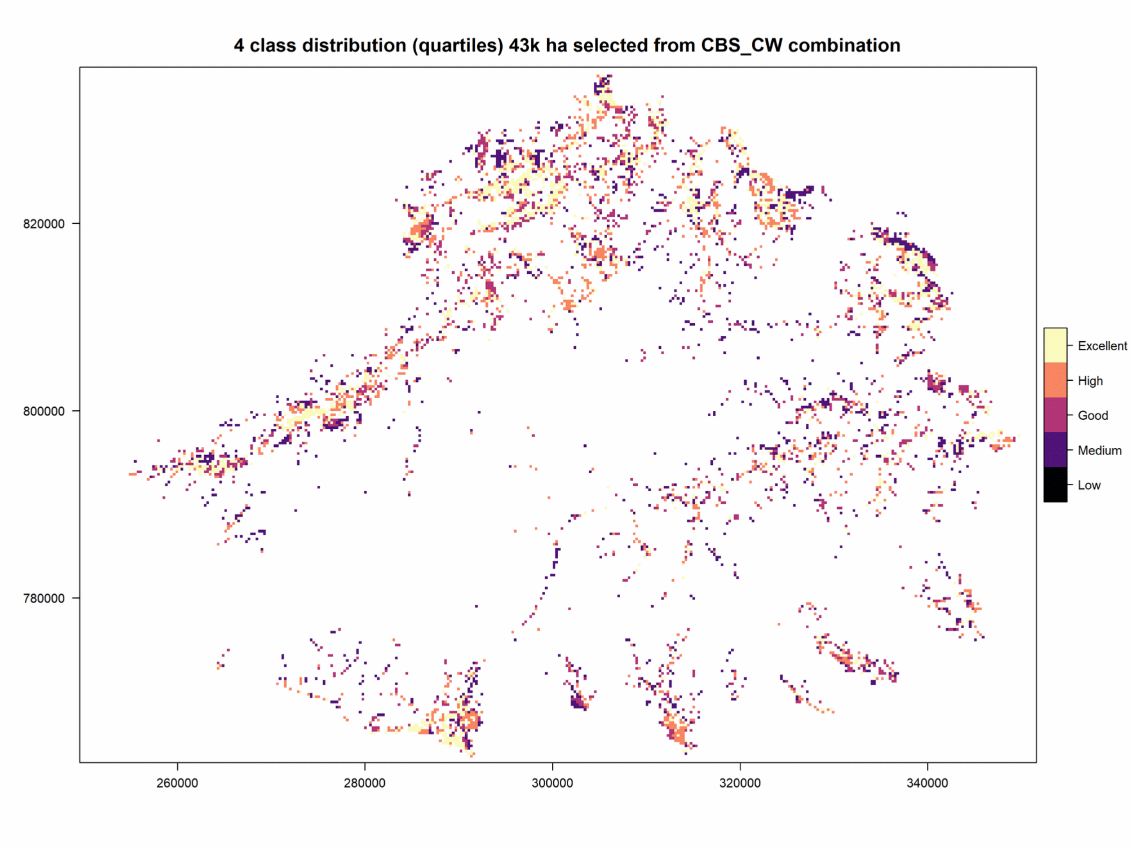


Figure 9.10. The categorization (quartiles) of Priority 4 results of the selected 43k ha for broadleaved (above) and conifer (below).

9.5 Multifunctionality and identification of hotspots

Further zonations were carried out and are shown in Fig. 9.11. The maps show that there are “quartet of win” (hotspots) areas within the CNP that can provide higher numbers of regulating and provisioning services represented by the criteria used in this study. Potential broadleaved hotspots (Fig. 9.11) are located in the left side of upper Spey valley and in the floodplain near Aviemore of the Spey river; in the high south-facing Perthshire valleys around Ben Atholl, Spittal of Glenshee and Glenclova; in the upper part of the River Dee catchment, along the valley of Cluny Water (Auchallater) and the valley of the River Gairn (north of Ballater).

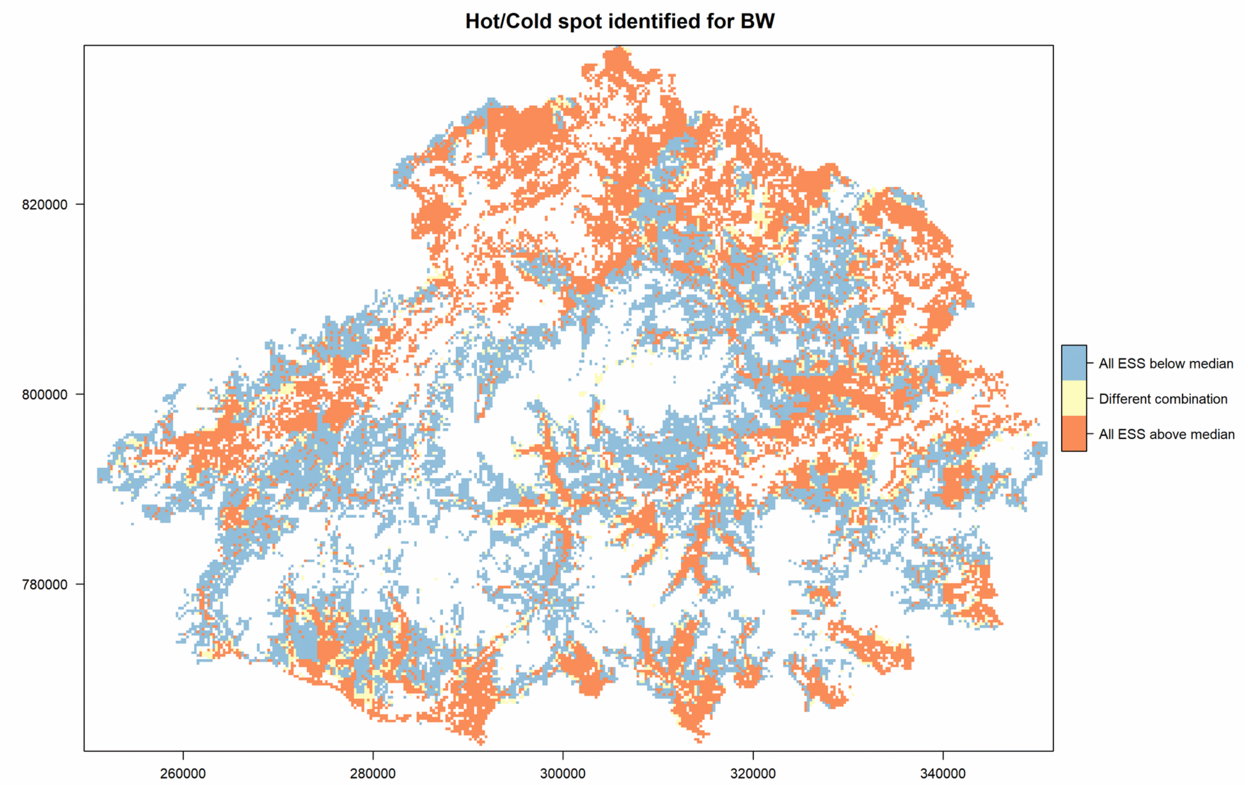


Figure 9.11. Broadleaved multifunctional hotspots (always above median) and coldspots (always below the median) derived from upper quartiles of the 4 different policy objective scenarios (Priority 1, Priority 2, Priority 3 and Priority 4, baseline scenario EQW was not included).

Most of the potential for conifer overlapped the areas already identified by broadleaved hotspots, however some there is evidence of specific differences between the two analyses. Specifically, both the middle slope of the upper Dee catchment (west of Braemar), along the Glen Geldie, Glen Dee and the glen above Bynack Lodge; also along Glen Gairn and Glen Fenzie and all the border areas along the A938 from Cock Bridge to Blairnamarrow, were suitable for enhancing the multiple benefits of native conifer.

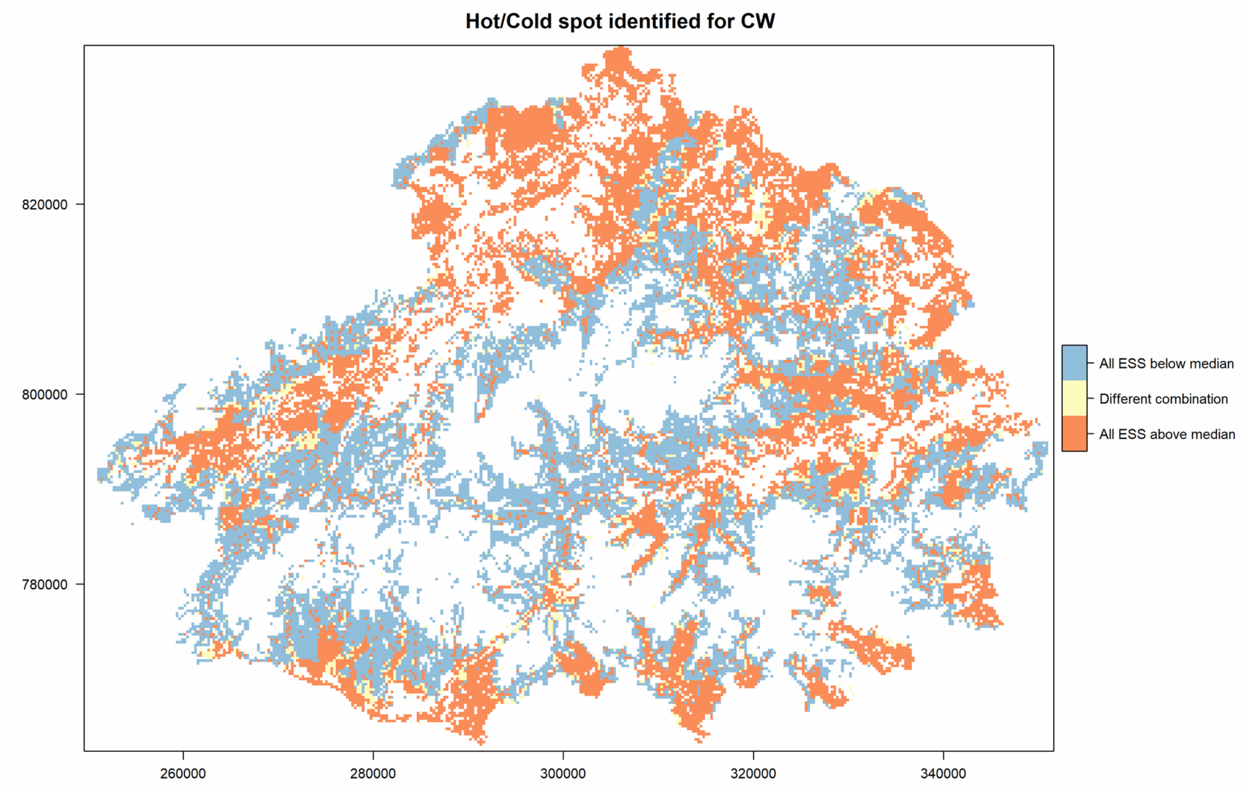


Figure 9.12. Conifer multifunctional hotspots (always above median) and coldspots (always below the median) derived from upper quartiles of the 4 different policy objective scenarios (Priority 1, Priority 2, Priority 3 and Priority 4, baseline scenario EQW was not included).

Conversely, there are also “quartets of lose” areas (coldspots), located mostly in the central part of the River Dee catchment and the western part of the study area, where planting schemes of new trees are not recommended. These areas are not contributing to increase connectivity and infiltration capacity, are areas where the effect of new forest to filter water for purification would be limited and where net gain carbon by biomass would not be high. Additionally, these areas accounts for preserving productive lands (arable), wader habitats and special conservation sites.

Finally, the quantity of hectares available for hotspots was calculated in order to verify if there is enough opportunity to satisfy CNP (national proportion) target with multifunctionality. Calculation reported a total of 112596 ha for broadleaved and 113621 for conifer, giving the decision makers a good range of manoeuvre for woodland expansion.

9.6 Introduction to land use scenarios

This section describes the development of quantitative, spatially explicit and alternative scenarios of future land use in Cairngorms National Park, which were constructed to support analyses of the multifunctionality of ecosystem services in the context of the national woodland expansion strategy (Scottish Government 2009; Reid, 2018, Scottish Government, 2021). The description of the following land use options introduces the concept of scenarios by defining them as "plausible futures" for the considered landscape. This exercise was designed to investigate the changes and, for comparison, the possible impact that could be seen as likely to occur, without being intended to provide future predictions. To evaluate the scenarios the paradigm of "carbon offsetting forestry" was applied. In a hypothetical context, where real stakeholders would be engaged, this allowed have a common value to compare the scenarios from the point of view of the net quantity of carbon sequestered by tree biomass. A number of three scenarios (A, B C) were tested, using the equal weighting (EQW) option from the sMCA work described in the previous sections of this chapter. Based on these, the land use change in the landscape was estimated and the change in potential carbon sequestration associated to these scenarios.

Here, the 43k ha were selected from sMCA analysis results for the EQW layer (see 9.3) both for broadleaved and conifer, those selection was then used to allocate the change to the land use baseline (Tab 9.2). In particular we produced 3 different scenarios to simulate an exclusive tree planting using broadleaved species (Scen A), an exclusive tree planting of native conifer (Scen B) and a mixed tree planting using commercial conifer species (Scen C). This land use change is equivalent to ca. 5% of the total CNP area and the main land use class affected by this change was improved grassland. Arable and semi-natural grassland were also affected but in a smaller proportion (ca. 1% each), whilst other land uses quantities remain constant.

A complete set of maps for each scenario and for each land use type is discussed in the following paragraphs. A comparison of the direction of land use change for the different scenarios and land use types shows that improved grassland land use decreases in all scenarios. The changes in quantity are equally applied, however the original setting of the baseline indeed affected the final results. For instance, the high concentration of conifer along the River Spey and the use of the distance from woodland as a criterion, have driven the land use change to have more conifer in the north part of the protected area.

Scenario	sMCA	Target woodland	Perc before (%)	Perc after (%)	Main LU lost
Scen A	EQW_BW	Native broadleaved	2.84	8.77	Impr. grass
Scen B	EQW_CW	Native conifer	5.07	11.01	Impr. grass
Scen C	EQW_CW	Commercial conifer	5.38	11.32	Impr. grass

Table 9.2. Main characteristics and the general quantity trend of the scenarios produced.

The scenarios introduced above anticipates the concept of scenarios by defining them as "plausible futures" for the region. This exercise was designed to investigate the changes and the possible impact that could be seen as likely to occur, without being intended to provide future predictions. To evaluate the scenarios produced the paradigm of "carbon offsetting forestry" was applied. This allowed the stakeholder simulation have a common value to compare the scenarios from the point of view of the quantity of carbon sequestered by tree biomass due to the afforestation. The following table 9.3 summarized the type of trees considered by each scenarios.

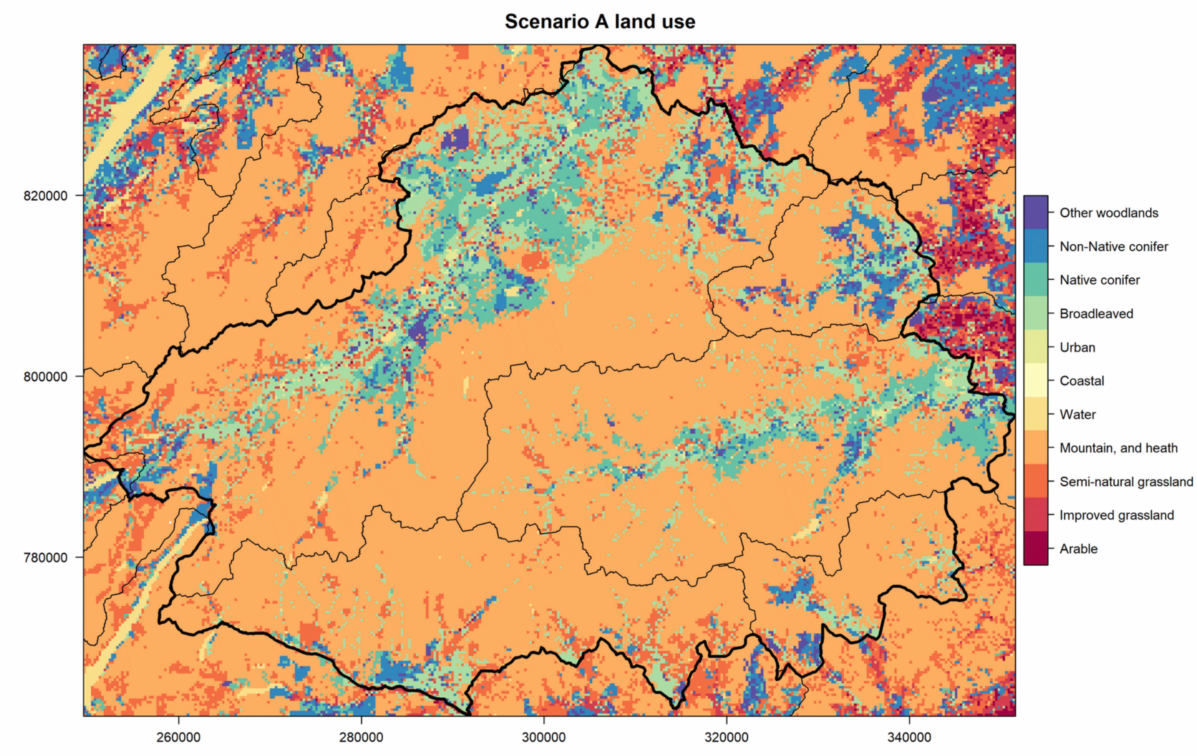
Scenario	Common Name	Latin name
	Ash	*Fraxinus excelsior*
	Common alder	*Alnus glutinosa*
	Downy birch	*Betula pubescens*
Scen A	Pedunculate Oak	*Quercus robur*
	Sessile oak	*Quercus petraea*
	Silver birch	*Betula pendula*
	Wych elm	*Ulmus glabra*
Scen B	Scots pine	*Pinus sylvestris*
	Douglas fir	*Pseudotsuga menziesii*
Scen C	Lodgepole pine	*Pinus contorta*
	Sitka spruce	*Picea sitchensis*

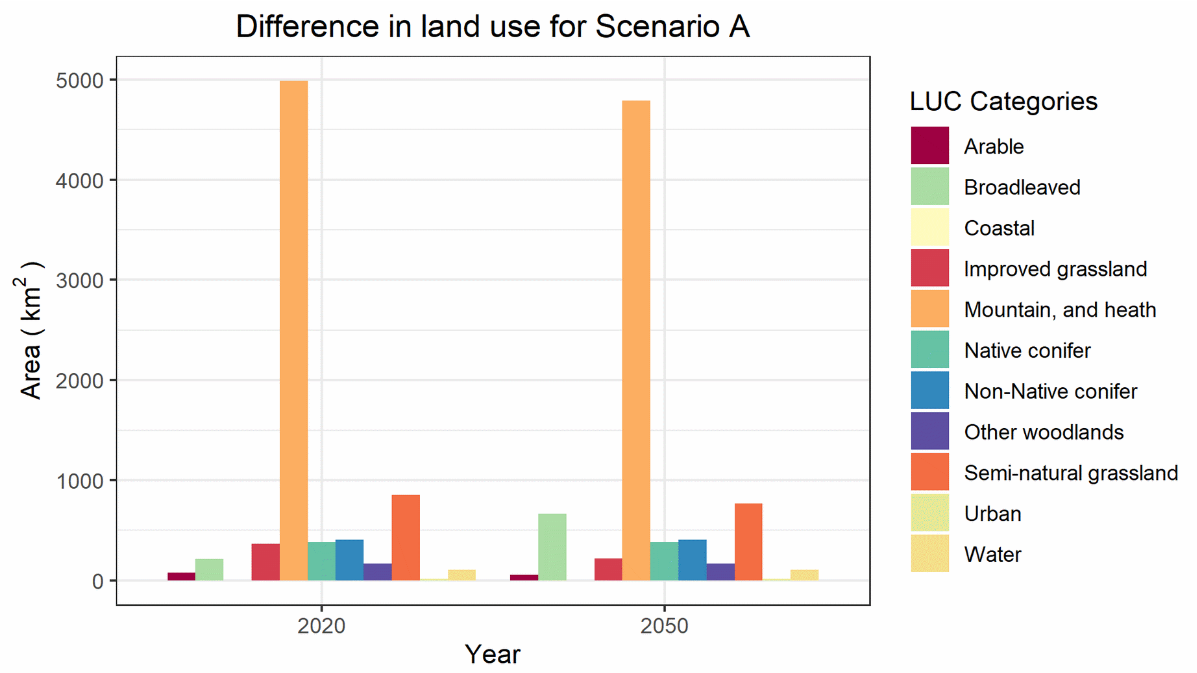
Table 9.3. List of the species used to calculate biomass produced applied to each scenario.

Scenario C focussed on continuity of the traditional planting patterns (non native conifer are currently equal the native conifer with 5.38% of the total area), with an emphasis on direct goods such as timber. The other two scenarios concentrated on emphasise ecological services, such as biodiversity (native trees deliver more species richness).

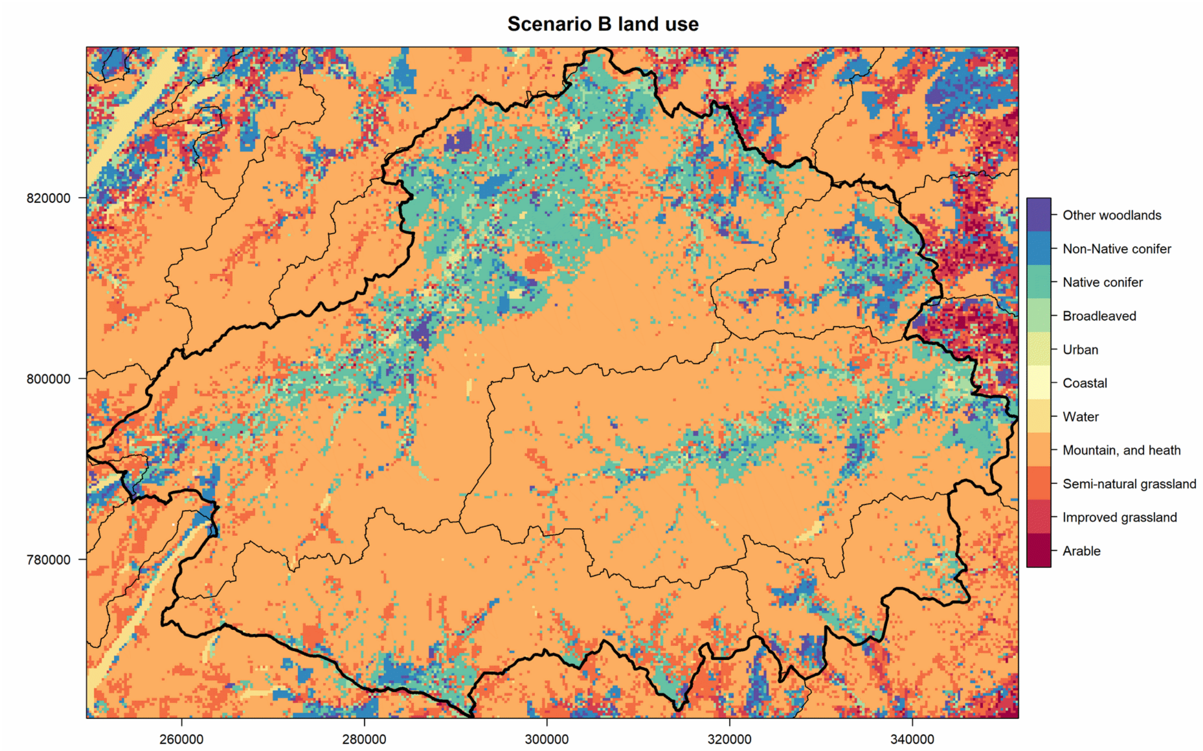
Scen A and B also acknowledged the importance of others indirect ecosystem services such as soil protection, water flow regulation and carbon sequestration, and other ways to promote the health and vitality of the whole landscape.

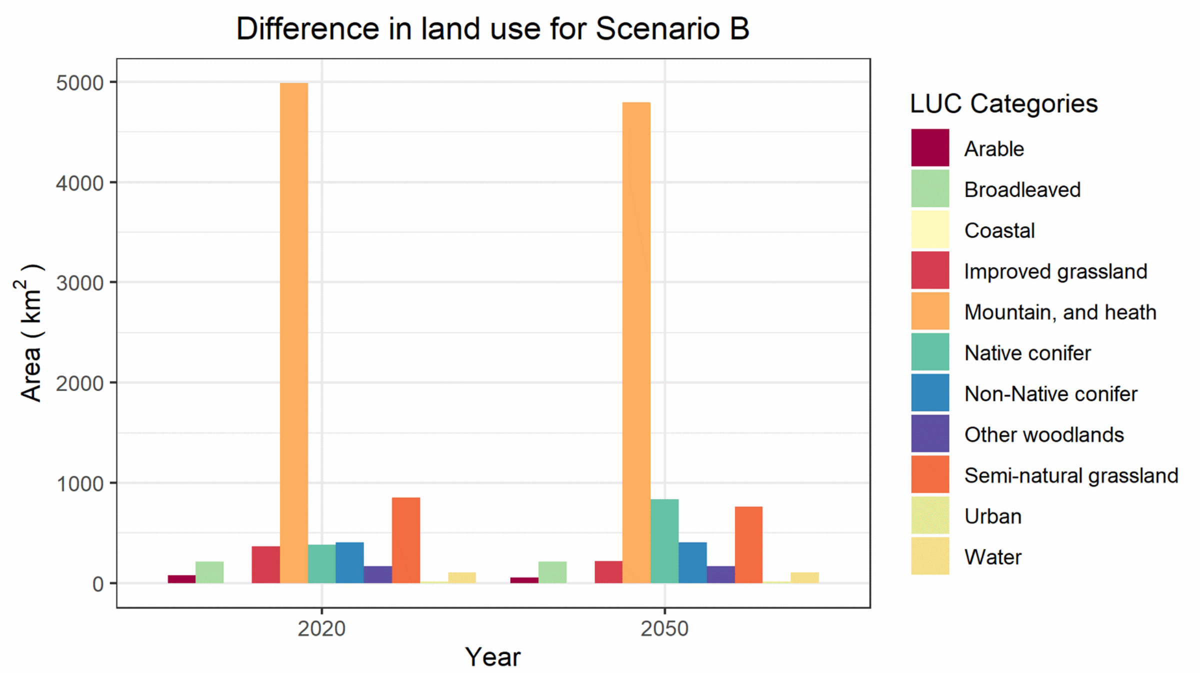
9.6.1 Scenario A



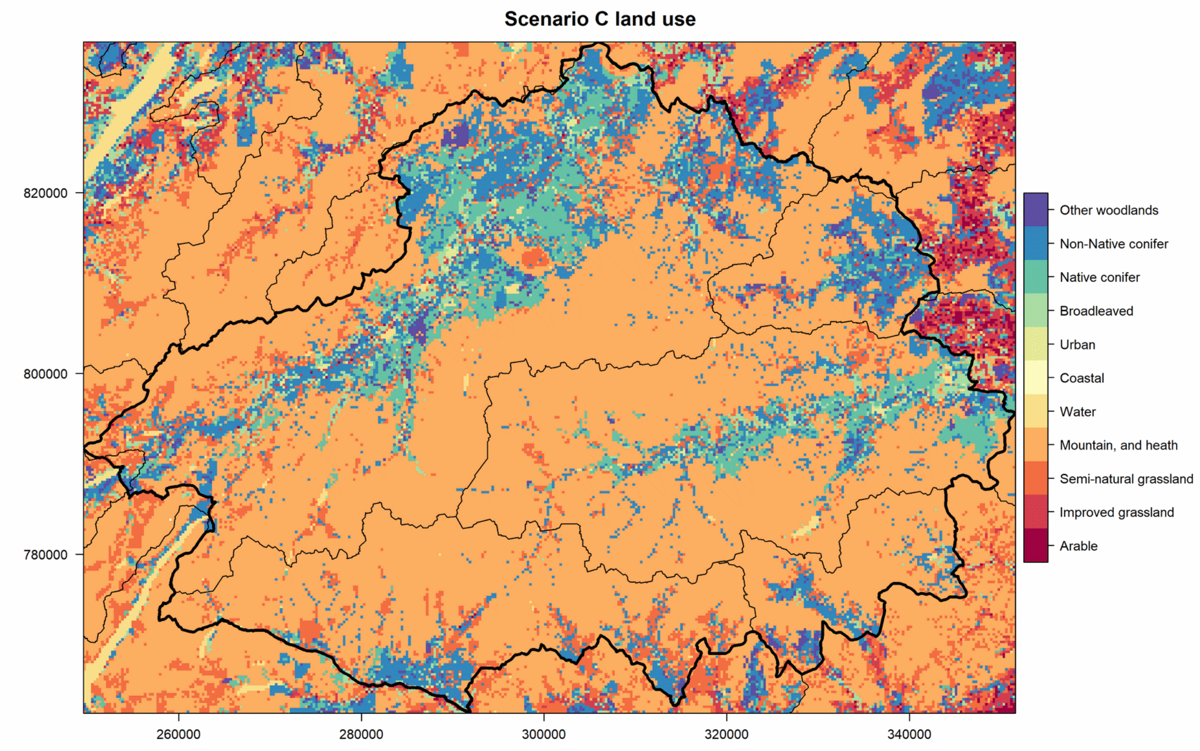


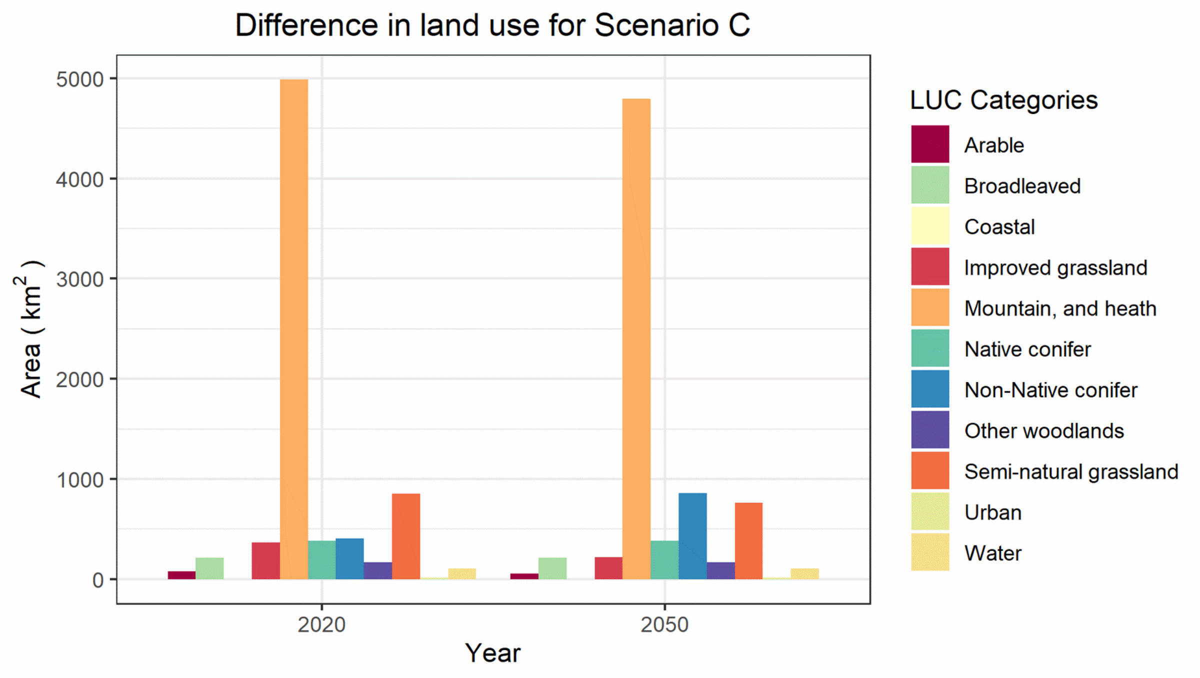
9.6.2 Scenario B





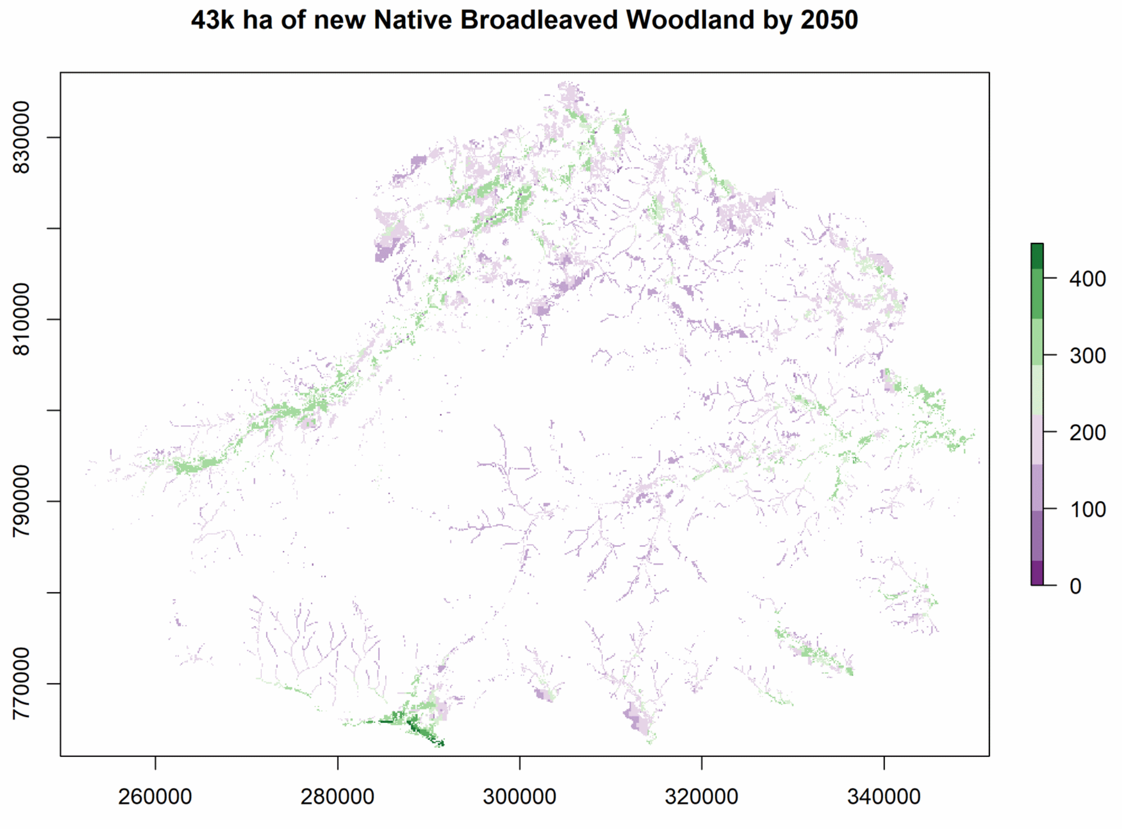
9.6.3 Scenario C



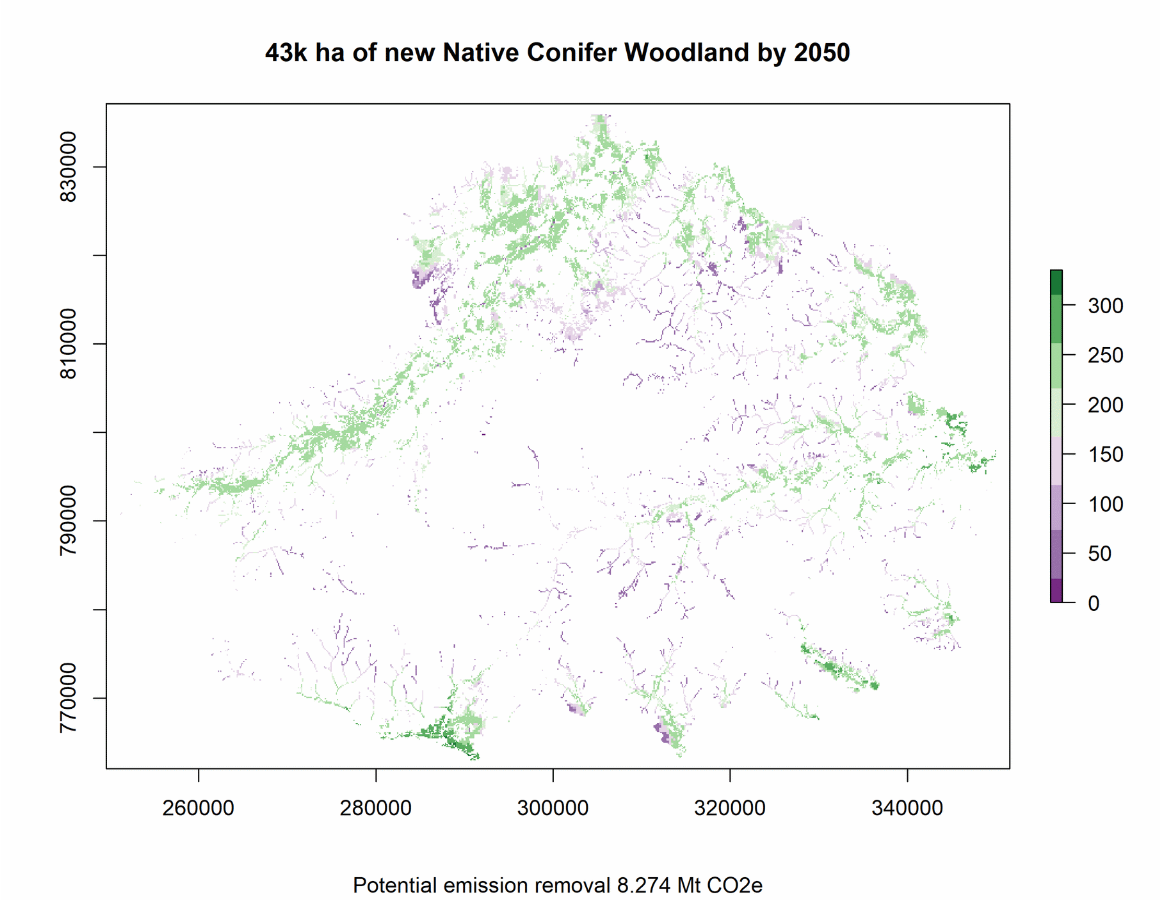


9.7 Scenarios results

The following maps represent the relative carbon sequestration by the tree biomass for each land use change scenario. The carbon sequestration figures as the total net carbon were calculated using the estimated biomass (tCO2-eq/ha) using the highest tree species YC possible for 2050 (see chapter 4). The total sum of the new biomass was then calculated for the 43k ha of new tree planting in each considered land use scenario. The effect of the afforestation applied is quite evident with Scen A holding a potential to sequester 9.495 million of tonnes of carbon dioxide equivalent (Mt CO2-eq). Interestingly, this potential decreases to 8.274 Mt CO-eq using only native conifer in Scen B, while Scen C recorded the highest potential with 14.524 Mt CO-eq.



Potential emission removal 9.495 Mt CO2e


43k ha of new Native Conifer Woodland by 2050
830000
810000
790000
770000
260000
280000
300000
320000
340000
300
250
200
150
100
50
0
Potential emission removal 8.274 Mt CO2e

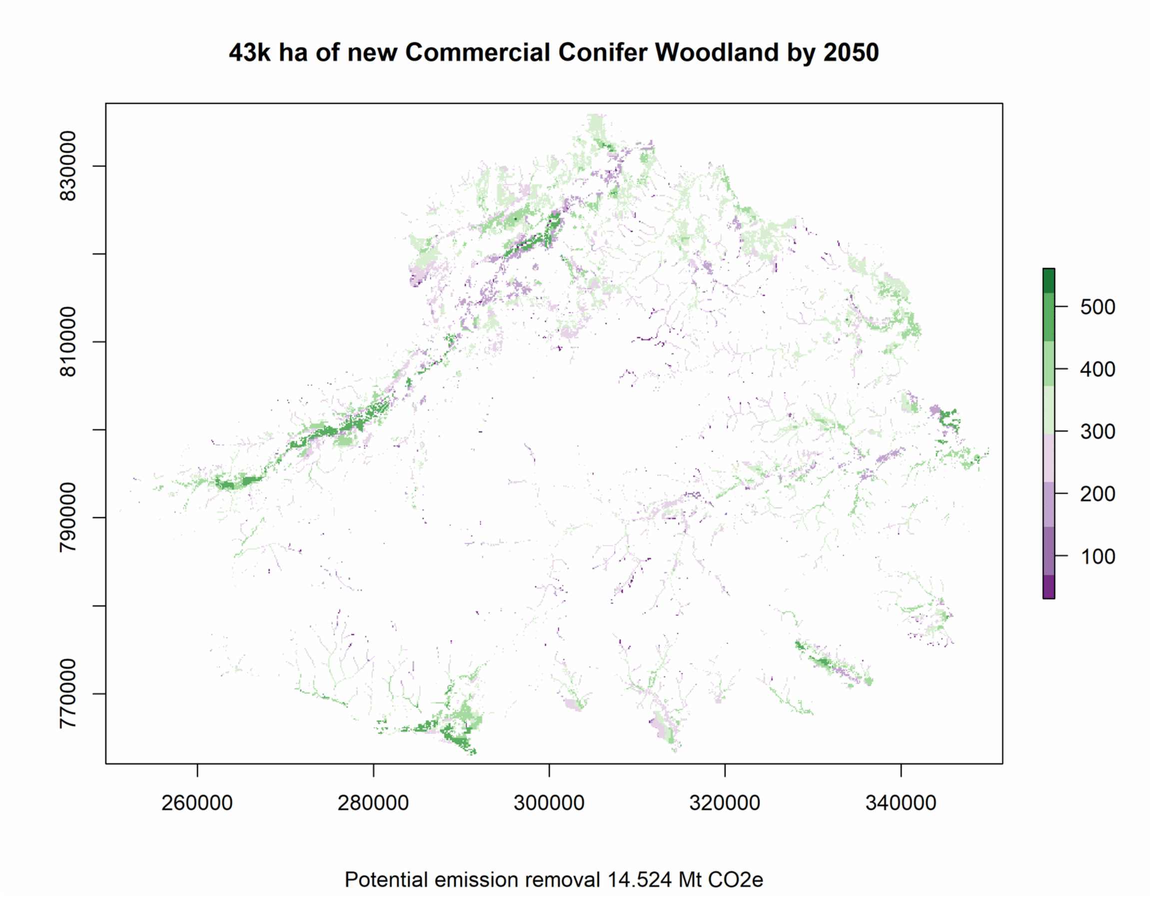


Figure 9.13. Carbon sequestration maps after afforestation for scenarios.

These results show that, using individual criteria, offer limited information to decision makers, with possible detrimental (and not planned) consequences. When using weighted combinations, the plantings are actually located preferentially in high-benefit areas. However, same areas can be expanded with different species concluding with different results. In fact, an expansion oriented to commercial trees (Scen C) can store the highest carbon quantity, however this option will leave (and further increase) the isolation of the native forest habitats. Bear in mind, that 14.524 Mt CO-eq is just 0.006% of the Scottish global annual footprint (ca. 70 Mt CO-eq) we can conclude that such a profit in carbon sequestration is not worth increasing the isolation of native forest species and a more conscious option (Scen A or Scen B) must be considered.

Summarising, other scenarios derived from emphasising different criteria in different ways can be suggested. We tested, among others, three scenarios based on the social and ecological understory linked to the forest sector. This change leads to landscapes with differences in carbon sequestration up to 1.5 times more (native versus non-native), but with little spatial networking and lower ecological value. In fact, would be interesting to explore more scenarios using the real engagement and participation of decision makers to identify where the most controversial conflict are located and how the these actors would solve the compensatory decision rules in the context of the forestry for climate change.

Chapter 10 – Discussions

10

10.1 General discussion

The definition of priorities based on specific appropriate criteria is crucial in forest conservation planning. The landscape attributes and indicators (criteria) for sustainable forest strategies can be selected among universal recognized standards under the umbrella of "services" that can be interpreted by local communities and decision makers either as part of mutual global effort for regulating (carbon stock), or to search for advantages that favour specific local function (water quality) (Mrosek, 2001). Carbon sequestration is increasingly and widely considered an important criterion for forest planning (Krcmar et al., 2005, Diaz-Balteiro and Romero, 2003, Paul et al., 2003). One aspect of such an approach that has been overlooked in much of the discussion concerning carbon forest sinks, but has recently drawn more attention, is the management of expectations for carbon uptake in the light of other forest services such as biodiversity. Choice of species for afforestation needs to trade-off carbon sequestration with potential soil release. The choice of tree species can enormously affect connectivity, biodiversity and understory plants with associated wildlife species.

The carbon criterion used in this study includes the interpretation of soil preparation practice as sensitivity but also employed a sophisticated carbon balance calculation. The paper that forms chapter 4 of this study, has shown that to avoid a net release of carbon, accounting for spatial heterogeneity and time evolution of tree species is crucial. Therefore, soil-based limitations need to be considered, as well as practices to ensure that any afforestation incentives don't produce undesired effects.

Modelling complex ecosystem services processes is constrained by data and resources available, therefore sometimes the application of simple models that can provide a decent level of representation with little available data is the most sensible and pragmatic choice to follow. Such models like InVEST were used to define priorities in this study, highlighting model limitations, but containing enough information to provide the level of choice required by decision makers. Nutrient and sediment export results defined in chapter 5 have shown that it is possible to offer evidence for the potential to achieve multiple objectives through targeted land use change despite the simplistic representation of bio-physical process.

Experimental evidence from upland control plots versus afforestation areas with broadleaved woodland, have shown that NFM measures could enhance soil infiltration rates and reduce bulk runoff coefficients (Iacob and Rowan, 2017). Chapter 6 has shown how is possible to narrow down the level of further investigation and apply complex modelling approaches once the strategic preliminary analysis has defined priority areas at national scale. GIS and spatial analysis applied to NFM was a useful tool to outline whether woodland expansion in Scotland could make a significant contribution to tackle flood risk considering a whole-catchment approach. These results are in line with other relevant findings from other scientific works.

It is widely recognised that habitat fragmentation is one of the major threats for biodiversity and ecosystem conservation (Lathrop and Bognar, 1998). These threats can be exacerbated by several land use activities such as forest management for commercial timber production with additional impact on wildlife (Lamberson et al., 1994). The exercise of modelling ecological networks of heterogeneous landscapes can help in understanding of such management activities on wildlife dispersal, hence assist forest ecosystem planning process (Vuilleumier and Prelaz-Droux, 2002). Circuit theory was applied to represent the current spatial connectivity (*sensu latu*) for native broadleaved and conifer forests. While chapter 7 has not provided evidence that outcomes from the application of Circuitscape can increase species populations, the analysis provided two proxy maps to be used as an index to inform and address conservation issues in managed forests, such as in the study area.

In this work, a landscape scale approach has been used with regard to the potential contribution of woodland expansion to the overall objective of sustainable management but specifically focused on four overarching ecological objectives, namely carbon sequestration, flooding mitigation, water quality and biodiversity. It is necessary to bear in mind that although many national parks still focus on conservation, the criteria and indicators for sustainable forest management are not specifically developed for conservation planning purposes. Therefore, additional criteria, other than those four combined in this study, were considered being relevant. The GIS-based sMCA approach used here is so simple and flexible that any number of criteria and indicators can be employed.

However, the planning of natural protected areas must take into account the difficulties some stakeholders may have to express their concerns and points of view during the weighting exercise.

10.2 Limitation and comparison

Clearly, the five combinations of maps proposed for the four criteria are not exhaustive and do not represent a universal value that can be extrapolated outside the landscape scale used in this study. However the methods applied in this study area represent a productive effort to stimulate discussion and help decision makers to consider valid options for planning forest in the study area. The zonation proposed here goes well beyond the zonation made by CNP in its Forestry Strategy 2018 (CNPFS – Figure 10.1) which is based on the Native Woodland model (Towers et al., 2004) only. The criteria endorsed here represent very important considerations in developing landscape plans and have not been included in previous studies.

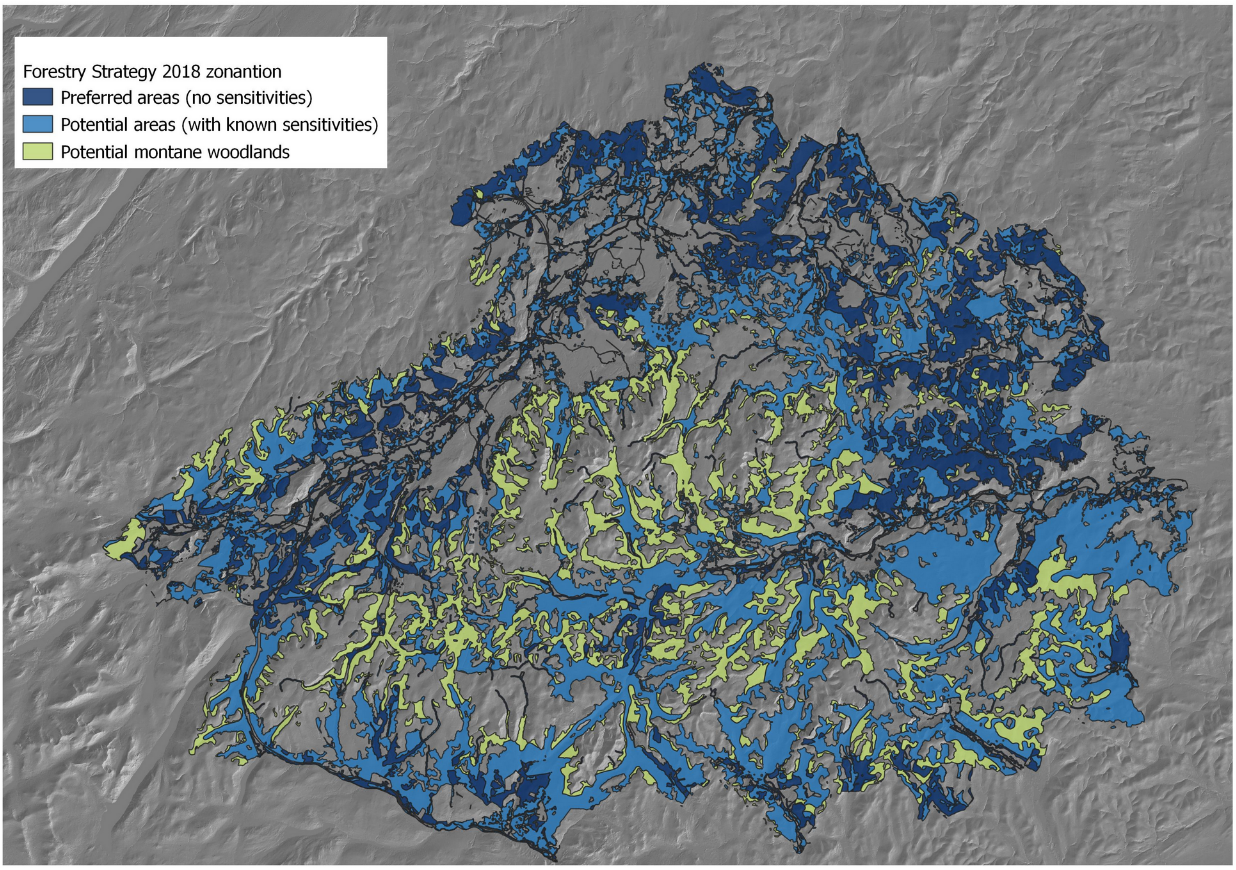


Figure 10.1. Zonation from Cairngorms National Park Forestry Strategy 2018.

Being aware that an "objective" point of view does not really exist in the forest planning procedure, further limitations in our approach can be recognised in the choice of the use of the quartile cutoff intervals of histograms for the definition of the hotspots. Even though this can be argued as arbitrary, the same level of criticism could also be applied to the definition of "preferred" area in the Forestry Strategy 2018 which assumes an indicative spatial approach and a straight direct dependency with the single criteria used. As a matter of fact, the definition of a scored suitability map offers, despite some overlap, a better spatial targeting of potentials for new woodland schemes. These potentials are well represented by the multi-functional hotspots (Fig. 9.4 – 9.10) which can help target areas for woodland expansion that be studied in detail in the field The GIS-based sMCA thus helps to reduce costs and time involved in the early stage of the planning process.

As well as the spatial aspect involved, this study has shown how forest planning options can be produced to meet national greenhouse gas (GHG) emission reduction targets and be quantitatively expressed (Mt CO2-eq/ha) allowing the incorporation of decision makers (simulated), into the conversation. Here, the use of a compromised (EQW) scenario can represent a social consensus that may help in preventing and reducing conflicts between the protected areas stakeholders (private and public). This is important, because potential woodland planting areas are likely to play an important role in the achievement of climate targets as well as the economic development.
The quantification of potential carbon sequestration with the three land use scenarios is a spatially targeted approach which provides a way to maximise GHG reductions relative to the local context. Hence, recommendations that go beyond simple spatial representation of sensitivities, can be suggested to improve environmental policy measures. These kinds of progressive investigations have been proposed as a way to reduce misunderstanding, potentially contributing to policy failure and misuse of offsets (Brown, 2020).

An additional conclusion from this study can be made regarding the general concern when potential carbon sequestration gains are prioritised (Scen C) through woodland planting using fast-growing non-native species. Bear in mind that potentially all the planting scenarios produced are likely to happen on grazing land with important socio-economic consequences and trade-offs, but the involvement of commercial species to cover this expansion risks increasing the isolation of native patches of forest rather than enhancing a multifunctional woodland transition.

Chapter 11 – Conclusions & Recommendations

11

11.1 Conclusions

In this work a detailed example of transformational challenge in tackling forest multifunctionality in the land use sector has been provided. The work presented in this thesis has shown that there is enough space to allocate new trees in the Cairngorms National Park to help Scottish Government to meet climate targets over the next 30 years. However, those areas that have been identified using sMCA need to be carefully chosen to avoid a net release of carbon. For this reason, the way suitability is defined is very important. Baggio C. et al., 2021 (submitted) highlights how crucial it is to consider important limitations like soil type the ground preparation practices to use before planting to ensure afforestation schemes don't produce undesired effects.

Further, the thesis demonstrated the use of an weighted linear combination (WLC) to combine attribute maps aiming to produce a suitability map that can simultaneously deliver water quality, flooding control, habitat connectivity and carbon sequestration benefits. Although, we have identified a quantity of potential multifunctional areas (hotspot) were identified, the analysis showed also that potential trade-offs with other land uses (improved grassland used for livestock productivity) have to be considered, confirming that a consequence of taking into account various dimensions simultaneously means that it may not be possible to optimise all the objectives at the same time (Munda, 2002).

With the time window narrowing for reducing global carbon emissions to safe levels (Höhne et al., 2020) it is important to contextualise this work and its concluding recommendations. In this work, even thought the time dimension was not specifically addressed, future ecological shifts linked to climate change, were considered. It should also be noted that strategic plans need support from political agenda extending over many years as forestry actions have long-lasting effect on economic, ecological and socio-cultural considerations (Kangas & Kangas, 2005). Therefore, such forestry strategy plans should not just cover large areas but also be monitored over long time periods.

In the past, the failure to heed recommendations like these has meant that woodland expansion and climate changes targets have not been achieved (Muñoz-Rojas et al., 2015), despite national initiative like "the right tree in the right place" (Forestry Commission, 2010). This hoped-for land use transformation has been bottlenecked by the complex relationship between policy and planning framework at national level. In national parks where sustainable development has started to shift from conservation only mechanisms to a cross-sector partnerships (Forsyth, 2010), environmental policy can be diversified in favour of local concerns, including the private sector. It is therefore important, that forest planning frameworks goes beyond simple indications such as CNPFS, and develop appropriate mechanisms to coordinate diverse policy goals that could offer real opportunity for improvement.

The sMCA approach presented in this thesis, endorses this complex relationship by trying to address challenging targets such as mitigation of climate change by comparing alternative plans in regard to all chosen aspects of interest (criteria). Based on these comparisons, potential groups of decision makers can make a comprehensive comparison of the alternative plans, taking into account all the interest factors for the new forest area, and other criteria affecting the choice of the woodland expansion plan. However, this process should be carried out through local stakeholders consultation to avoid potential risk of failing to recognize individual opinions, needs and wishes. This thesis has demonstrated a decision-support method for the multiple-criteria evaluation and for the comparison of alternative forest expansion plan which can be used for further planning exercises and analysis at a more detailed level.

Although this study has been contextualised in a policy framework which is increasingly framed in terms of 'multi-functionality', the land sector in the past had real difficulties in facing the big challenge of enhancing land use synergies and minimisinge conflicts. Poor capability land areas were often designed to expand conservation benefits (woodland expansion) while the highest quality areas were often protected for food production (Slee et al., 2014). In this work, we intentionally left this focus to one side, because we think this should be specifically assessed through a much more intense debate between stakeholders, which is not specifically the subject of this study. However, the results from sMCA offer the chance to slightly touch on this topic. As previously stated, the analysis has shown that most of the suitable areas for newly planted woodlands are mostly represented by improved grassland, thus opening up potential for future discussion on the land use conflicts thereby arising.

My personal concern is that recently forest ecosystems in general, and more specifically protected areas, are receiving a lot of pressure from various fronts, because they are seen as the silver-bullet to mitigate carbon emissions, as the solution for biodiversity conservation, as the alleviation of flooding. The reality is that there is a very little benefit in terms of carbon sequestration that woodlands can achieve under the current national strategic targets, and at the same time, the value of biodiversity conservation may be not recognised as important if the individual landowners making land use decisions are unable to capture the economic benefits. Hence, I believe that land transformation should be realised through a more intelligent use of incentives. Defined priorities can be achieved by introducing incentives schemes to "compensate more who deliver more", to those who adopt a wider and more sustainable use of the land, no matter if the land is public or private. Within this context, the application of well developed criteria to be used in sMCA approaches like those showcased in this thesis can make the difference in harmonising multiple sectors and in delivering future critical targets.

References

Adams, W. M. (2012). Private and networked: large conservation areas in Scotland. Ecos, 33(3/4), 124-133.

Adriaensen, F., Chardon, J. P., De Blust, G., Swinnen, E., Villalba, S., Gulinck, H., & Matthysen, E. (2003). The application of 'least-cost'modelling as a functional landscape model. Landscape and urban planning, 64(4), 233-247.

Agriculture and Rural Affairs for Scotland, England, and Wales (i.e.The Scottish Government: Agriculture, DEFRA and the Welsh Assembly Government Department for Environment, Planning and Countryside). http://edina.ac.uk/agcensus/

Akram-Lodhi, A. H. 2008. World Development Report 2008: Agriculture for Development. Development and Change 39:1145-1161.

Alexander, J. M., Chalmandrier, L., Lenoir, J., Burgess, T. I., Essl, F., Haider, S., & Pellissier, L. (2018). Lags in the response of mountain plant communities to climate change. Global change biology, 24(2), 563-579.

Allen, R. G., Pereira, L. S., Raes, D., & Smith, M. (1998). Crop evapotranspiration. Guidelines for computing crop water requirements. Retrieved from http://www.fao.org/docrep/x0490e/x0490e00.HTM

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop evapotranspiration - Guidelines for computing crop water requirements, FAO Irrigation and drainage paper 56. Rome, Italy.

Andrea Baggio Compagnucci, David Donnelly, Alessandro Gimona, Laura Poggio, Justin Irvine (2015). An interactive Ecosystem Services tool to support land use decision making Aberdeenshire Regional Land Use Pilot. International Spring University on Ecosystem Services Modeling. Bilbao, 6 April 2015.

Andy Wightman, 11. March 2018. Open Up the Registers · Categories: Freedom of Information, Land Registration, Who Owns Scotland. Land Matter the blog and website of Andy Wightman.

Arnold, J. G., & Fohrer, N. (2005). SWAT2000: current capabilities and research opportunities in applied watershed modelling. Hydrological Processes: An International Journal, 19(3), 563-572.

Baggio Compagnucci, A.; Donnelly, D.; Poggio, L.; Castellazzi, M.; Irvine, R.J.; Gimona, A.; (2017) Aberdeenshire Regional Land Use Pilot. An interactive Ecosystem Services tool to support land use decision making. Conference: BEES-BES, Cardiff.

Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013a. Spatial dynamics of ecosystemservice flows: a comprehensive approach to quantifying actual services. Ecosyst. Serv. 4, 117–125.http://dx.doi.org/10.1016/j.ecoser.2012.07.012.

Bailey, N., Lee, J. T., & Thompson, S. (2006). Maximising the natural capital benefits of habitat creation: spatially targeting native woodland using GIS. Landscape and Urban Planning, 75(3-4), 227-243.

Bennett, E. M., and P. Balvanera. 2007. The future of production systems in a globalized world. Frontiers in Ecology and the Environment 5:191-198.

Bennett, E. M., G. D. Peterson, and L. J. Gordon. 2009. Understanding relationships among multiple ecosystem services. Ecology letters 12:1394-1404.

Berry, P. M., Dawson, T. P., Harrison, P. A., Pearson, R., & Butt, N. (2003). The sensitivity and vulnerability of terrestrial habitats and species in Britain and Ireland to climate change. Journal for Nature Conservation, 11(1), 15-23.

Bibby, C. J. (1998). Selecting areas for conservation. *Conservation science and action*, 176-201.

Birch, J. C., Thapa, I., Balmford, A., Bradbury, R. B., Brown, C., Butchart, S. H., & Thomas, D. H. (2014). What benefits do community forests provide, and to whom? A rapid assessment of ecosystem services from a Himalayan forest, Nepal. Ecosystem Services, 8, 118-127.

Bob McIntosh, Native pinewoods in Scotland: perspectives on policy and management, Forestry: An International Journal of Forest Research, Volume 79, Issue 3, July 2006, Pages 303–307, https://doi.org/10.1093/forestry/cpl017

Bolker, B. M., Pacala, S. W., & Neuhauser, C. (2003). Spatial dynamics in model plant communities: what do we really know?. The American Naturalist, 162(2), 135-148.

Boorman, D. B., Hollis, J. M., & Lilly, A. (1995). Hydrology of soil types: a hydrologically-based classification of the soils of the United Kingdom. Retrieved from http://www.ceh.ac.uk/products/publications/documents/ih126hydrologyofsoiltypes.pdf

Boumans, R., Roman, J., Altman, I., Kaufman, L., 2015. The multiscale integratedmodel of ecosystem services (MIMES): simulating the interactions of coupledhuman and natural systems. Ecosyst. Serv. 12, 30–41.http://dx.doi.org/10.1016/j.ecoser.2015.01.004

Branquart, E.; Liégeois, S. Normes de Gestion pour Favoriser la Biodiversité Dans les Bois Soumis au Régime Forestier (Complément à la Circulaire n◦2619); Ministère de la Région Wallonne—Direction Générale des Ressources Naturelles et de l'Environnement: Jambes, Belgium, 2005; p. 84

Brice R. Rea (1998) The Cairngorms — a landscape of selective linear erosion, Scottish Geographical Magazine, 114:2, 124-129, DOI: 10.1080/00369229818737041

Brooks, T. M., Mittermeier, R. A., da Fonseca, G. A., Gerlach, J., Hoffmann, M., Lamoreux, J. F., & Rodrigues, A. S. (2006). Global biodiversity conservation priorities. Science, 313(5783), 58-61.

Brown, G., & Weber, D. (2012). Measuring change in place values using public participation GIS (PPGIS). Applied Geography, 34, 316-324.

Brown, I. (2020). Challenges in delivering climate change policy through land use targets for afforestation and peatland restoration. *Environmental Science & Policy*, *107*, 36-45.

Bruinderink, G. G., Van Der Sluis, T., Lammertsma, D., Opdam, P., & Pouwels, R. (2003). Designing a coherent ecological network for large mammals in northwestern Europe. Conservation Biology, 17(2), 549-557.

Burton, R. J. (2004). Reconceptualising the 'behavioural approach'in agricultural studies: a socio-psychological perspective. Journal of Rural studies, 20(3), 359-371.

Cairngorms Local Development Strategy 2014-2020. https://cairngorms.co.uk/wp-content/uploads/2015/07/150409CairngormsLDS_Submission_V1.0.pdf

Cairngorms National Park Authority. Benchmark Review of the Cairngorms National Park for the Cairngorms Economic Forum 2013. FINAL REPORT.

Cairngorms National Park Authority. State of the Park Report 2006a

Calder, I. R. & Aylward, B., 2006. Forests and Floods: Moving to an evidence-based approach to watershed and integrated flood management. Water International, 31(1): 87-99.

Cerretelli, S., Poggio, L., Gimona, A., Yakob, G., Boke, S., Habte, M., & Black, H. (2018). Spatial assessment of land degradation through key ecosystem services: the role of globally available data. Science of the Total Environment, 628, 539-555.

CNPA 2008. The Forests of the Cairngorms, Cairngorms National Park, Forest and Woodland Framework. ISBN 978-0-9548782-7-6

CNPA 2015. Active Cairngorms, Cairngorms National Park, ISBN 978-1-906071-85-1

CNPA 2018, Cairngorms Forestry Strategy. CNPA 2018. All rights reserved. ISBN 978-1-906071-92-9

CNPA, 2019 Cairngorms Nature Action Plan 2019-2024. ISBN 978-1-906071-93-6 https://cairngorms.co.uk/wp-content/uploads/2019/02/CairngormsNatureAction19_24PlanFinal.pdf

CNPA 2019b. Economic Action Plan 2019-2022, Cairngorms National Park

Combe, M. M. (2016). The Land Reform (Scotland) Act 2016: another answer to the Scottish land question. Juridical Review, 2016, 291-313.

Cord, A. F., B. Bartkowski, M. Beckmann, A. Dittrich, K. Hermans-Neumann, A. Kaim, N. Lienhoop, K. Locher-Krause, J. Priess, C. Schroter-Schlaack, N. Schwarz, R. Seppelt, M. Strauch, T. Vaclavik, and M. Volk. 2017. Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead. Ecosystem Services 28:264-272.

Costanza, R., & Ruth, M. (1998). Using dynamic modeling to scope environmental problems and build consensus. Environmental management, 22(2), 183-195.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. 1998. The value of the world's ecosystem services and natural capital (Reprinted from Nature, vol 387, pg 253, 1997). Ecological Economics 25:3-15.

Dadson Simon J., Hall Jim W., Murgatroyd Anna, Acreman Mike, Bates Paul, Beven Keith, Heathwaite Louise, Holden Joseph, Holman Ian P., Lane Stuart N., O'Connell Enda, Penning-Rowsell Edmund, Reynard Nick, Sear David, Thorne Colin and Wilby Rob 2017. A restatement of the natural science evidence concerning catchment-based 'natural' flood management in the UK Proc. R. Soc. A.47320160706

Dai, F. C., Lee, C. F., & Zhang, X. H. (2001). GIS-based geo-environmental evaluation for urban land-use planning: a case study. Engineering geology, 61(4), 257-271.

Dale MRT, Fortin MJ (2010) From graphs to spatial graphs. In: Futuyma DJ, Shafer HB, Simberloff D, editors. Annual Review of Ecology, Evolution, and Systematics. Palo Alto: Annual Reviews. 21–38.

D'Eon, R. G., Glenn, S. M., Parfitt, I., & Fortin, M. J. (2002). Landscape connectivity as a function of scale and organism vagility in a real forested landscape. Conservation Ecology, 6(2).

De Groot, R. S., R. Alkemade, L. Braat, L. Hein, and L. Willemen. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological complexity 7:260-272.

Delpla, I., Jung, A. V., Baures, E., Clement, M., & Thomas, O. (2009). Impacts of climate change on surface water quality in relation to drinking water production. Environment international, 35(8), 1225-1233.

Directive, W.F., 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Official journal of the European communities, 22(12), p.2000.

Díaz-Balteiro, L., & Romero, C. (2003). Forest management optimisation models when carbon captured is considered: a goal programming approach. *Forest ecology and management*, *174*(1-3), 447-457.

Doerr, V. A., Doerr, E. D., & Davies, M. J. (2011). Dispersal behaviour of Brown Treecreepers predicts functional connectivity for several other woodland birds. Emu-Austral Ornithology, 111(1), 71-83.

Efstratiadis A, Koussis AD, Koutsoyiannis D, Mamassis N (2013) Flood design recipes vs. reality: can predictions for ungauged basins be trusted? Nat Hazards Earth Syst Sci Discuss1:7387–7416.

Eldridge, D.J. and Freudenberger, D., 2005. Ecosystem wicks: woodland trees enhance water infiltration in a fragmented agricultural landscape in eastern Australia. Austral Ecology, 30, 336–347. doi:10.1111/aec.2005.30.issue-3

Elith, J., & Leathwick, J. R. (2009). Species distribution models: ecological explanation and prediction across space and time. Annual review of ecology, evolution, and systematics, 40, 677-697.

EMEP. (2012). Transboundary Acidification, Eutrophication and Ground Level Ozone in Europe in 2010. Retrieved from http://emep.int/publ/reports/2012/status_report_1_2012.pdf

Estreguil, C., Caudullo, G., De Rigo, D., & San Miguel, J. (2013). Forest landscape in Europe: pattern, fragmentation and connectivity. EUR Scientific and Technical Research, 25717.

Etheridge, B., Summers, R.W., Green, R.E. (1997) The effects of illegal killing and destruction of nests by humans on the population dynamics of the hen harrier Circus cyaneus in Scotland. Journal of Applied Ecology, 34, 1081–1105.

Etienne, R. S., Vos, C. C., & Jansen, M. J. (2003). Ecological impact assessment in data-poor systems: a case study on metapopulation persistence. Environmental Management, 32(6), 760-777.

Falkenmark, M. and Folke, C., 2002. The ethics of socio-ecohydrological catchment management: towards hydrosolidarity.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global consequences of land use. Science 309:570-574.

Forestry Commission Scotland. The Scottish Forestry Strategy (2006) (Edinburgh. Available at: http://www.forestry.gov.uk/pdf/sfs2006fcfc101.pdf/$file/sfs2006fcfc101.pdf (accessed: 10/10/2014))

Fürst, C., Opdam, P., Inostroza, L. et al. Evaluating the role of ecosystem services in participatory land use planning: proposing a balanced score card. Landscape Ecol 29, 1435–1446 (2014). https://doi.org/10.1007/s10980-014-0052-9

Gagkas, Zisis & Lilly, Allan. (2019). Downscaling soil hydrological mapping used to predict catchment hydrological response with random forests. Geoderma. 341. 216-235. 10.1016/j.geoderma.2019.01.048.

Galpern, P., Manseau, M., & Fall, A. (2011). Patch-based graphs of landscape connectivity: a guide to construction, analysis and application for conservation. Biological conservation, 144(1), 44-55.

García-Nieto, A. P., García-Llorente, M., Iniesta-Arandia, I., & Martín-López, B. (2013). Mapping forest ecosystem services: from providing units to beneficiaries. Ecosystem Services, 4, 126-138.

Geneletti, D., & van Duren, I. (2008). Protected area zoning for conservation and use: A combination of spatial multicriteria and multiobjective evaluation. Landscape and urban planning, 85(2), 97-110.

German, L., Taye, H., Charamila, S., Tolera, T. and Tanui, J., 2006. The many meanings of collective action: lessons on enhancing gender inclusion and equity in watershed management (No. 577-2016-39158).

Giandotti M (1934) Previsione delle piene e delle magre dei corsid'acqua Memorie e studi idrografici, Servizio Idrografico Italiano, Report No 2.

Gimona A., Irvine J., Donnelly D., Baggio Compagnucci A., 2016. Aberdeenshire Regional Land Use Pilot: Mapping the consEquences of LanD use Change (MELODIC). The James Hutton Institute. SEFARI. https://knowledgescotland.webarchive.sefari.scot/briefings/briefings6d26.html?id=397

Gimona, A., & Birnie, R. V. (2002). Spatio-temporal modelling of broad scale heterogeneity in soil moisture content: a basis for an ecologically meaningful classification of soil landscapes. Landscape ecology, 17(1), 27-41.

Gimona, A., and D. van der Horst. 2007. Mapping hotspots of multiple landscape functions: a case study on farmland afforestation in Scotland. Landscape Ecology 22:1255-1264.

Gimona, A., Poggio, L., Brown, I., & Castellazzi, M. (2012). Woodland networks in a changing climate: threats from land use change. Biological conservation, 149(1), 93-102.

Gimona, A., Poggio, L., Brown, I., & Castellazzi, M. (2012). Woodland networks in a changing climate: threats from land use change. Biological conservation, 149(1), 93-102.

Gimona, A.; Baggio Compagnucci, A. (2019) Predicting the quality status of biological elements (Water Framework Directive) for surface water in Scotland using the output of InVEST models., Report for RESAS Objective 1.4.2cii, The James Hutton Institute, Aberdeen, 13pp.

Gimona, A.; Baggio Compagnucci, A.; Donnelly, D.; Irvine, R.J., (2016) Aberdeenshire Regional Land Use Pilot: A web-based spatial tool to support land use decision making., In: Brooker, R., Hester, A. & Pakeman, R. (eds.). Ecosystem Services. The James Hutton Institute, Craigiebuckler, Aberdeen, pp16-17.

Gimona, A.; Baggio Compagnucci, A.; Hewitt, R.; Castellazzi, M.; Poggio, L. (2018) Simulating the effect of linear features on nutrient and sediment export at the catchment scale., Report.

Goicoechea, A., Hansen, D. R., & Duckstein, L. (1982). Multiobjective decision analysis with engineering and business applications. John Wiley & Sons Incorporated.

Green, R.E. & Etheridge, B. (1999) Breeding success of the hen harrier Circus cyaneus in relation to the distribution of grouse moors and the red fox Vulpes vulpes. Journal of Applied Ecology, 36, 472–483.

Grimaldi, S., Petroselli, A., Tauro, F. and Porfiri, M., 2012. Time of concentration: a paradox in modern hydrology. Hydrological Sciences Journal, 57 (2), 217–228

Guermond Y (2008) The modeling process in geography. Wiley, New York.

Haberl, H., K. H. Erb, F. Krausmann, V. Gaube, A. Bondeau, C. Plutzar, S. Gingrich, W. Lucht, and M. Fischer-Kowalski. 2007. Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. Proceedings of the National Academy of Sciences of the United States of America 104:12942-12945.

Haines-Young, R., and M. Potschin. 2011. Common international classification of ecosystem services (CICES): 2011 Update. Nottingham: Report to the European Environmental Agency.

Hall, A. M., Gillespie, M. R., Thomas, C. W., & Ebert, K. (2013). Scottish landform examples: the Cairngorms–a pre-glacial upland granite landscape. Scottish Geographical Journal, 129(1), 2-14.

Harper, A. B., Powell, T., Cox, P. M., House, J., Huntingford, C., Lenton, T. M., & Shu, S. (2018). Land-use emissions play a critical role in land-based mitigation for Paris climate targets. *Nature communications*, *9*(1), 1-13.

Hassan, R., R. Scholes, and N. Ash. 2005. Ecosystems and human well-being: Current state and trends. Millennium Ecosystem Assessment. Islandpress, Washington DC.

Hebe Carus and George Campbell, Futurescapes. RSPB. https://www.rspb.org.uk/globalassets/downloads/documents/futurescapes/futurescapes-caledonianforest-booklet.pdf

Herrick, J. E., and J. Sarukhan. 2007. A strategy for ecology in an era of globalization. Frontiers in Ecology and the Environment 5:172-181.

Hess, G. R., & Fischer, R. A. (2001). Communicating clearly about conservation corridors. Landscape and urban planning, 55(3), 195-208.

Hobbs, R. (2009). Woodland restoration in Scotland: ecology, history, culture, economics, politics and change. Journal of Environmental Management, 90(9), 2857-2865.

Holling, C. S. 1996. Engineering resilience versus ecological resilience. Engineering within ecological constraints 31:32.

Hou, J.; Guo, K.; Liu, F.; Han, H.; Liang, Q.; Tong, Y.; Li, P. Assessing Slope Forest Effect on Flood Process Caused by a Short-Duration Storm in a Small Catchment. Water 2018, 10, 1256.

Houdet, J., Trommetter, M., & Weber, J. (2012). Understanding changes in business strategies regarding biodiversity and ecosystem services. Ecological Economics, 73, 37-46.

Houghton, R. A., J. R. Hackler, and R. L. Cushman. 2001. Carbon Flux to the Atmosphere From Land-use Changes: 1850 to 1990 (NDP-050/R1). Carbon Dioxide Information Analysis Center (CDIAC), Oak Ridge National Laboratory (ORNL), Oak Ridge, TN (USA).

https://cairngorms.co.uk/wp-content/uploads/2020/03/Cairngorms-Climate-Change-Conference-2020-Hester-Baggio-Gimona-FINAL.pdf

Huang, I. B., Keisler, J., & Linkov, I. (2011). Multi-criteria decision analysis in environmental sciences: Ten years of applications and trends. *Science of the total environment*, *409*(19), 3578-3594.

Hwang, C. L., & Lin, M. J. (2012). Group decision making under multiple criteria: methods and applications (Vol. 281). Springer Science & Business Media.

Intergovernmental Panel on Climate Change. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, pp 151.

Innes, J. E., & Booher, D. E. (1999). Consensus building and complex adaptive systems: A framework for evaluating collaborative planning. Journal of the American planning association, 65(4), 412-423.

Jackson, D.L. 2000. Guidance on the interpretation of the Biodiversity Broad Habitat Classification (terrestrial and freshwater types): Definitions and the relationship with other classifications, JNCC Report No. 307, JNCC, Peterborough, ISSN 0963-8091.

Jankowski, P. (2001). GIS for group decision making. CRC Press.

Jones, A. T. (1999). The Caledonian pinewood inventory of Scotland's native Scots pine woodlands. Scottish Forestry, 53(4), 237-242.

Jönsson, P., & Eklundh, L. (2004). TIMESAT—a program for analyzing time-series of satellite sensor data. Computers & geosciences, 30(8), 833-845.

Kangas, J., & Kangas, A. (2005). Multiple criteria decision support in forest management—the approach, methods applied, and experiences gained. Forest ecology and management, 207(1-2), 133-143.

Kangas, J., Kangas, A., Leskinen, P., & Pykäläinen, J. (2001). MCDM methods in strategic planning of forestry on state-owned lands in Finland: applications and experiences. *Journal of Multi-Criteria Decision Analysis*, *10*(5), 257-271.

Kempe Nick. The power of landowners in the Cairngorms National Park. October 14, 2016. www.Parkwatchscotland.co.uk.

Keeney, R. L., Raiffa, H., & Meyer, R. F. (1993). Decisions with multiple objectives: preferences and value trade-offs. Cambridge university press.

Kerr, J., 2002. Watershed development, environmental services, and poverty alleviation in India. World development, 30(8), pp.1387-1400.

Kindlmann P, Burel F (2008) Connectivity measures: a review. Landscape Ecology 23: 879–890.

King, R. S., Richardson, C. J., Urban, D. L., & Romanowicz, E. A. (2004). Spatial dependency of vegetation–environment linkages in an anthropogenically influenced wetland ecosystem. Ecosystems, 7(1), 75-97.

Kirpich ZP (1940) Time of concentration of small agricultural watersheds. J Civ Eng 10(6):362

Kline, J. D., and M. J. Mazzotta. 2012. Evaluating tradeoffs among ecosystem services in the management of public lands. Gen. Tech. Rep. PNW-GTR-865. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. 48 p 865.

Knickel, K., and H. Renting. 2000. Methodological and conceptual issues in the study of multifunctionality and rural development. Sociologia Ruralis 40:512-+.

Koch, E. W., E. B. Barbier, B. R. Silliman, D. J. Reed, G. M. E. Perillo, S. D. Hacker, E. F. Granek, J. H. Primavera, N. Muthiga, S. Polasky, B. S. Halpern, C. J. Kennedy, C. V. Kappel, and E. Wolanski. 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Frontiers in Ecology and the Environment 7:29-37.

Kochli, D. A., and P. Brang. 2005. Simulating effects of forest management on selected public forest goods and services: A case study. Forest Ecology and Management 209:57-68.

Krcmar, E., van Kooten, G. C., & Vertinsky, I. (2005). Managing forest and marginal agricultural land for multiple tradeoffs: compromising on economic, carbon and structural diversity objectives. *Ecological Modelling*, *185*(2-4), 451-468.

Kumar, P. 2012. The economics of ecosystems and biodiversity: ecological and economic foundations. Routledge.

Lafortezza, R., G. Sanesi, and J. Q. Chen. 2013. Large-scale effects of forest management in Mediterranean landscapes of Europe. Iforest-Biogeosciences and Forestry 6:342-346.

Lee, H., and S. Lautenbach. 2016. A quantitative review of relationships between ecosystem services. Ecological Indicators 66:340-351.

Lehmann, A., Jaquet, J. M., & Lachavanne, J. B. (1997). A GIS approach of aquatic plant spatial heterogeneity in relation to sediment and depth gradients, Lake Geneva, Switzerland. Aquatic botany, 58(3-4), 347-361.

Levins, Richard. "The strategy of model building in population biology." American scientist 54.4 (1966): 421-431.

Li MH, Chibber P (2008) Overland flow time of concentration onvery flat terrains. Trans Res Rec 2060:133–140

Lilly, A. (2010, August). A hydrological classification of UK soils based on soil morphological data. In 19th World Congress of Soil Science, Soil Solutions for a Changing World (Vol. 16). IUSS Brisbane, Australia.

Lindborg, R., Gordon, L. J., Malinga, R., Bengtsson, J., Peterson, G., Bommarco, R., & Smith, H. G. (2017). How spatial scale shapes the generation and management of multiple ecosystem services. Ecosphere, 8(4), e01741.

Linstead, C. & Gurnell, A.M., 1999. Large woody debris in British headwater rivers: physical habitat role and management guide. Environment Agency R&D Technical Report W185. Environment Agency, Bristol.

Local Development Plan, 2020. Main Issue Report, Flood Risk Assessment. Cairngorms National Park Authority, Avimore, 2020.

Logsdon, R. A., Kalcic, M. M., Trybula, E. M., Chaubey, I., & Frankenberger, J. R. (2015). Ecosystem services and Indiana agriculture: farmers’ and conservationists’ perceptions. International Journal of Biodiversity Science, Ecosystem Services & Management, 11(3), 264-282.

Longley, P. A., Goodchild, M. F., Maguire, D. J., & Rhind, D. W. (2005). Geographic information systems and science. John Wiley & Sons.

Lust, N., Geudens, G., & Nachtergale, L. (2001). Aspects of biodiversity of Scots pine forests in Europe. Silva Gandavensis, 66.

Mace, G. M., K. Norris, and A. H. Fitter. 2012. Biodiversity and ecosystem services: a multilayered relationship. Trends in Ecology & Evolution 27:19-26.

Maes, J., M. L. Paracchini, G. Zulian, M. B. Dunbar, and R. Alkemade. 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biological Conservation 155:1-12.

Mander, Ü., K. Helming, and H. Wiggering. 2007. Multifunctional land use: meeting future demands for landscape goods and services. Pages 1-13 Multifunctional land use. Springer.

Marie Castellazzi, Alessandro Gimona, Andrea Baggio, Justin Irvine, Laura Poggio, Andrew Coleman (2016). Sustainable Land Management Options Tool, ESCOM – 20th April 2016

Mark A, Marek PE (2011) Hydraulic design manual. Tex Dep Trans, Design div, Texas

Marrotte, R. R., Bowman, J., Brown, M. G., Cordes, C., Morris, K. Y., Prentice, M. B., & Wilson, P. J. (2017). Multi-species genetic connectivity in a terrestrial habitat network. Movement ecology, 5(1), 1-11.

Martin-Lopez, B., I. Iniesta-Arandia, M. Garcia-Llorente, I. Palomo, I. Casado-Arzuaga, D. G. Del Amo, E. Gomez-Baggethun, E. Oteros-Rozas, I. Palacios-Agundez, B. Willaarts, J. A. Gonzalez, F. Santos-Martin, M. Onaindia, C. Lopez-Santiago, and C. Montes. 2012. Uncovering Ecosystem Service Bundles through Social Preferences. Plos One 7.

Mason, W.L. & Alia, R. (2000). Current and future status of Scots pine (Pinus sylvestris L.) forests in Europe In: W.L. Mason, R Alia, G.I. Forrest, AF.M. Olsthoorn and K. Sjoberg (Eds). Silviculture and Biodiversity of Scots pine Forests in Europe. Investigaci6n Agraria, Fuera de Serie, Nr.1, 317-336.

McCuen, R. H. (2009). Uncertainty analyses of watershed time parameters. Journal of Hydrologic Engineering, 14(5), 490-498.

McRae, B. H., & Shah, V. B. (2009). Circuitscape user's guide. The University of California, Santa Barbara.

McRae BH, Beier P (2007) Circuit theory predicts gene flow in plant and animal populations. Proceedings of the National Academy of Sciences of the United States of America 104: 19885–19890.

McRae, B. H., Dickson, B. G., Keitt, T. H., & Shah, V. B. (2008). Using circuit theory to model connectivity in ecology, evolution, and conservation. Ecology, 89(10), 2712-2724.

McRae, B. H., Dickson, B. G., Keitt, T. H., & Shah, V. B. (2008). Using circuit theory to model connectivity in ecology, evolution, and conservation. Ecology, 89(10), 2712-2724.

MEA. 2005. Ecosystems and human well-being: synthesis. Island Press Washington, DC.

Met Office, 2013. Met Office: UKCP09 Available data sets [WWW Document]. URL http://www.metoffice.gov.uk/climatechange/science/monitoring/ukcp09/available/

Minang, P. A., van Noordwijk, M., Freeman, O. E., Mbow, C., de Leeuw, J., & Catacutan, D. (Eds.). (2014). Climate-smart landscapes: multifunctionality in practice. ASB Partnership for The Tropical Forest margins.

Minckler, L. S. (1941). The right tree in the right place. Journal of Forestry, 39(8), 685-688.

Minkkinen, K., Byrne, K. A., & Trettin, C. (2008). Climate impacts of peatland forestry. *Peatlands and climate change*, 98-122.

Mitášová, H., & Hofierka, J., 1993. Interpolation by regularized spline with tension: II. Application to terrain modeling and surface geometry analysis. Mathematical geology, 25(6), 657-669.

Mitchell SJ. Coutts MP, Grace J. A synopsis of windthrow in British Columbia: occurrence, implications, assessment and management, Wind and Trees, 1995Cambridge, UK, Cambridge University Press(pg. 448-459)

Mitchell, R. J., Bellamy, P. E., Ellis, C. J., Hewison, R. L., Hodgetts, N. G., Iason, G. R., & Taylor, A. F. S. (2019). OakEcol: A database of Oak-associated biodiversity within the UK. Data in brief, 25, 104120.

MLURI, 1988 The land cover of Scotland final report, Macaulay Land Use Research Institute, Aberdeen (1988)

Moilanen A (2011) On the limitations of graph-theoretic connectivity in spatial ecology and conservation. Journal of Applied Ecology 48: 1543–1547.

Montgomery, D. R. (2007). Soil erosion and agricultural sustainability. Proceedings of the National Academy of Sciences, 104(33), 13268-13272.

Morton, D., Rowland, C., Wood, C. Meek, L., Marston, C., Smith, G., Wadsworth, R., Simpson, I.C. 2011. Final Report for LCM2007 - the new UK land cover map. Countryside Survey Technical Report No 11/07 NERC/Centre for Ecology & Hydrology 112pp. (CEH Project Number: C03259).

Mostert, E. (2003). The European water framework directive and water management research. Physics and Chemistry of the Earth, Parts A/B/C, 28(12-13), 523-527.

Mouchet, M. A., P. Lamarque, B. Martin-Lopez, E. Crouzat, P. Gos, C. Byczek, and S. Lavorel. 2014. An interdisciplinary methodological guide for quantifying associations between ecosystem services. Global Environmental Change-Human and Policy Dimensions 28:298-308.

Mulligan, M., Guerry, A., Arkema, K., Bagstad, K., & Villa, F. (2010). Capturing and quantifying the flow of ecosystem services. Framing the flow: innovative approaches to understand, protect and value ecosystem services across linked habitats. UNEP World Conservation Monitoring Centre, Cambridge, UK, 26-33.

Muñoz-Rojas, J., Nijnik, M., González-Puente, M., & Cortines-García, F. (2015). Synergies and conflicts in the use of policy and planning instruments for implementing forest and woodland corridors and networks; a case study in NE Scotland. Forest Policy and Economics, 57, 47-64.

Nabuurs, G. J., Pussinen, A., Van Brusselen, J., & Schelhaas, M. J. (2007). Future harvesting pressure on European forests. European Journal of Forest Research, 126(3), 391-400.

National forestry inventory 2015

Nedkov, S., & Burkhard, B. (2012). Flood regulating ecosystem services—Mapping supply and demand, in the Etropole municipality, Bulgaria. Ecological Indicators, 21, 67-79.

Newey S., Fielding D., Wilson M., 2016. Mapping farmland wader distribution and population change to identify priority areas for conservation and management action. The James Hutton Institute. Technical Report available on: https://www.hutton.ac.uk/sites/default/files/files/Newey_etal_IdentifyingWaderPriorityAreas_InternallyReviewed%26%20Signedoff_FINAL.pdf

Niedziałkowski, K., Blicharska, M., Mikusiński, G., & Jędrzejewska, B. (2014). Why is it difficult to enlarge a protected area? Ecosystem services perspective on the conflict around the extension of the Białowieża National Park in Poland. Land use policy, 38, 314-329.

Nieminen, M., Koskinen, M., Sarkkola, S., Laurén, A., Kaila, A., Kiikkilä, O., & Ukonmaanaho, L. (2015). Dissolved organic carbon export from harvested peatland forests with differing site characteristics. *Water, Air, & Soil Pollution*, *226*(6), 1-12.

Nieminen, M., Sarkkola, S., & Laurén, A. (2017). Impacts of forest harvesting on nutrient, sediment and dissolved organic carbon exports from drained peatlands: A literature review, synthesis and suggestions for the future. *Forest ecology and management*, *392*, 13-20.

Nisbet, T. R.; Thomas, H., 2006. The role of woodland in flood control: a landscape perspective.

Nisbet, T., 2005. Water use by trees. Forestry Commission Information Note 65. Forestry Commission, Edinburgh, pp. 1-8.

Nisbet, T., Silgram, M., Shah, N., Morrow, K., & Broadmeadow, S. (2011). Woodland for water: woodland measures for meeting water framework directive objectives. Forest Research Monograph, 4, 156.

Nisbet, T.R. and Thomas, H., 2006. The role of woodland in flood control – a landscape perspective. In 'Water and the landscape: the landscape ecology of freshwater ecosystems'. Proceedings of the 14th Annual IALE(UK) Conference, Eds. B. Davies and S. Thompson, p118-125. IALE(UK), Oxford.

Oberdorff, T., Pont, D., Hugueny, B., & Chessel, D. (2001). A probabilistic model characterizing fish assemblages of French rivers: a framework for environmental assessment. Freshwater Biology, 46(3), 399-415.

Olden, J. D., Jackson, D. A., & Peres-Neto, P. R. (2002). Predictive models of fish species distributions: a note on proper validation and chance predictions. Transactions of the American fisheries society, 131(2), 329-336.

OS Terrain 5 GML 3.2, Coverage: Scotland, Updated Jan 2018, Ordnance Survey, GB. © Crown copyright and database rights 2018 OS Licence number 100019294. Downloaded: June 2018

Palacios-Agundez, I., De Manuel, B. F., Rodríguez-Loinaz, G., Pena, L., Ametzaga-Arregi, I., Alday, J. G., & Onaindia, M. (2014). Integrating stakeholders' demands and scientific knowledge on ecosystem services in landscape planning. Landscape ecology, 29(8), 1423-1433.

Panagos P., Borrelli P., Meusburger K., Yu B., Klik A., Lim K.J., Yang J.E, Ni J., Miao C., Chattopadhyay N., Sadeghi S.H., Hazbavi Z., Zabihi M., Larionov G.A., Krasnov S.F., Garobets A., Levi Y., Erpul G., Birkel C., Hoyos N., Naipal V., Oliveira P.T.S., Bonilla C.A., Meddi M., Nel W., Dashti H., Boni M., Diodato N., Van Oost K., Nearing M.A., Ballabio C., 2017. Global rainfall erosivity assessment based on high-temporal resolution rainfall records. Scientific Reports 7: 4175. DOI: 10.1038/s41598-017-04282-8.

Panagos P., Van Liedekerke M., Jones A., Montanarella L., "European Soil Data Centre: Response to European policy support and public data requirements"; (2012a) Land Use Policy, 29 (2), pp. 329-338. doi:10.1016/j.landusepol.2011.07.003

Panagos P., Van Liedekerke M., Jones A., Montanarella L., "European Soil Data Centre: Response to European policy support and public data requirements"; (2012b) Land Use Policy, 29 (2), pp. 329-338. doi:10.1016/j.landusepol.2011.07.003

Panagos P., Van Liedekerke M., Jones A., Montanarella L., "European Soil Data Centre: Response to European policy support and public data requirements"; (2012c) Land Use Policy, 29 (2), pp. 329-338. doi:10.1016/j.landusepol.2011.07.003

Pandey, D. N. (2002). Global climate change and carbon management in multifunctional forests. Current science, 593-602.

Patterson, G., Nelson D., Robertson, P., Tullis, J., 2014. Published by Forestry Commission Scotland January 2014 ISBN 978 0 85538 899 7 © Crown Copyright 2014

Paul, K. I., Polglase, P. J., & Richards, G. P. (2003). Sensitivity analysis of predicted change in soil carbon following afforestation. *Ecological Modelling*, *164*(2-3), 137-152.

Phua, M. H., & Minowa, M. (2005). A GIS-based multi-criteria decision making approach to forest conservation planning at a landscape scale: a case study in the Kinabalu Area, Sabah, Malaysia. *Landscape and urban planning*, *71*(2-4), 207-222.

Pliscoff, P., Simonetti, J. A., Grez, A. A., Vergara, P. M., & Barahona-Segovia, R. M. (2020). Defining corridors for movement of multiple species in a forest-plantation landscape. Global Ecology and Conservation, 23, e01108.

Poff, N. L. (2002). Ecological response to and management of increased flooding caused by climate change. Philosophical Transactions of the Royal Society of London. Series A: Mathematical, Physical and Engineering Sciences, 360(1796), 1497-1510.

Poggio, L., Gimona, A., & Brown, I. (2012). Spatio-temporal MODIS EVI gap filling under cloud cover: An example in Scotland. ISPRS journal of photogrammetry and remote sensing, 72, 56-72.

Polatidis, H., Haralambopoulos, D. A., Munda, G., & Vreeker, R. (2006). Selecting an appropriate multi-criteria decision analysis technique for renewable energy planning. Energy Sources, Part B, 1(2), 181-193.

Pollard, S. and Du Toit, D., 2008. Integrated water resource management in complex systems: How the catchment management strategies seek to achieve sustainability and equity in water resources in South Africa. Water SA, 34(6), pp.671-679.

Pomeroy, R., & Douvere, F. (2008). The engagement of stakeholders in the marine spatial planning process. Marine policy, 32(5), 816-822.

Postel, S. L., & Thompson Jr, B. H. (2005, May). Watershed protection: Capturing the benefits of nature's water supply services. In Natural Resources Forum (Vol. 29, No. 2, pp. 98-108). Oxford, UK: Blackwell Publishing, Ltd..

Potts, G.R. (1998) Global dispersion of nesting hen harriers Circus cyaneus: implications for grouse moors in the UK. Ibis, 140, 76–88.

R Development Core Team (2015). R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. http://www.R-project.org

Ram, C., Montibeller, G., & Morton, A. (2011). Extending the use of scenario planning and MCDA for the evaluation of strategic options. Journal of the Operational Research Society, 62(5), 817-829.

Raudsepp-Hearne, C., G. D. Peterson, and E. M. Bennett. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proceedings of the National Academy of Sciences of the United States of America 107:5242-5247.

Redhead, J. W., Stratford, C., Sharps, K., Jones, L., Ziv, G., Clarke, D., ... & Bullock, J. M. (2016). Empirical validation of the InVEST water yield ecosystem service model at a national scale. Science of the Total Environment, 569, 1418-1426.

Redhead, J. W., Stratford, C., Sharps, K., Jones, L., Ziv, G., Clarke, D., ... & Bullock, J. M. (2016). Empirical validation of the InVEST water yield ecosystem service model at a national scale. Science of the Total Environment, 569, 1418-1426.

Reich, R. M., Joy, S. M., & Reynolds, R. T. (2004). Predicting the location of northern goshawk nests: modeling the spatial dependency between nest locations and forest structure. Ecological Modelling, 176(1-2), 109-133.

Reid, C. (2018). Scotland's Forestry Strategy 2019-2029. Available online: https://forestry.gov.scot/forestry-strategy

Rodriguez, J. P., T. D. Beard, E. M. Bennett, G. S. Cumming, S. J. Cork, J. Agard, A. P. Dobson, and G. D. Peterson. 2006. Trade-offs across space, time, and ecosystem services. Ecology and Society 11.

Rural Affairs, Food and Environment Research Strategy for 2016-2021 (2015). Scottish Government, ISBN:978178440991. https://www.gov.scot/publications/rural-affairs-food-environment-research-strategy-2016-2021/

Saaty, T. L. (1980). The Analytic Hierarchy Process: Planning, Priority Setting, Resource Allocation: McGraw-Hill. New York.

Salt, D., Castellazzi, M., Poggio, L., Baggio Compagnucci, A., & Gimona, A. (2018, April). Cloud computing for ecosystem services a

Sayer, J., Sunderland, T., Ghazoul, J., Pfund, J. L., Sheil, D., Meijaard, E., ... & Buck, L. E. (2013). Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proceedings of the national academy of sciences, 110(21), 8349-8356.

Schmitz, S., Adams, R., & Walsh, C. (2012). The use of continuous data versus binary data in MTC models: A case study in rheumatoid arthritis. BMC medical research methodology, 12(1), 1-17.

Schröter, M., & van Oudenhoven, A. P. (2016). Ecosystem services go beyond money and markets: reply to Silvertown. *Trends in Ecology & Evolution*, *31*(5), 333-334.

Scoble J, Lowe AJ (2010) A case for incorporating phylogeography and landscape genetics into species distribution modelling approaches to improve climate adaptation and conservation planning. Diversity and Distributions 16:343–353.

Scott Newey1*, Debbie Fielding1 , and Mark Wilson, 20Mapping farmland wader distributions and population change to identify wader priority areas for conservation and management action

Scottish Environment Protection Agency (SEPA). 2015. Natural Flood Management Handbook.

Scottish Government (2021). Scotland's Carbon Footprint: 1998-2017. s. Available online: https://www.gov.scot/news/scotlands-carbon-footprint-1998-2017/ [last accessed 12.04.2021].

Scottish Government (2009). Climate Change (Scotland) Act 2009. Edinburgh, UK. Available online: https://www.legislation.gov.uk/asp/2009/12/contents

Scottish Government, 11 March 2020. Planning and Environmental Appeals Division. Appeal Decision Notice. Enforcement notice appeal reference: ENA-001-2003.

Scottish Government. (2013). Low Carbon Scotland: Meeting the Emissions Reduction Targets 2013–2027–The Second Report on Proposals and Policies.

Scottish Parliament, 2012. ECONOMY, ENERGY AND TOURISM COMMITTEE 5th Meeting 2012, Session 4Official report. Published in Edinburgh by APS Group Scotland. e-format available ISBN 978-1-4061-8296-5

Seppelt, R., S. Lautenbach, and M. Volk. 2013. Identifying trade-offs between ecosystem services, land use, and biodiversity: a plea for combining scenario analysis and optimization on different spatial scales. Current Opinion in Environmental Sustainability 5:458-463.

Shuttleworth, C. M., Lurz, P. W., Geddes, N., & Browne, J. (2012). Integrating red squirrel (Sciurus vulgaris) habitat requirements with the management of pathogenic tree disease in commercial forests in the UK. Forest Ecology and Management, 279, 167-175.

Smith, P. (2012). Soils and climate change. *Current opinion in environmental sustainability*, *4*(5), 539-544.

Spear SF, Balkenhol N, Fortin MJ, McRae BH, Scribner K (2010) Use of resistance surfaces for landscape genetic studies: considerations for parameter#ization and analysis. Molecular Ecology 19: 3576–3591.

Steiniger, S., & Hay, G. J. (2009). Free and open source geographic information tools for landscape ecology. Ecological informatics, 4(4), 183-195.

Stevenson-Holt, C. D., Watts, K., Bellamy, C. C., Nevin, O. T., & Ramsey, A. D. (2014). Defining landscape resistance values in least-cost connectivity models for the invasive grey squirrel: a comparison of approaches using expert-opinion and habitat suitability modelling. PloS one, 9(11), e112119.

Stosch, K.C., Quilliam, R.S., Bunnefeld, N. and Oliver, D.M., 2017. Managing multiple catchment demands for sustainable water use and ecosystem service provision. Water, 9(9), p.677.

"Sustainable Tourism". Cairngorms National Park Authority. 15 November 2019. Retrieved 8 December 2019

Swiler, L. P., Hough, P. D., Qian, P., Xu, X., Storlie, C., & Lee, H. (2014). Surrogate models for mixed discrete-continuous variables. In Constraint Programming and Decision Making (pp. 181-202). Springer, Cham.

Talbot, C. J., Bennett, E. M., Cassell, K., Hanes, D. M., Minor, E. C., Paerl, H., & Xenopoulos, M. A. (2018). The impact of flooding on aquatic ecosystem services. Biogeochemistry, 141(3), 439-461.

Temel, J., Jones, A., Jones, N., & Balint, L. (2018). Limits of monetization in protecting ecosystem services. *Conservation Biology*, *32*(5), 1048-1062.

The Scottish Government (2011) Getting The Best From Our Land: a guide to Scotland’s first land use strategy, The Scottish Government, Edinburgh (www.scotland.gov.uk)

Thomas, H. J., Paterson, J. S., Metzger, M. J., & Sing, L. (2015). Towards a research agenda for woodland expansion in Scotland. Forest Ecology and Management, 349, 149-161.

Tischendorf, L., & Fahrig, L. (2000). On the usage and measurement of landscape connectivity. Oikos, 90(1), 7-19.

Uhde, B., Hahn, W. A., Griess, V. C., & Knoke, T. (2015). Hybrid MCDA methods to integrate multiple ecosystem services in forest management planning: a critical review. Environmental management, 56(2), 373-388.

Ujházy, N., Molnár, Z., Bede-Fazekas, Á., & Biró, M. (2020). Do farmers and conservationists perceive landscape changes differently?. Ecology and Society, 25(3).

Ureta, J. Carl, et al. "Using stakeholders' preference for ecosystems and ecosystem services as an economic basis underlying strategic conservation planning." Heliyon 6.12 (2020): e05827. https://doi.org/10.1016/j.heliyon.2020.e05827

Urban, D. L., Minor, E. S., Treml, E. A., & Schick, R. S. (2009). Graph models of habitat mosaics. Ecology letters, 12(3), 260-273.

Vallet, A., B. Locatelli, H. Levrel, S. Wunder, R. Seppelt, R. J. Scholes, and J. Oszwald. 2018. Relationships Between Ecosystem Services: Comparing Methods for Assessing Tradeoffs and Synergies. Ecological Economics 150:96-106.

Van der Plas, F., Ratcliffe, S., Ruiz-Benito, P., Scherer-Lorenzen, M., Verheyen, K., Wirth, C., & Allan, E. (2018). Continental mapping of forest ecosystem functions reveals a high but unrealised potential for forest multifunctionality. Ecology letters, 21(1), 31-42.

Van Oudenhoven, A. P., Petz, K., Alkemade, R., Hein, L., & de Groot, R. S. (2012). Framework for systematic indicator selection to assess effects of land management on ecosystem services. *Ecological Indicators*, *21*, 110-122.

Vanguelova, E., Chapman, S., Perks, M., Yamulki, S., Randle, T., Ashwood, F., & Morison, J. (2018). Afforestation and restocking on peaty soils–new evidence assessment. *CXC Report, May*.

Verhagen, W., Van Teeffelen, A. J., Compagnucci, A. B., Poggio, L., Gimona, A., & Verburg, P. H. (2016). Effects of landscape configuration on mapping ecosystem service capacity: a review of evidence and a case study in Scotland. Landscape ecology, 31(7), 1457-1479.

Vis, C., Hudon, C., & Carignan, R. (2003). An evaluation of approaches used to determine the distribution and biomass of emergent and submerged aquatic macrophytes over large spatial scales. Aquatic Botany, 77(3), 187-201.

Walmsley, J., Carden, M., Revenga, C., Sagona, F. and Smith, M., 2001. Indicators of sustainable development for catchment management in South Africa-Review of indicators from around the world. Water Sa, 27(4), pp.539-550.

Wang, C., Gimona, A., Compagnucci, A. B., and Jiang, Y.: Use of Digital and 3D Visualisation Technology in Planning for Woodland Expansion, EGU General Assembly 2020, Online, 4–8 May 2020, EGU2020-1243, https://doi.org/10.5194/egusphere-egu2020-1243, 2019

Warner, J.F., 2006. More sustainable participation? Multi-stakeholder platforms for integrated catchment management. Water resources development, 22(1), pp.15-35.

Watson, R., S. Albon, R. Aspinall, M. Austen, B. Bardgett, I. Bateman, P. Berry, W. Bird, R. Bradbury, and C. Brown. 2011. UK National Ecosystem Assessment: understanding nature's value to society. Synthesis of key findings. Information Press.

Watts, K., Eycott, A. E., Handley, P., Ray, D., Humphrey, J. W., & Quine, C. P. (2010). Targeting and evaluating biodiversity conservation action within fragmented landscapes:

an approach based on generic focal species and least-cost networks. Landscape Ecology, 25(9), 1305-1318.

Wentworth, J. 2014. Catchment - Wide Flood Management. Parliamentary Office of Science and Technology POSTnote Number 484.

Werritty A and Hoey T B (2002) Trends and Issues: Geomorphological Changes and Trends in Scotland – Trends in River Channels and Processes, Report to Scottish Natural Heritage.

Werritty, A. and Chatterton, J., 2004. Future flooding Scotland. Department of Trade and Industry.

Wightman, A. (2013). The poor had no lawyers: who owns Scotland and how they got it. Birlinn.

Williams, P. H. (1998). Key sites for conservation: area-selection methods for biodiversity. *Conservation in a changing world*, 211-249.

With, K. A., Gardner, R. H., & Turner, M. G. (1997). Landscape connectivity and population distributions in heterogeneous environments. Oikos, 151-169.

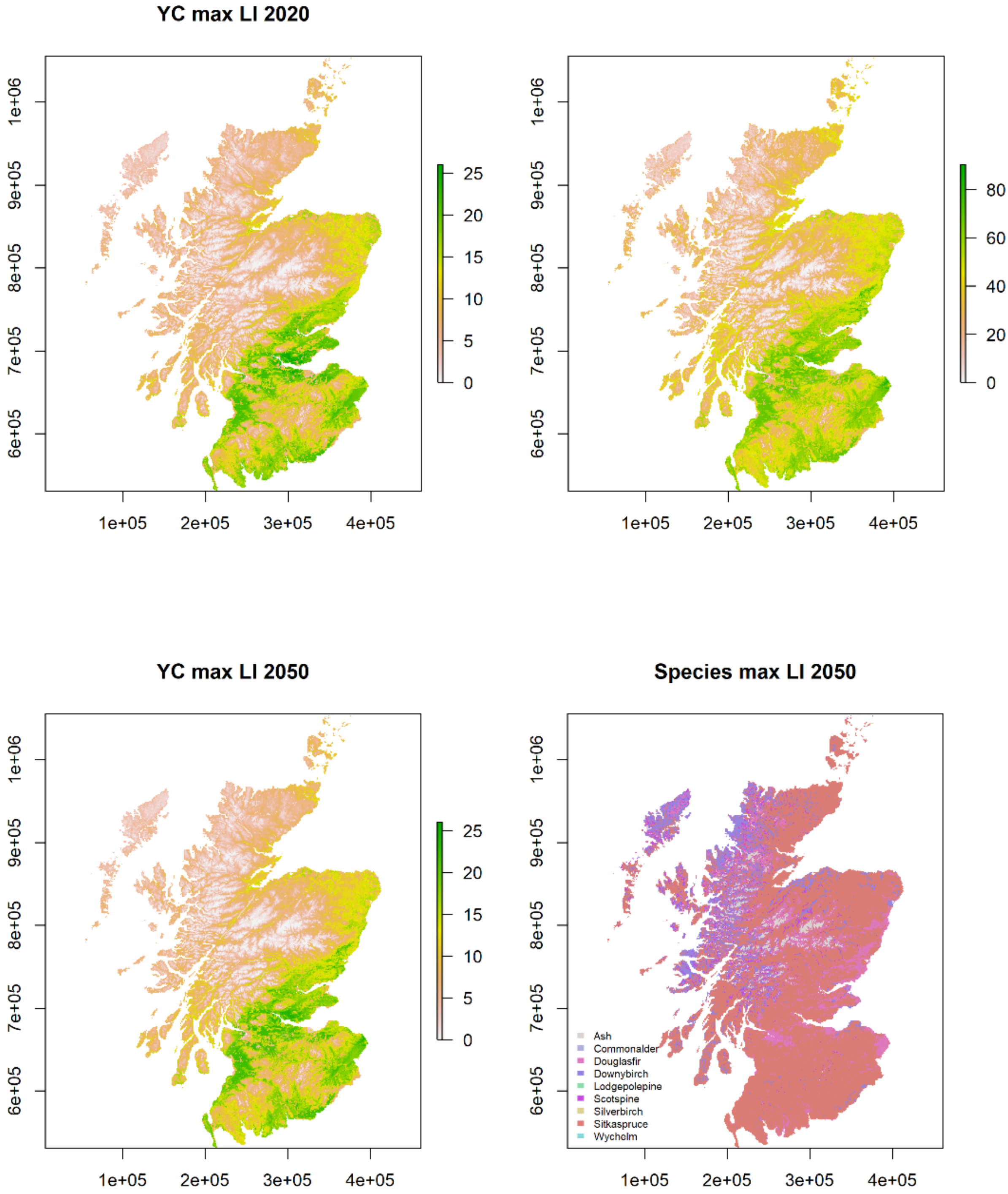
Wray A., (2015). The British survey of fertiliser practice. Fertiliser use on farm crops for crop year 2015. Department for Environment Food & Rural Affairs (DEFRA).

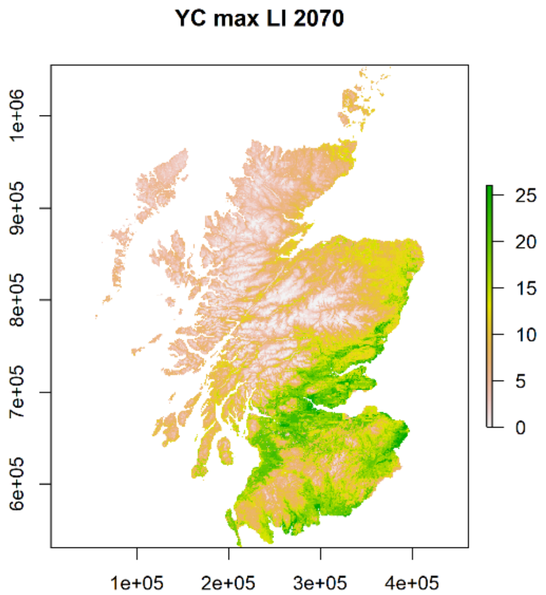
Wright, S. J., & Muller-Landau, H. C. (2006). The Future of Tropical Forest Species 1. Biotropica: The Journal of Biology and Conservation, 38(3), 287-301.

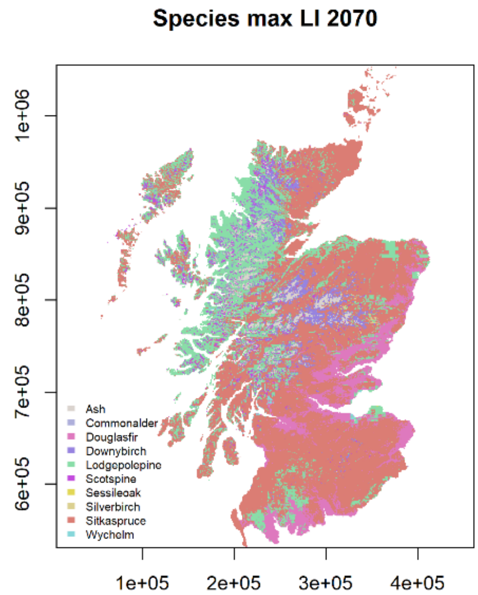
Zucchella, A., & Previtali, P. (2019). Circular business models for sustainable development: A “waste is food” restorative ecosystem. Business Strategy and the Environment, 28(2), 274-285.

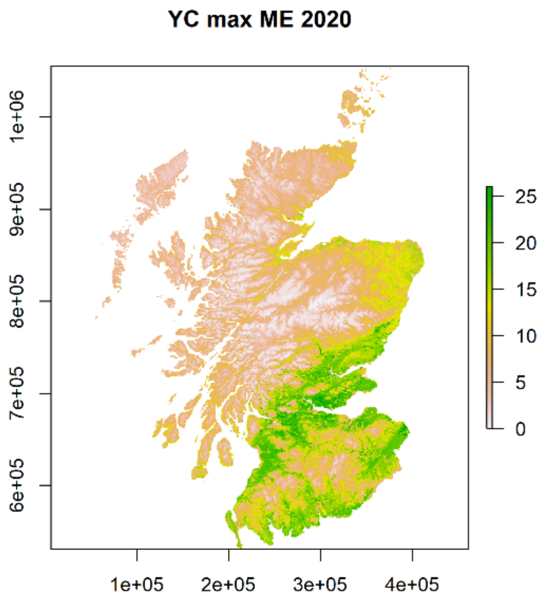
Appendix A

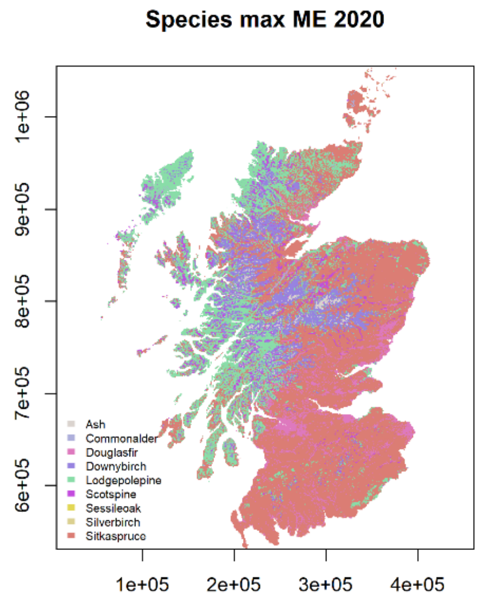
The Following maps are the results of the highest YC modelled for the years and the corresponding species. LI and ME sets indicates modelled YC using the two methods: *lower-end interval* and *median estimates.*



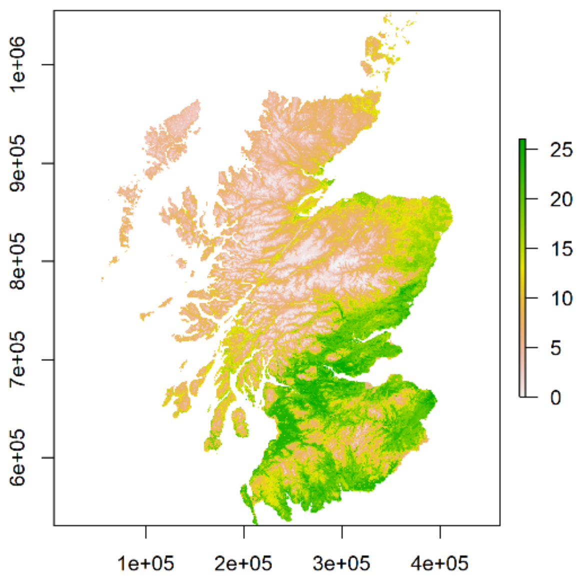

YC max LI 2070
1e+06
9e+05
8e+05
7e+05
6e+05
1e+05
2e+05
3e+05
4e+05
25
20
15
10
5
0


Species max LI 2070
1e+06
9e+05
8e+05
7e+05
6e+05
1e+05
2e+05
3e+05
4e+05
Ash
Commonalder
Douglasfir
Downybirch
Lodgepolepine
Scotspine
Sessileoak
Silverbirch
Sitkaspruce
Wycholm

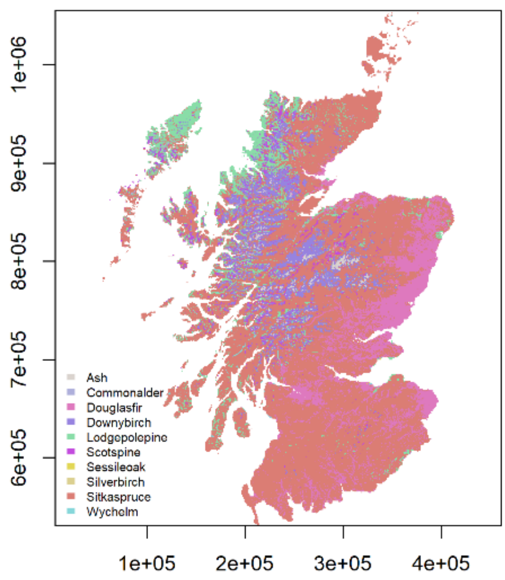

YC max ME 2020
1e+06
9e+05
8e+05
7e+05
6e+05
1e+05
2e+05
3e+05
4e+05
25
20
15
10
5
0


Species max ME 2020
1e+06
9e+05
8e+05
7e+05
6e+05
1e+05
2e+05
3e+05
4e+05
Ash
Commonalder
Douglasfir
Downybirch
Lodgepolepine
Scotspine
Sessileoak
Silverbirch
Sitkaspruce

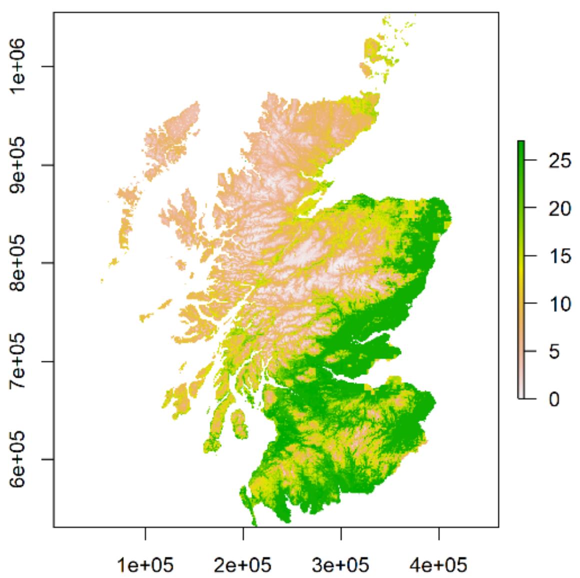
YC max ME 2050



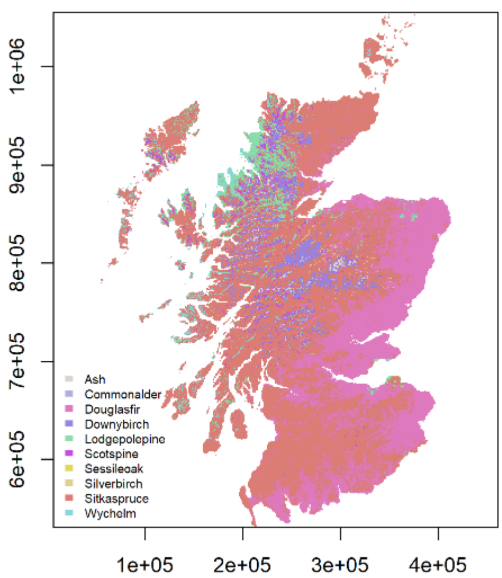
Species max ME 2050

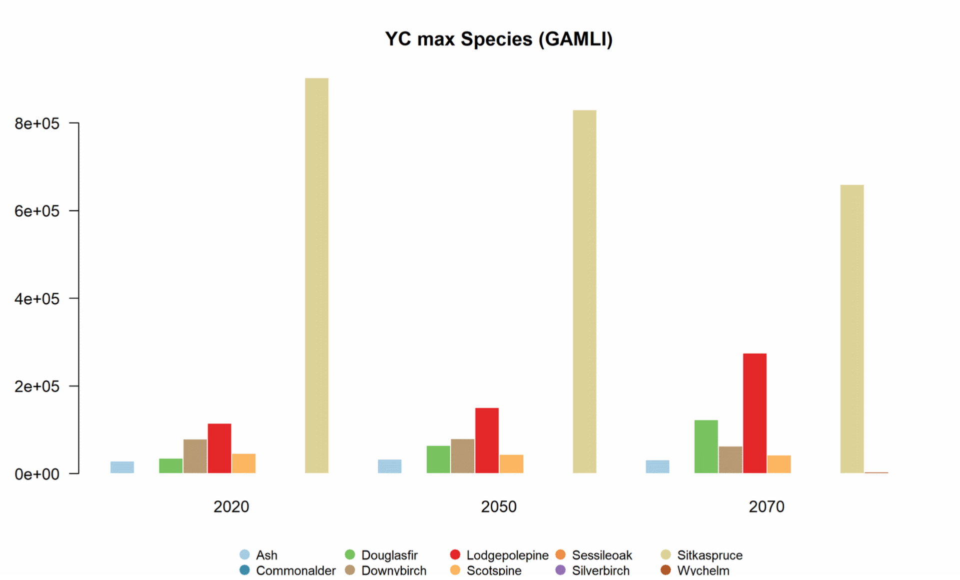


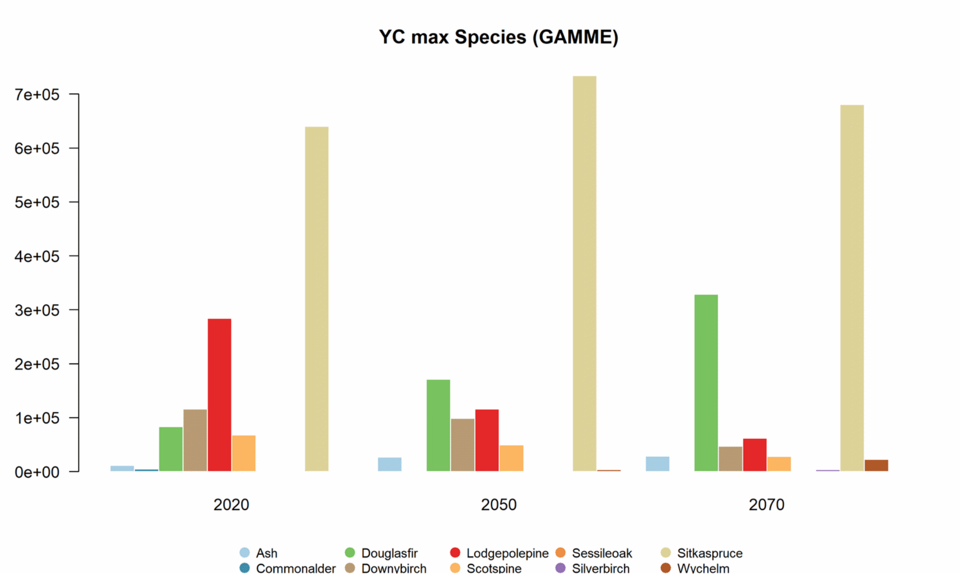
YC max ME 2070



Species max ME 2070

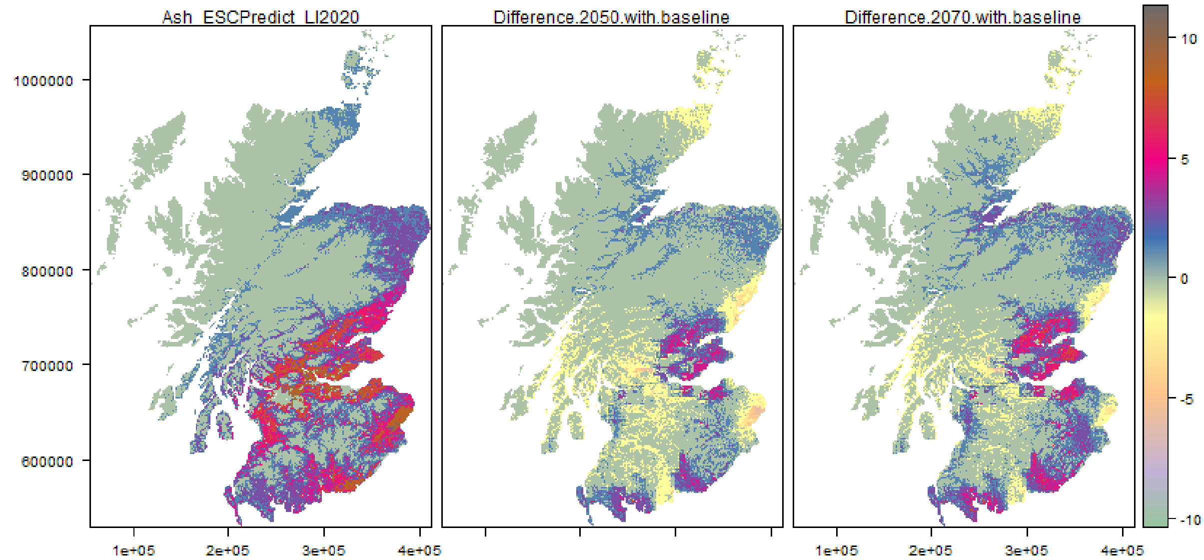


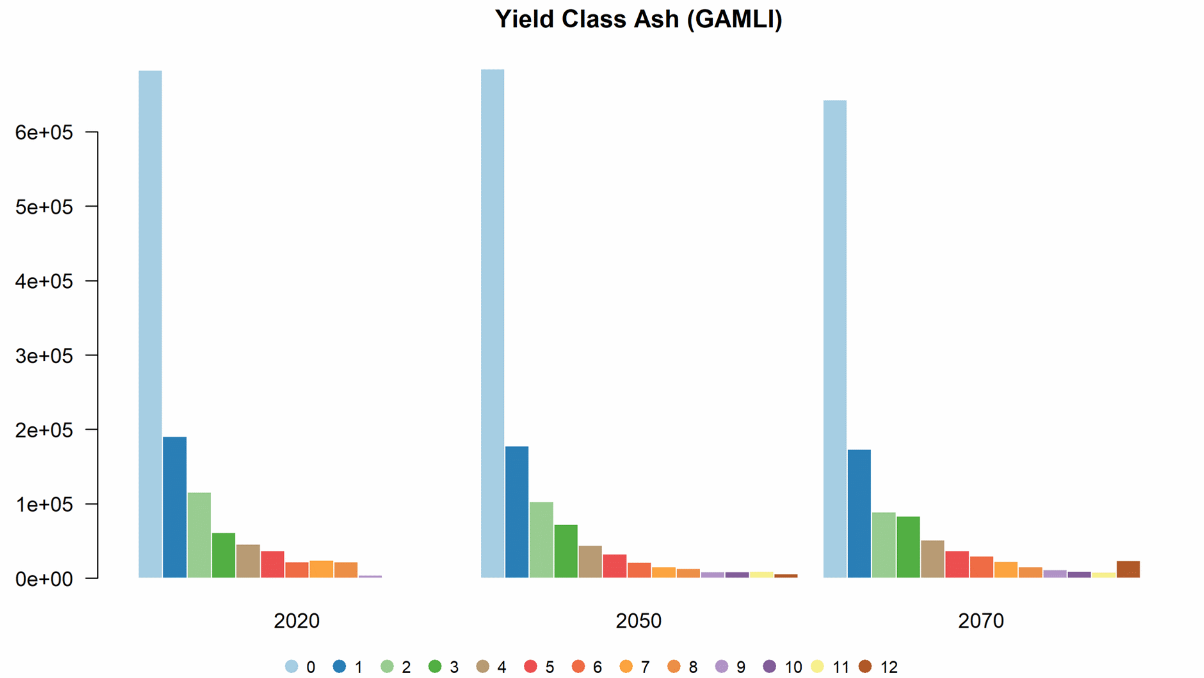

YC max Species (GAMLI)
8e+05
6e+05
4e+05
2e+05
0e+00
2020
2050
2070
Ash
Commonalder
Douglasfir
Downybirch
Lodgepolepine
Scotspine
Sessileoak
Silverbirch
Sitkaspruce
Wychelm

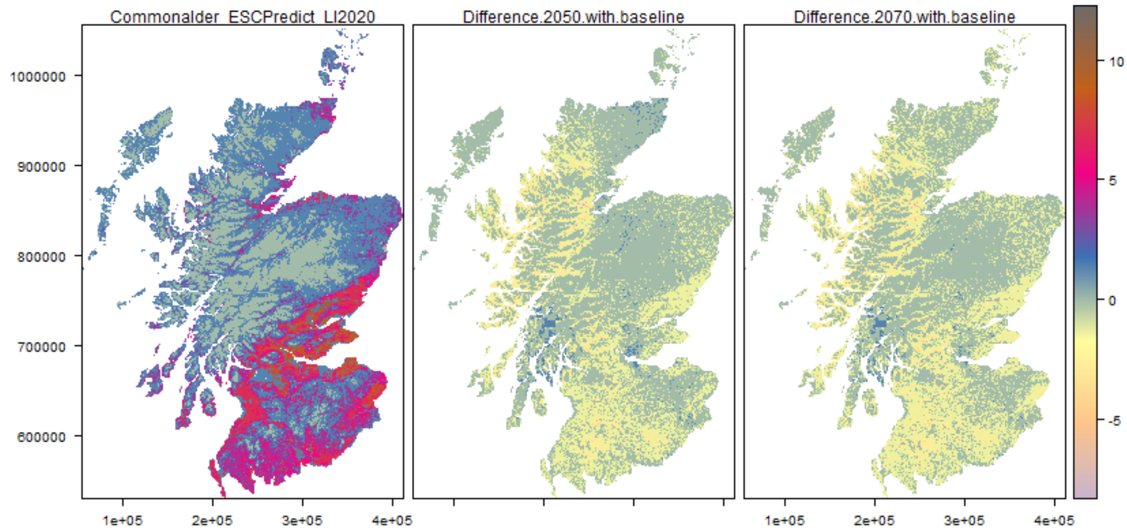

YC max Species (GAMME)
7e+05
6e+05
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
Ash
Commonalder
Douglasfir
Downybirch
Lodgepolepine
Scotspine
Sessileoak
Silverbirch
Sitkaspruce
Wychelm

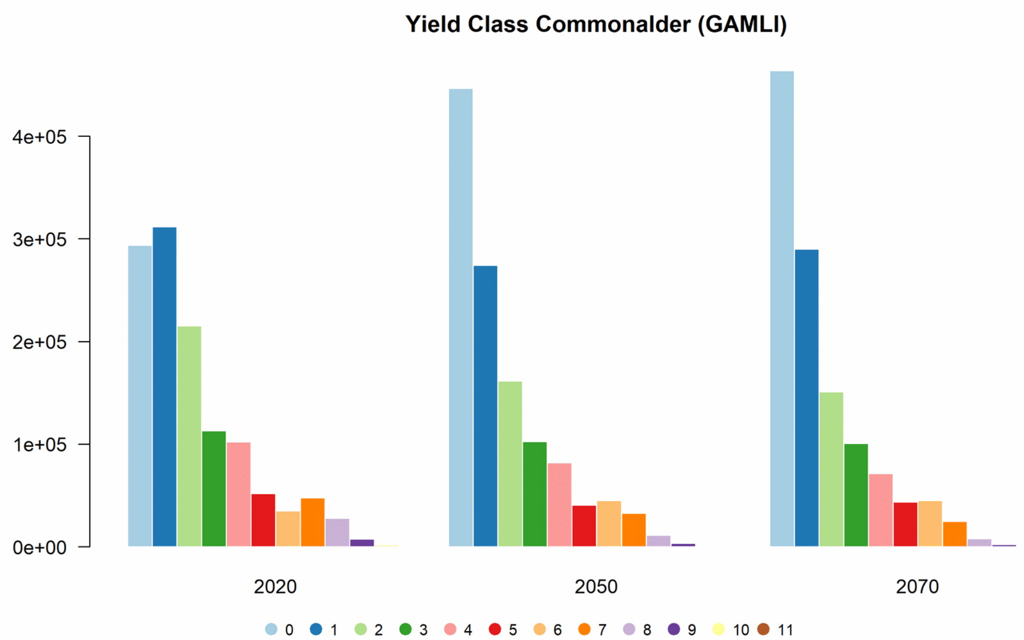
Appendix B

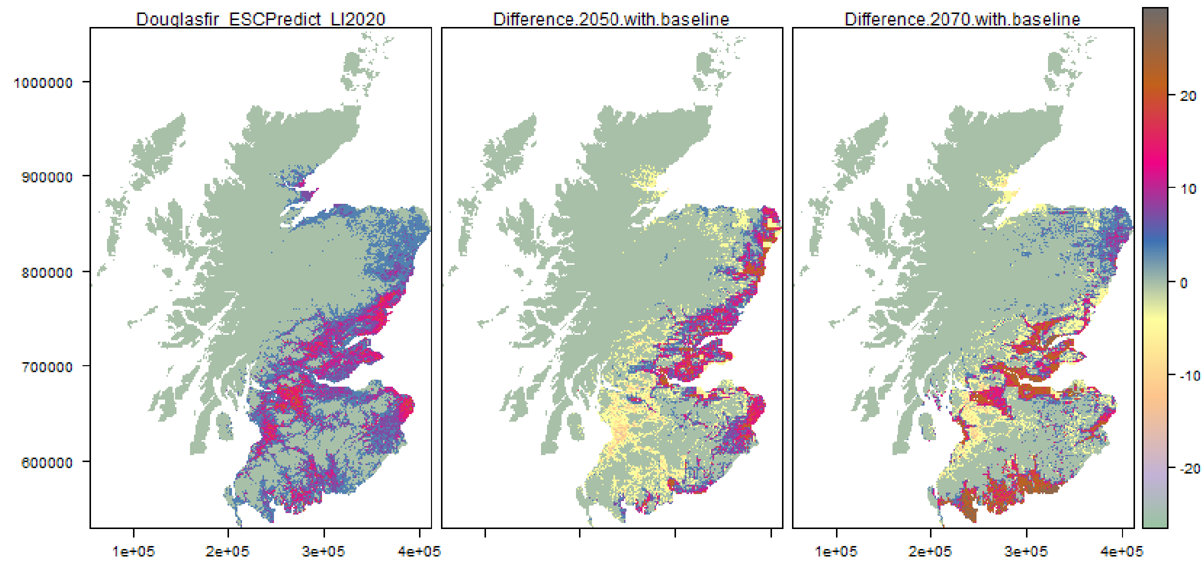
This appendix contains maps and relative charts of the modelled YC for each single species. LI and ME sets indicate modelled YC using the two methods: *lower-end of the confidence interval* and *median estimates.*

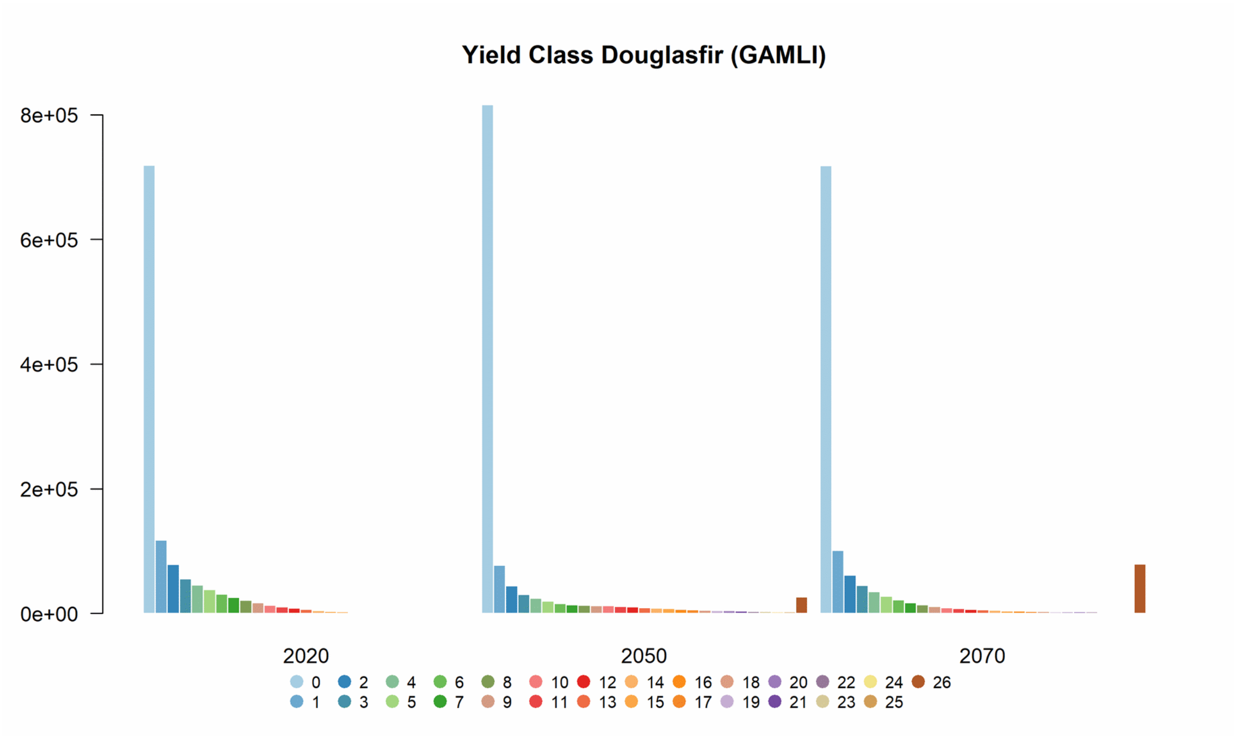


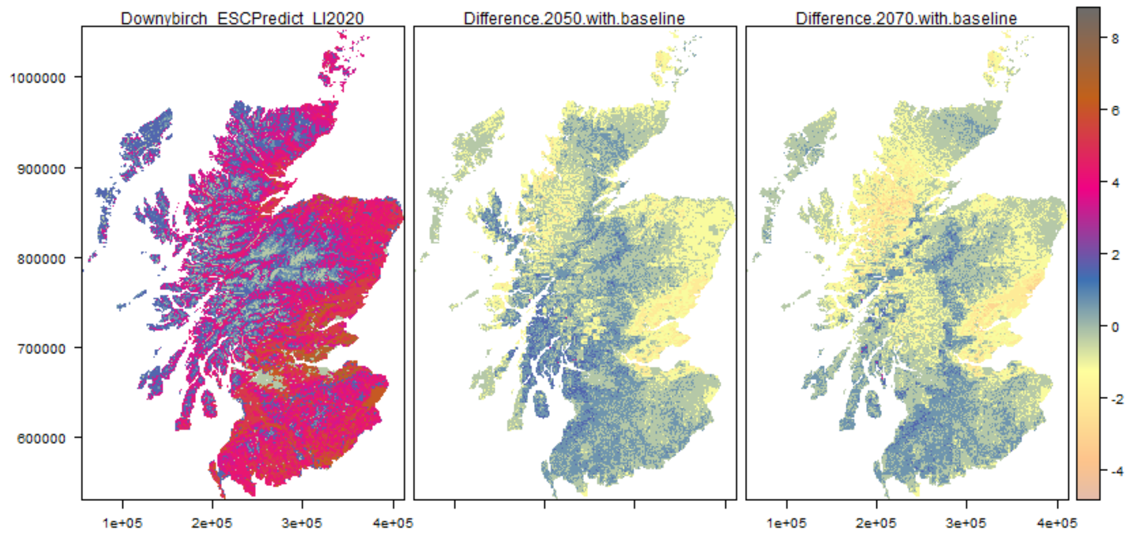



Commonalder_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
10
5
0
-5

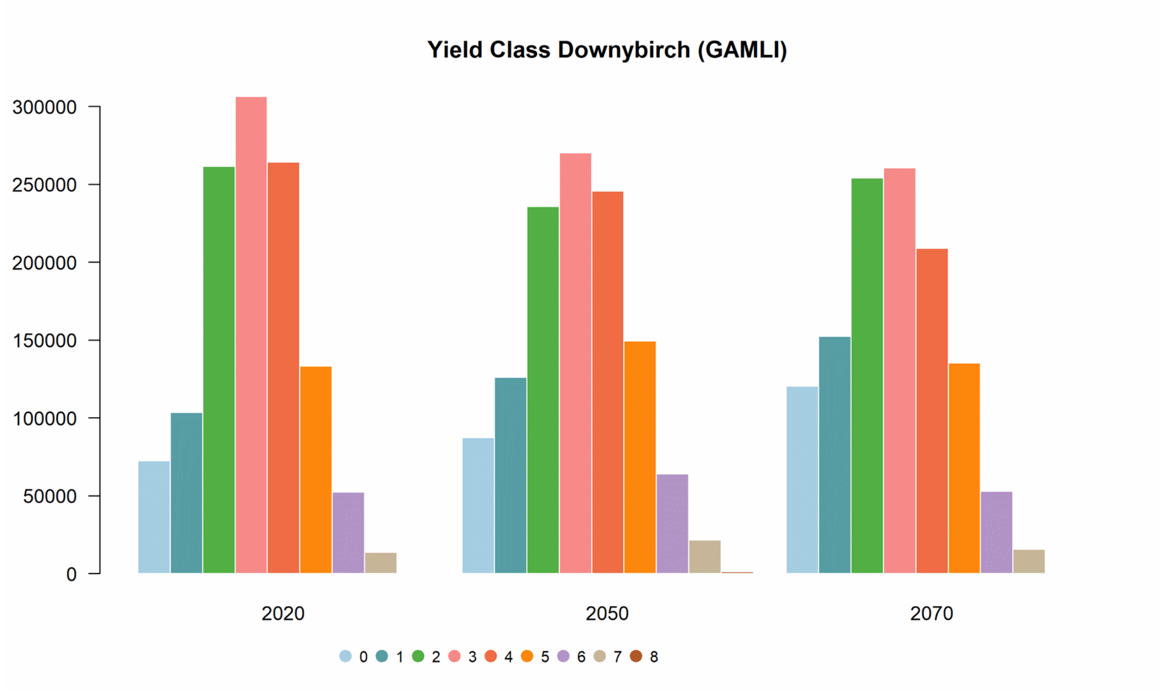

Yield Class Commonalder (GAMLI)
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11


Douglasfir_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
20
10
0
-10
-20

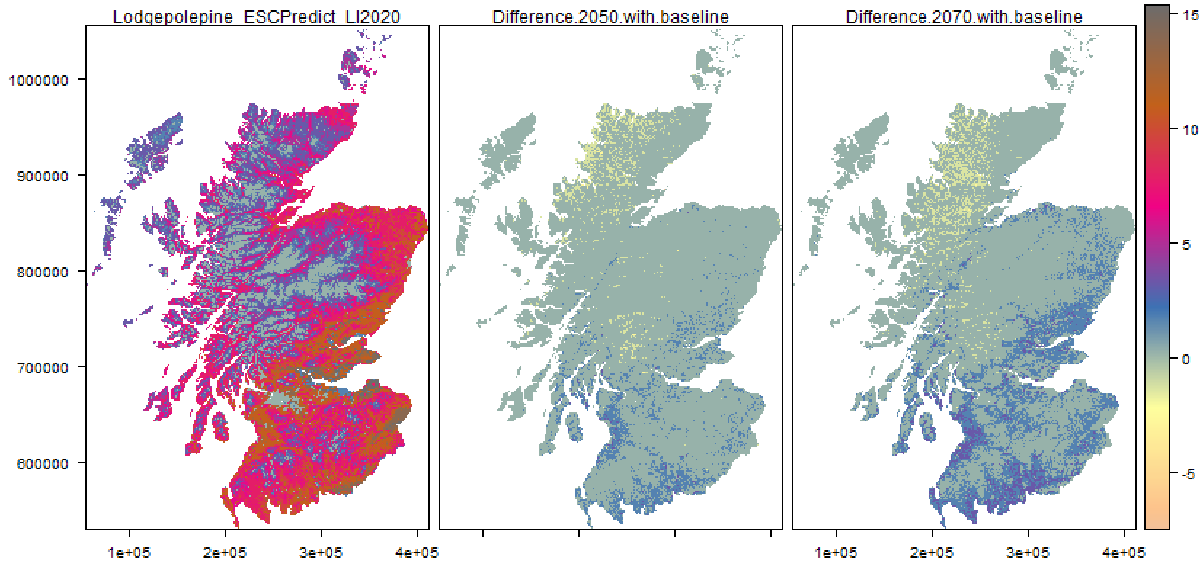

Yield Class Douglasfir (GAMLI)
8e+05
6e+05
4e+05
2e+05
0e+00
2020
2050
2070
0 2 4 6 8 10 12 14 16 18 20 22 24 26
1 3 5 7 9 11 13 15 17 19 21 23 25

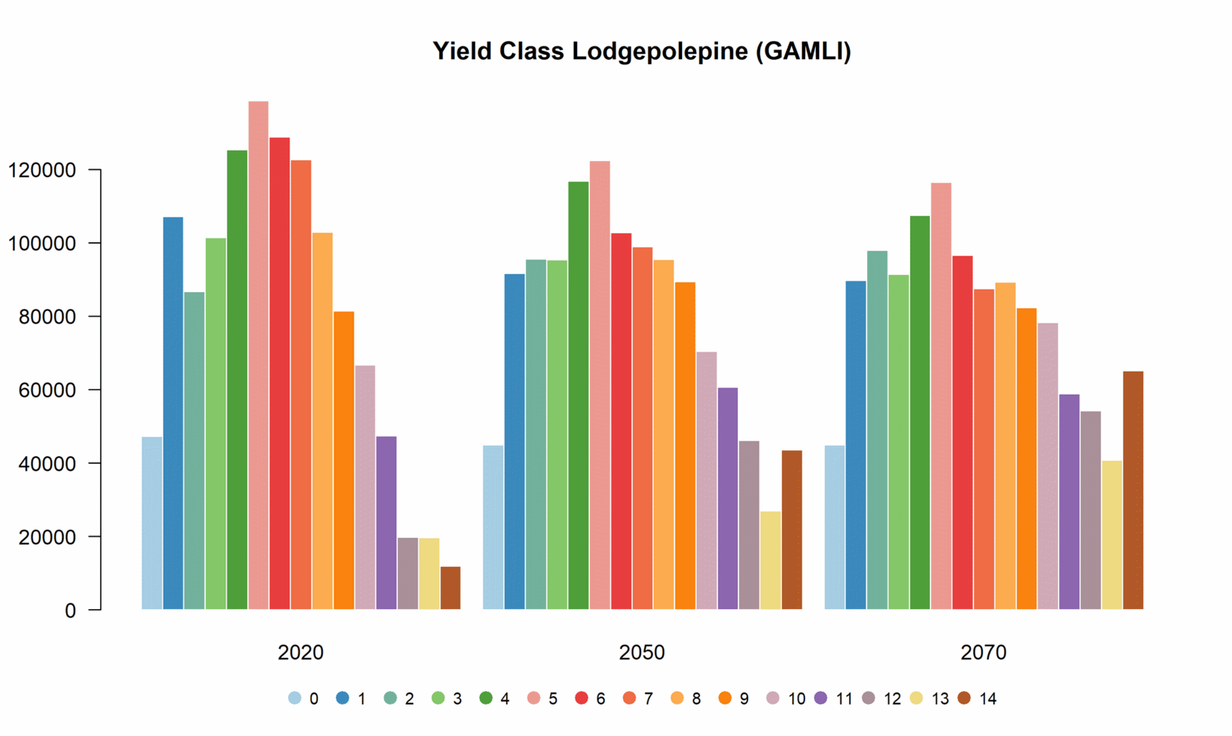


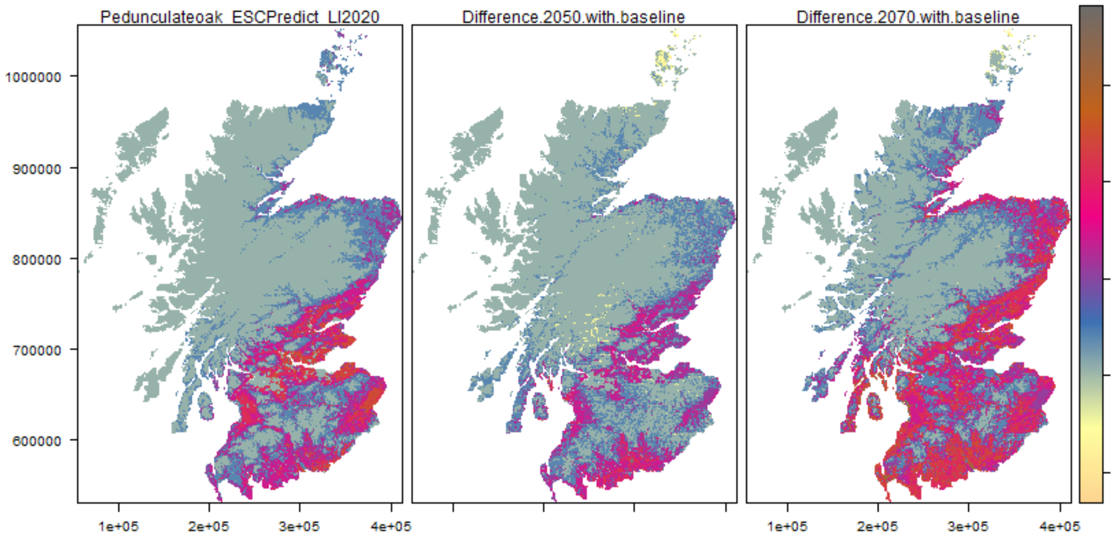
Downybirch_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
8
6
4
2
0
-2
-4

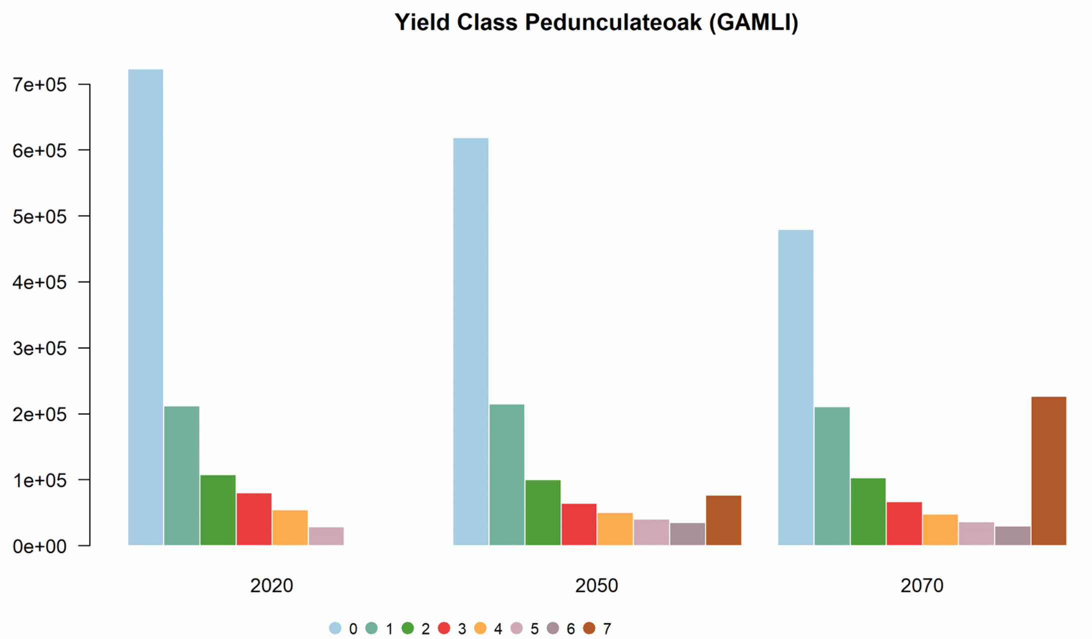


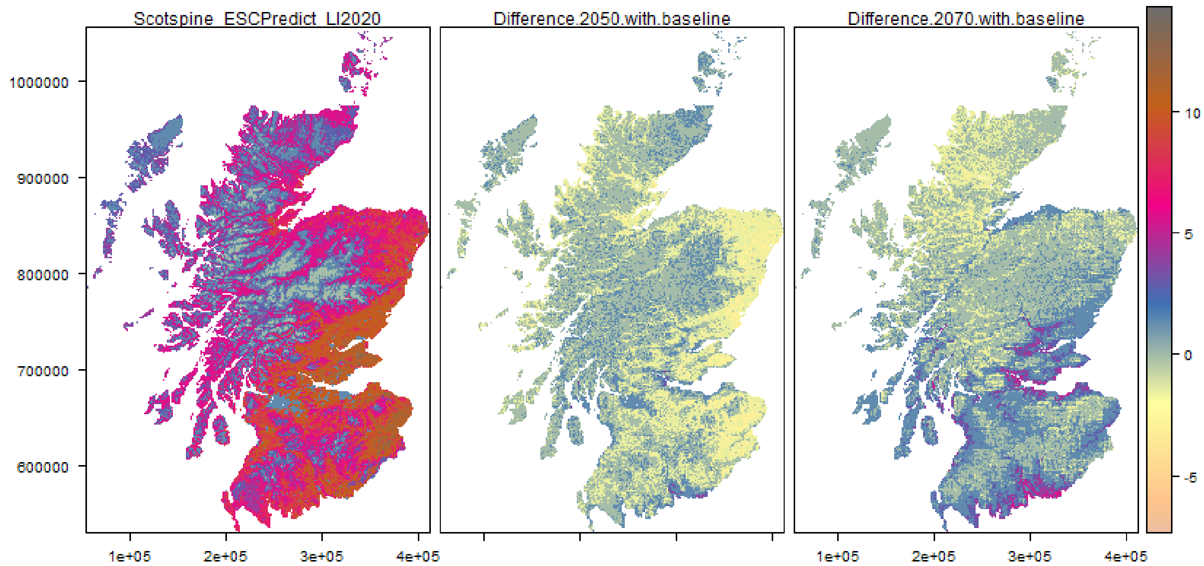
Yield Class Downybirch (GAMLI)
300000
250000
200000
150000
100000
50000
0
2020
2050
2070
0
1
2
3
4
5
6
7
8


Lodgepolepine_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
15
10
5
0
-5

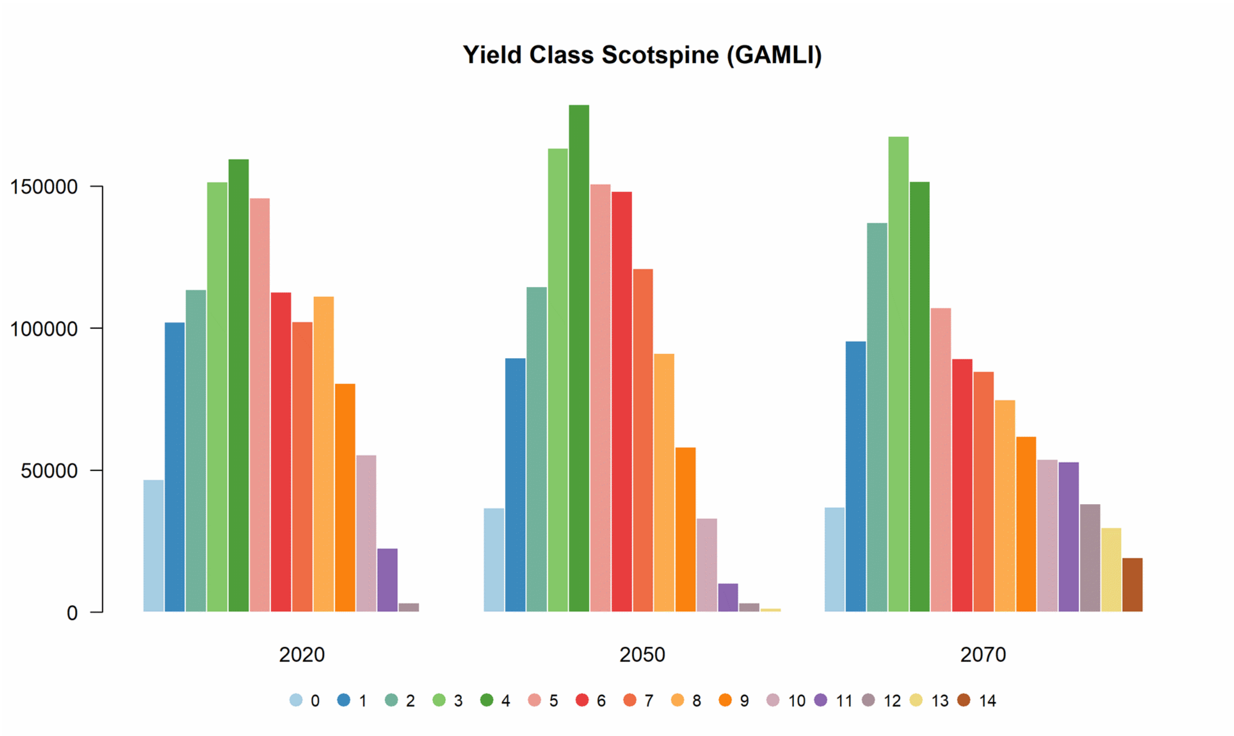

Yield Class Lodgepolepine (GAMLI)
120000
100000
80000
60000
40000
20000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14


Pedunculateoak_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
6
4
2
0
-2

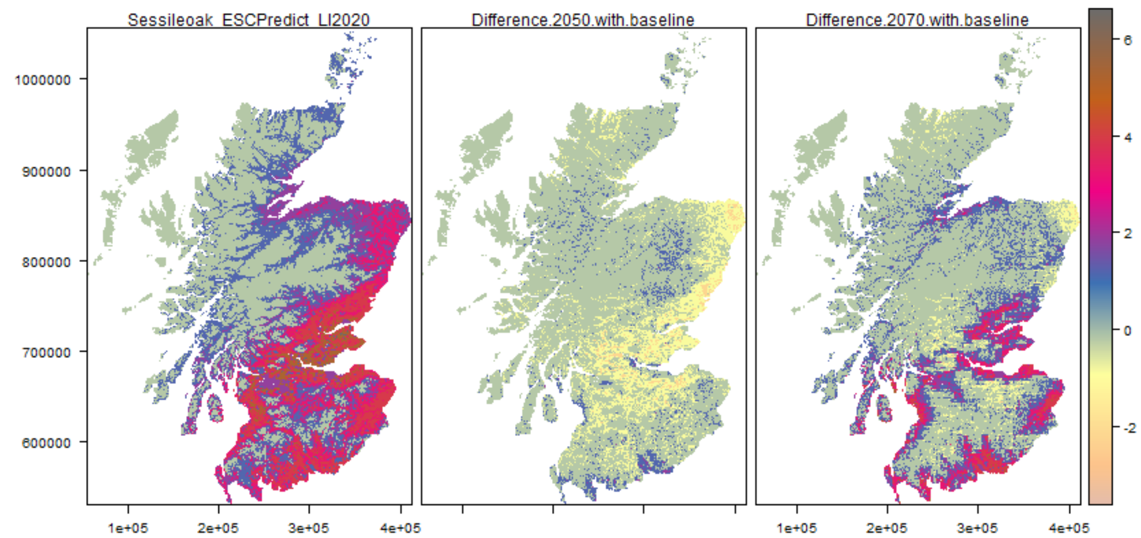

Yield Class Pedunculateoak (GAMLI)
7e+05
6e+05
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7

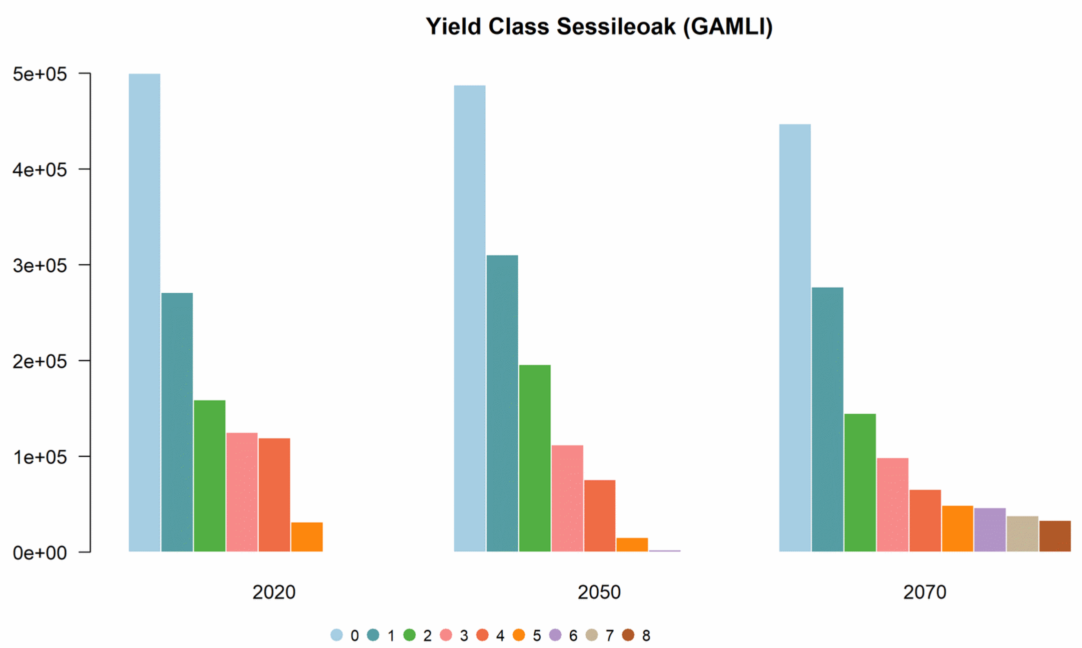


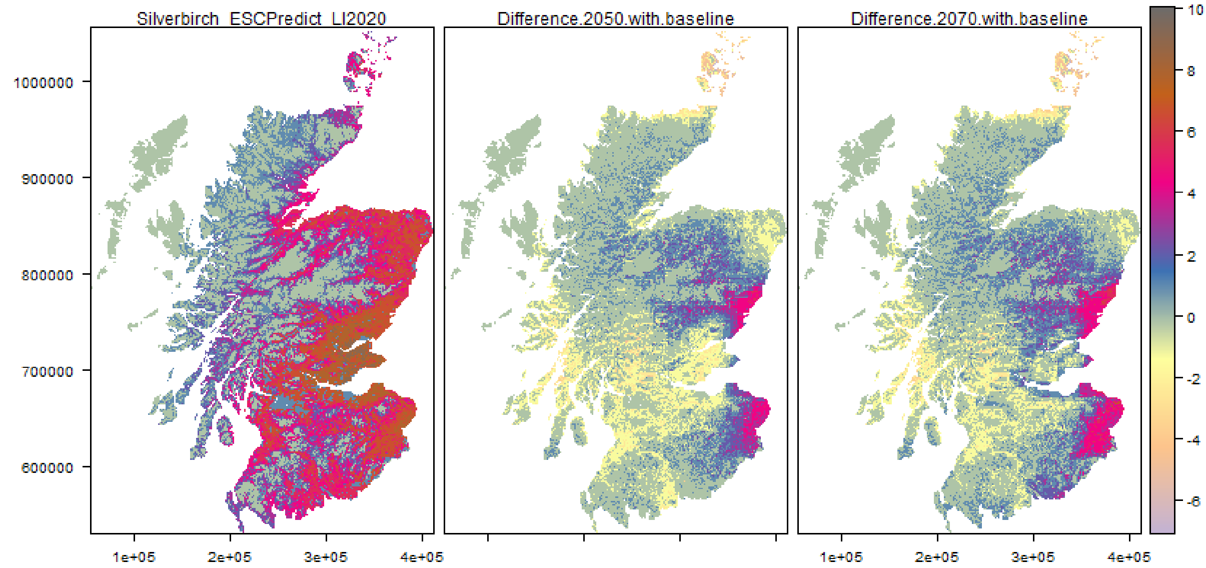
Scotspine_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
10
5
0
-5

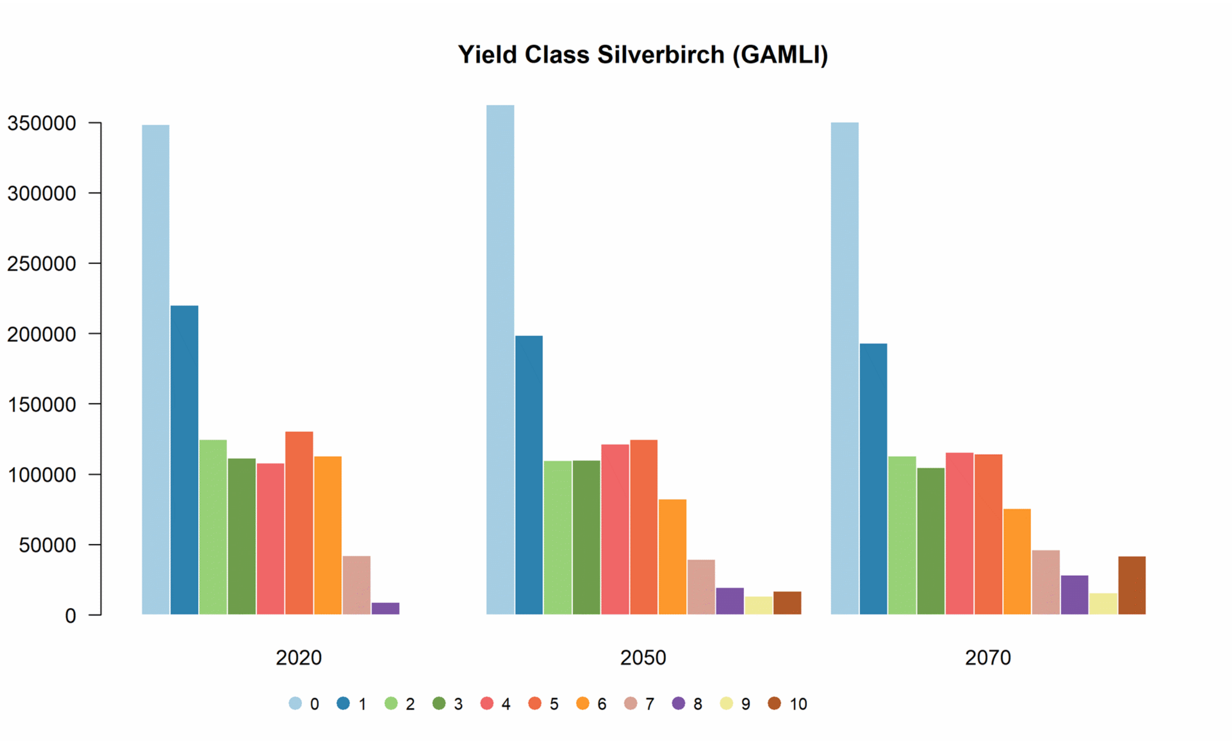


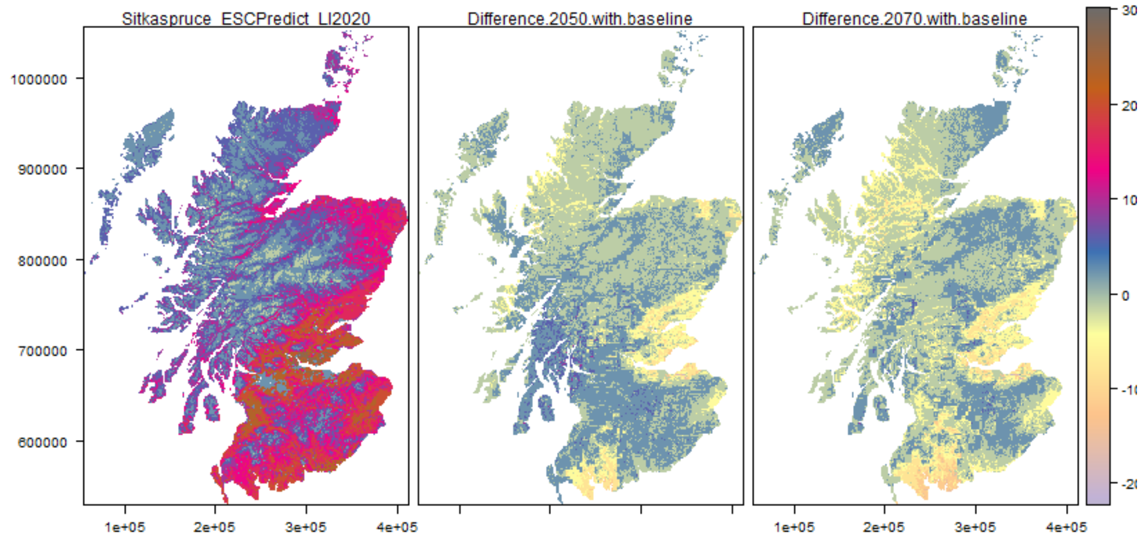
Yield Class Scotspine (GAMLI)
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14


Sessileoak_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
6
4
2
0
-2

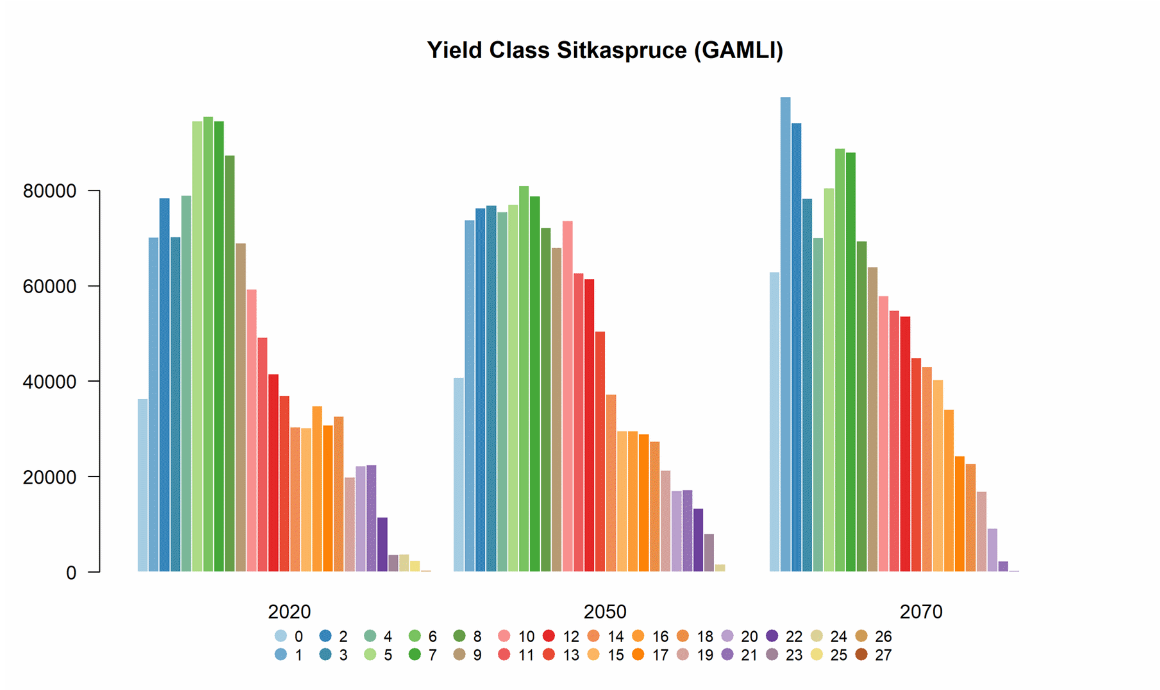

Yield Class Sessileoak (GAMLI)
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7 8


Silverbirch_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
10
8
6
4
2
0
-2
-4
-6

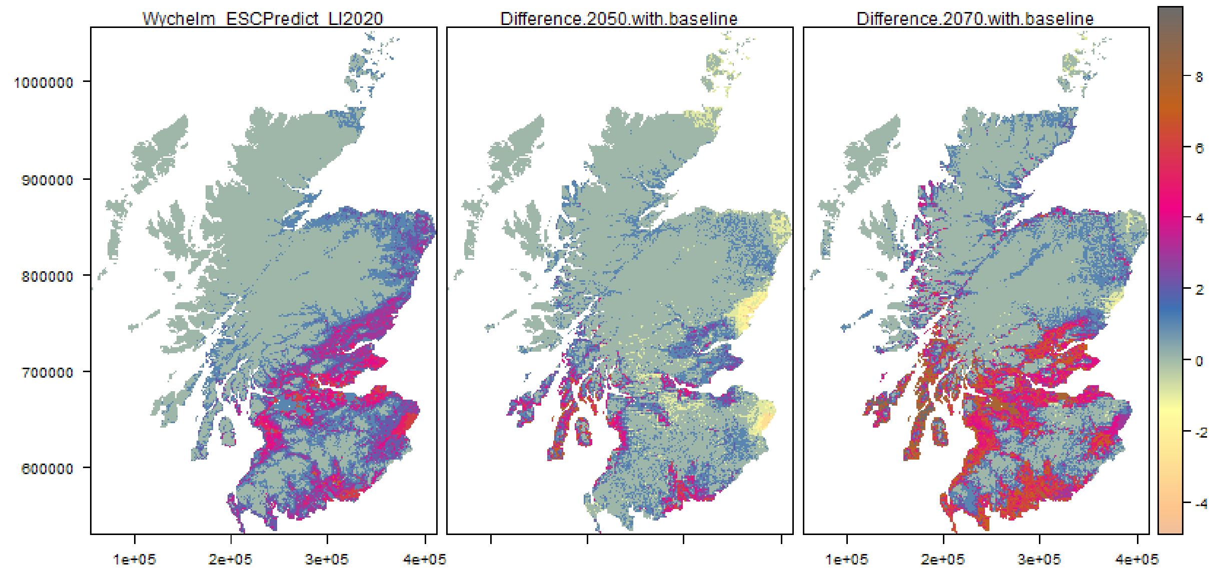

Yield Class Silverbirch (GAMLI)
350000
300000
250000
200000
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10

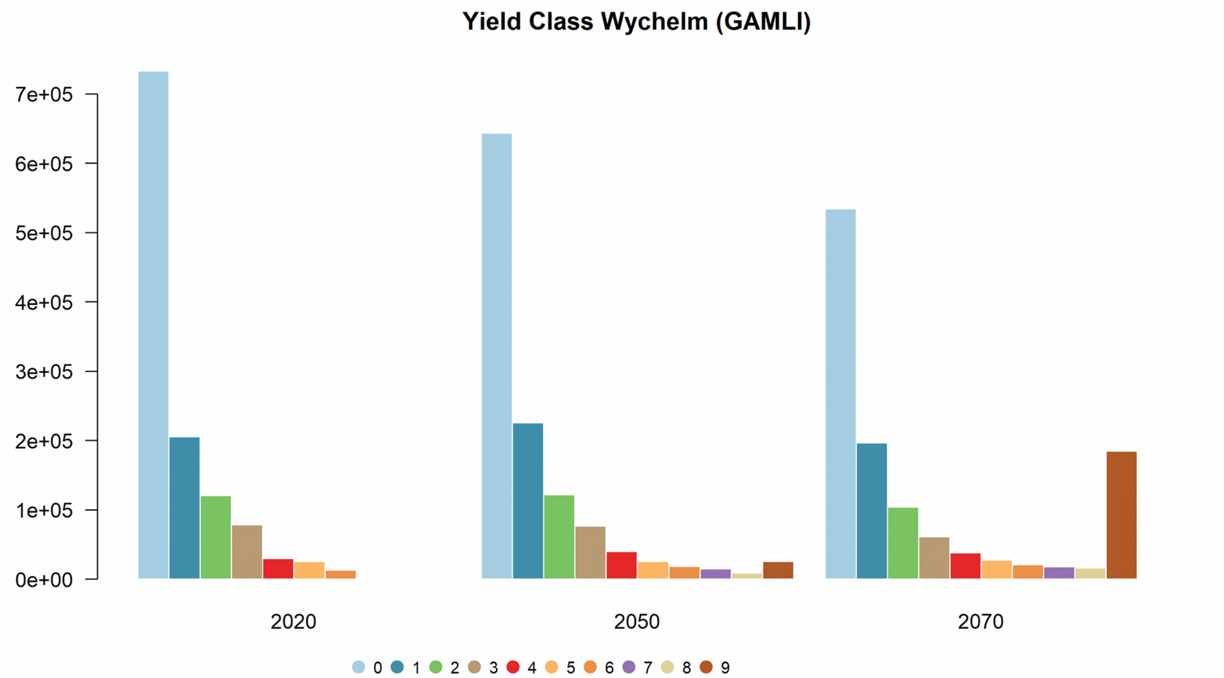


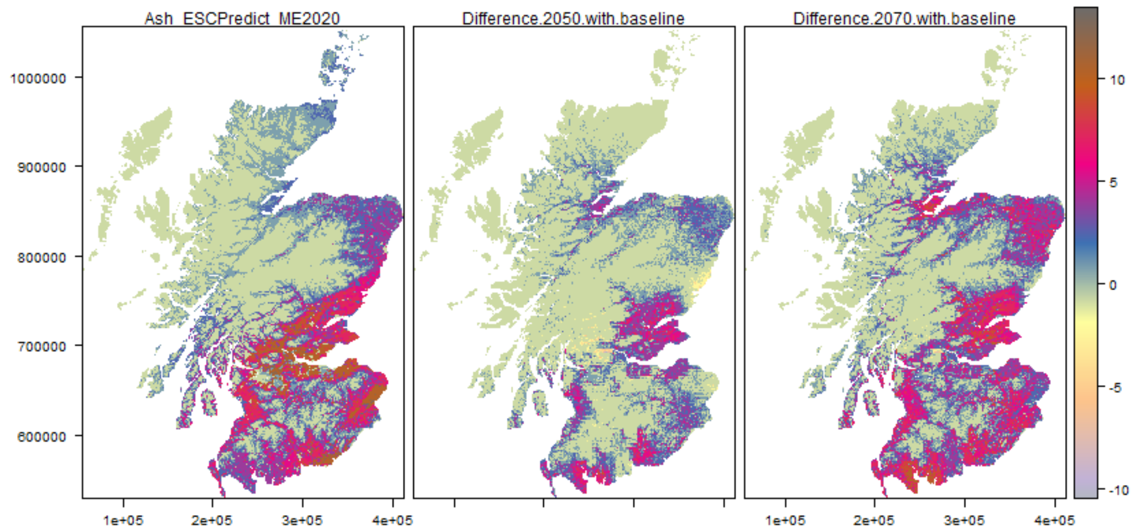
Sitkaspruce_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
30
20
10
0



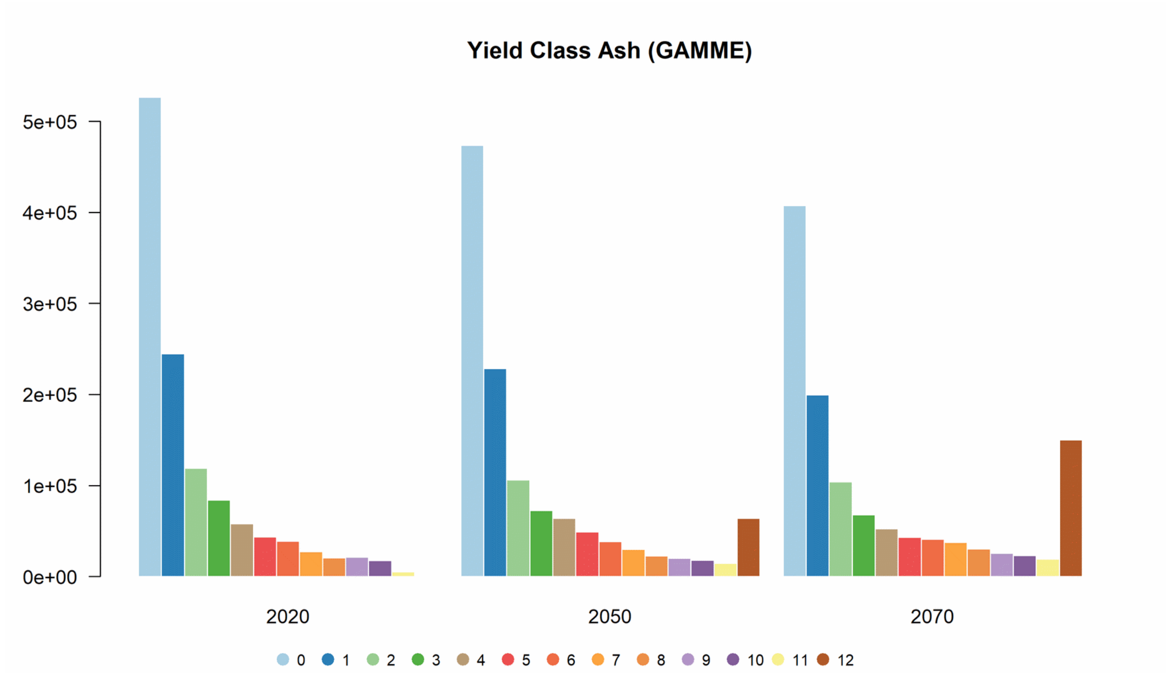
Yield Class Sitkaspruce (GAMLI)
80000
60000
40000
20000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27


Wychelm_ESCPredict_LI2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
8
6
4
2
0
-2
-4

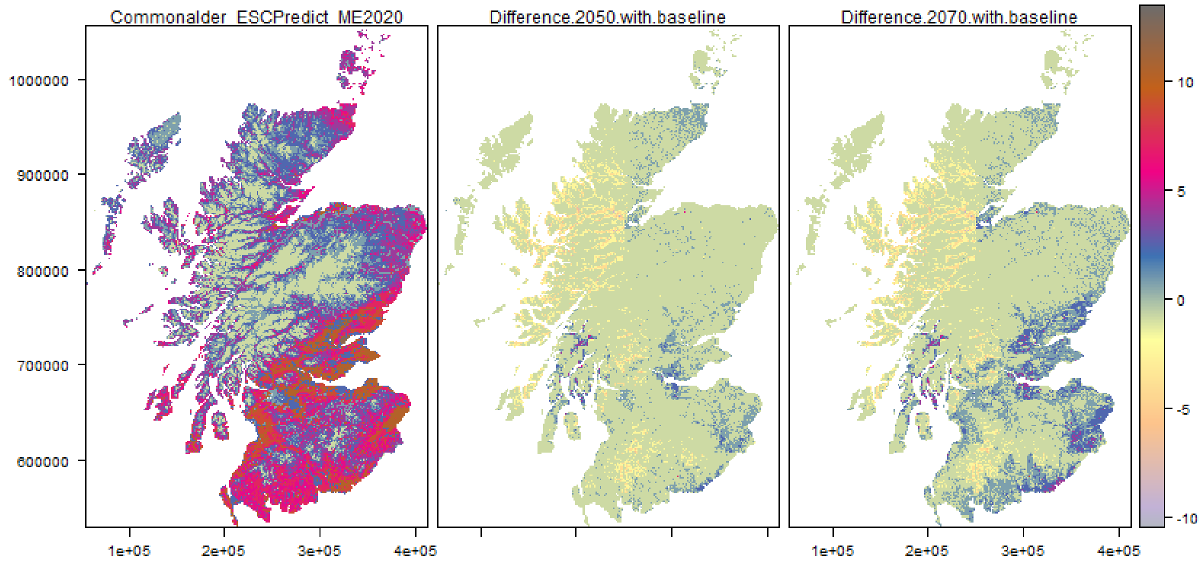

Yield Class Wychelm (GAMLI)
7e+05
6e+05
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7 8 9



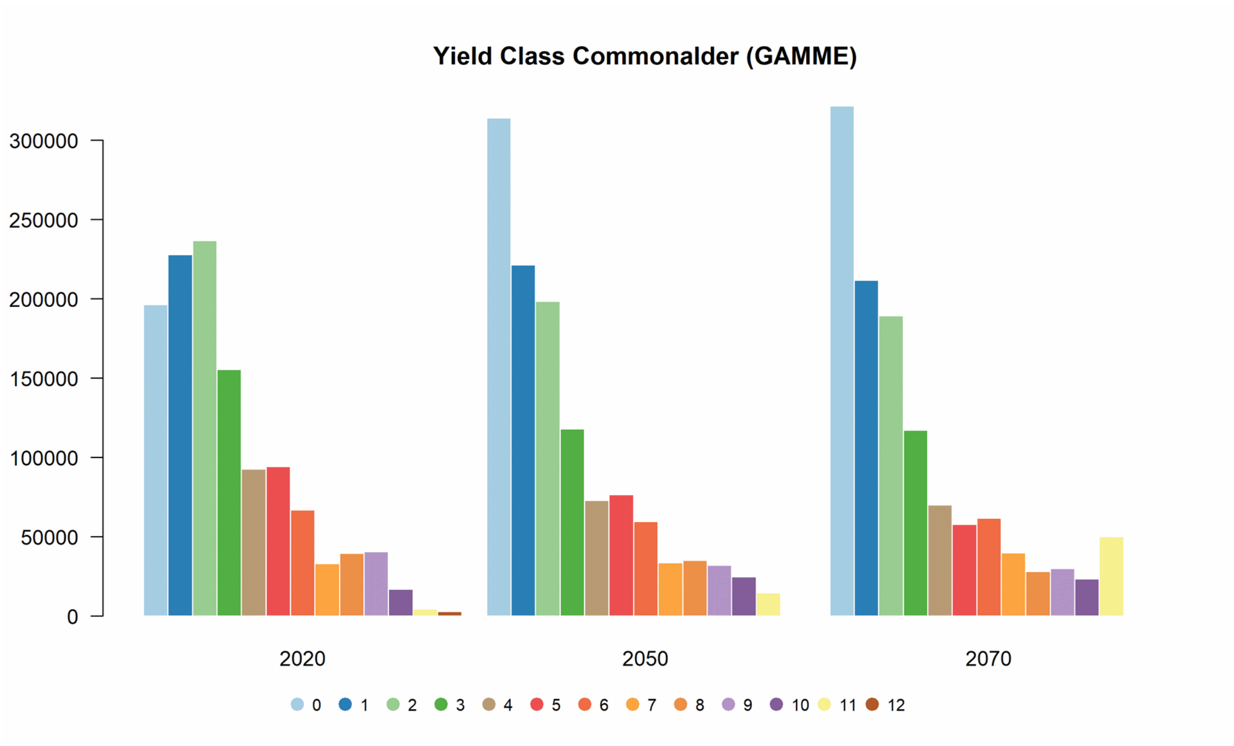
Ash_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
10
5
0
-5
-10



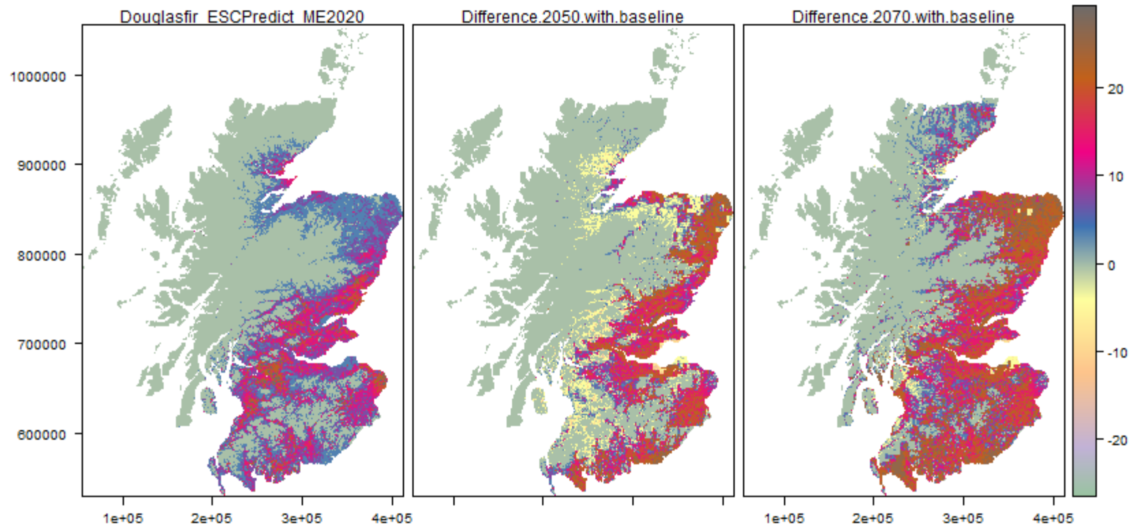
Yield Class Ash (GAMME)
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12



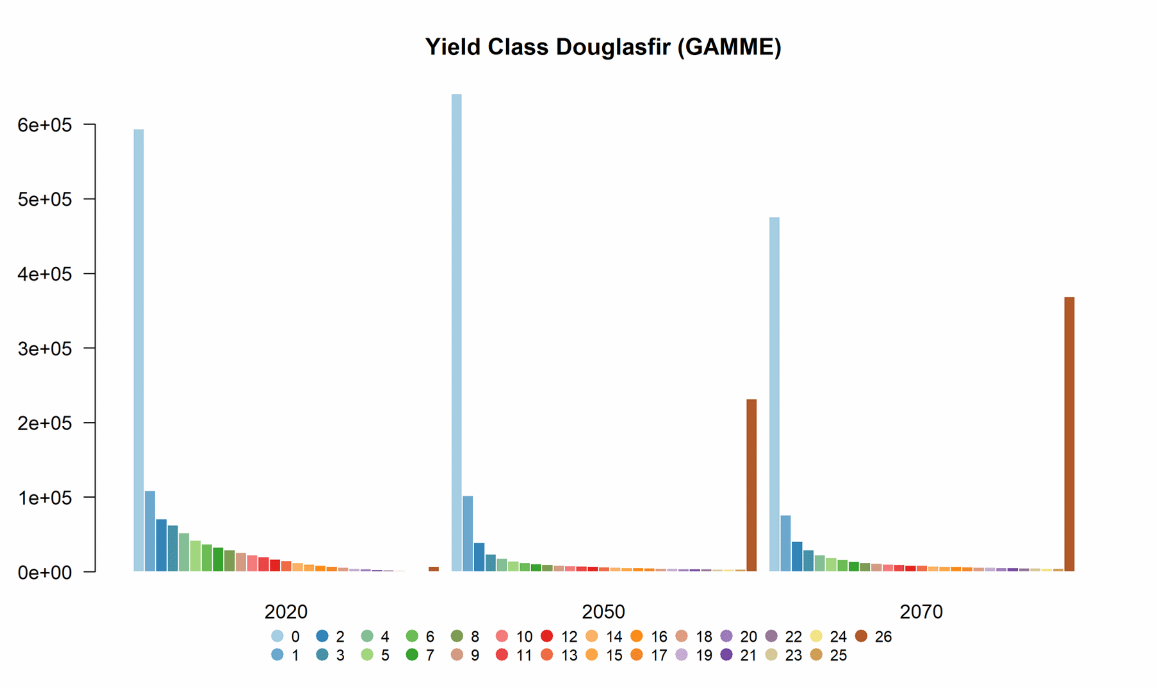
Commonalder_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
10
5
0
-5
-10



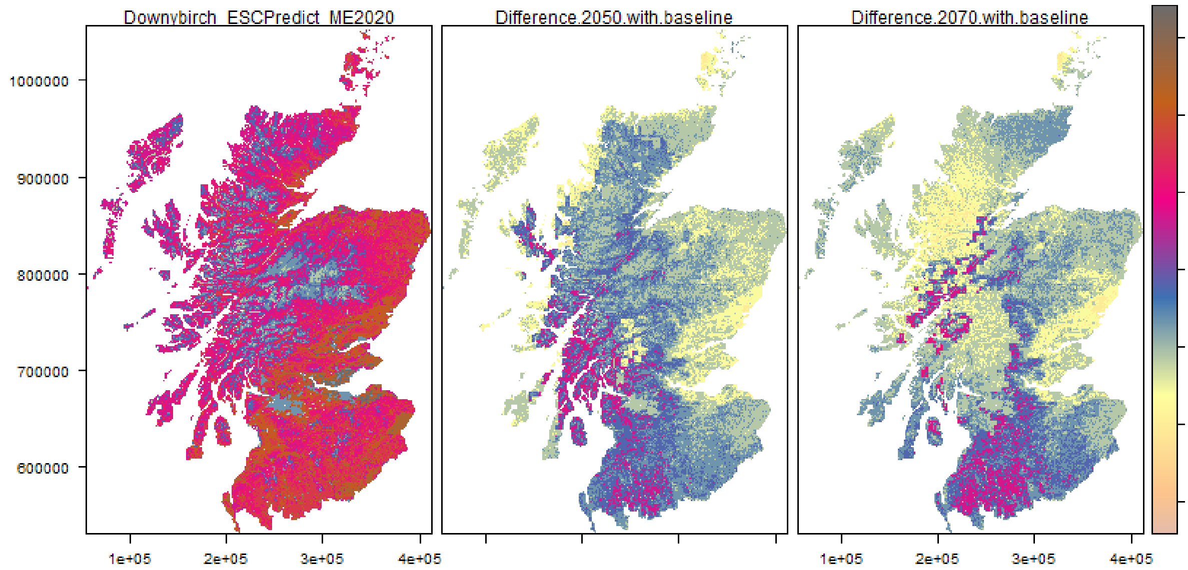
Yield Class Commonalder (GAMME)
300000
250000
200000
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12

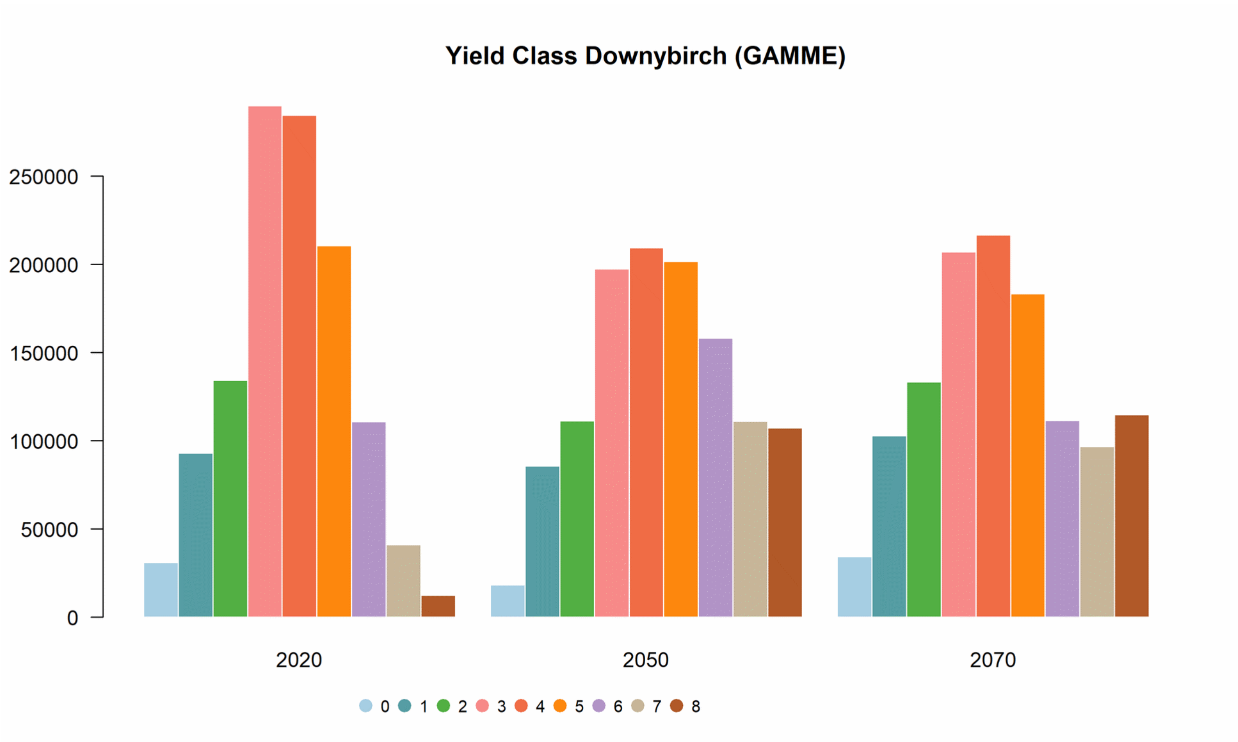


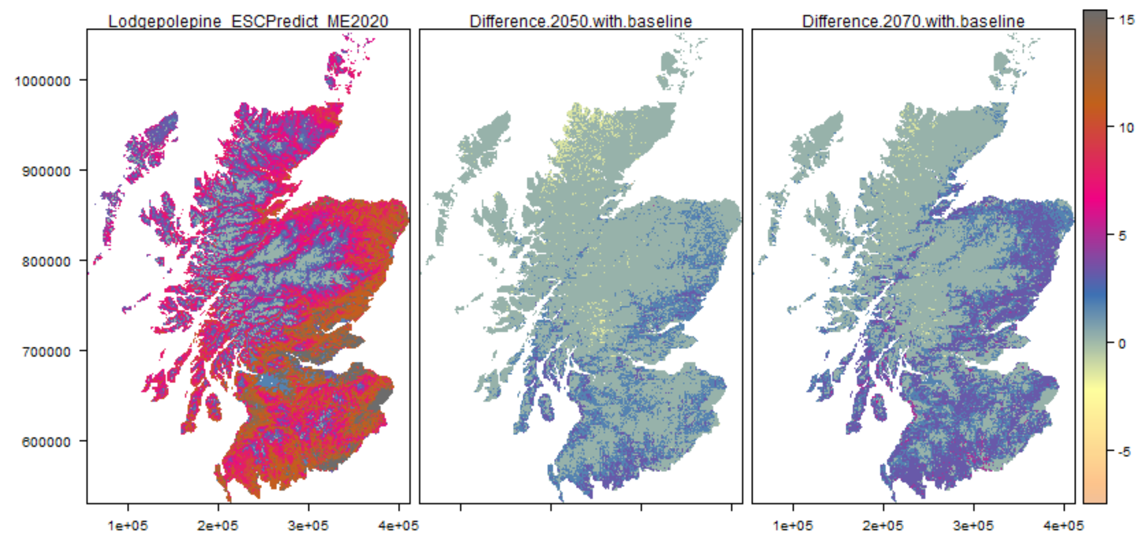
Douglasfir_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
20
10
0
-10
-20

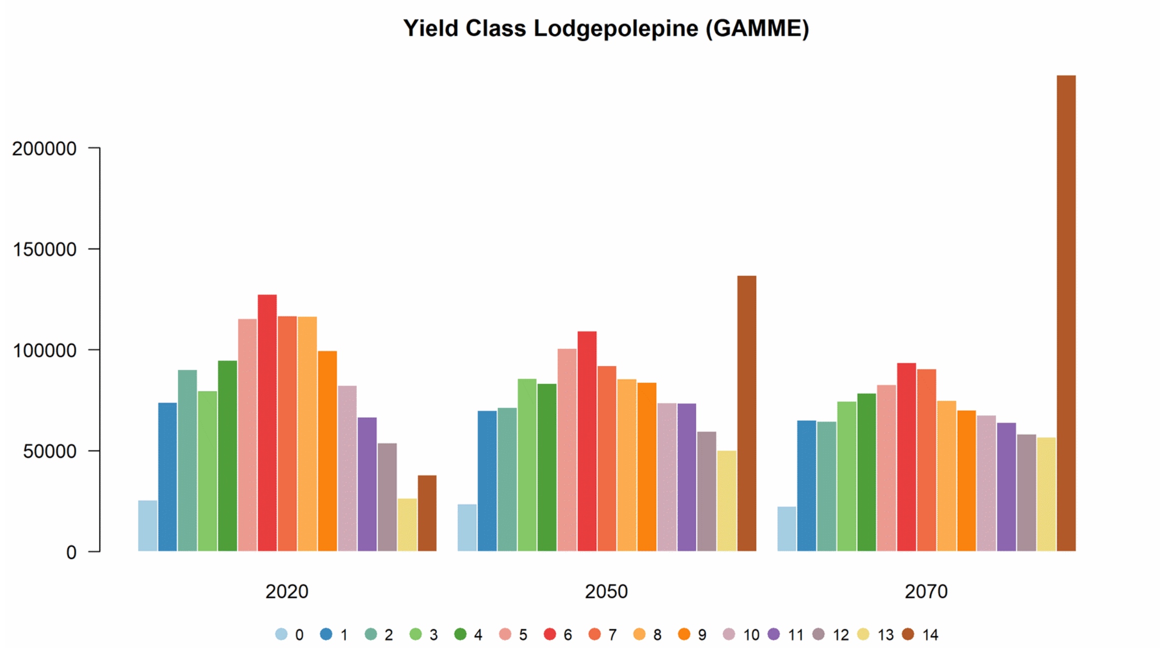


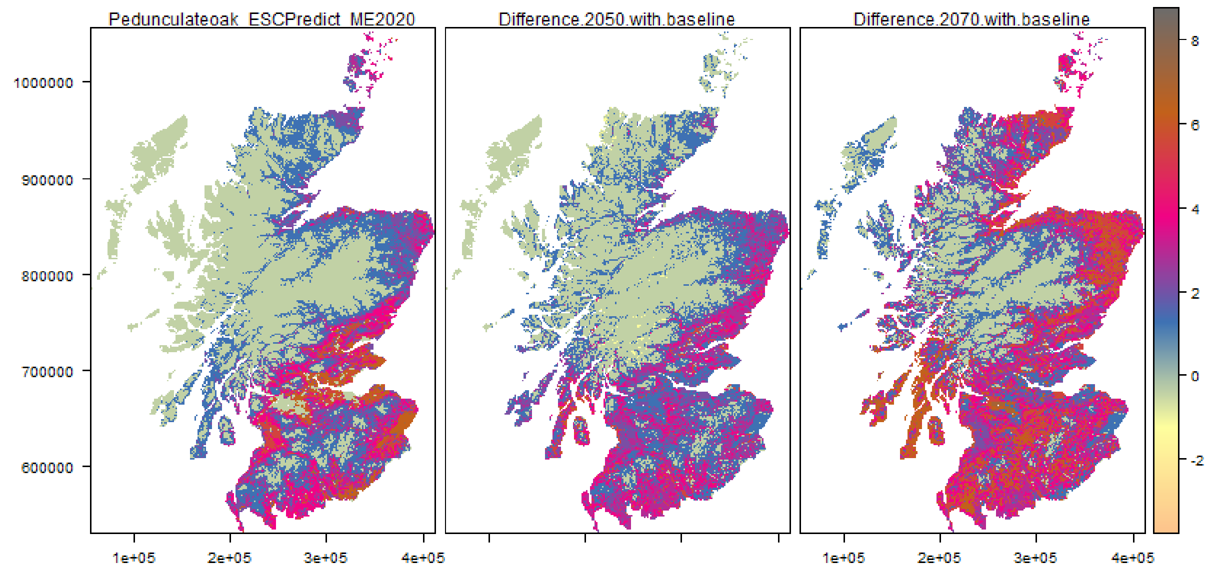
Yield Class Douglasfir (GAMME)
6e+05
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26

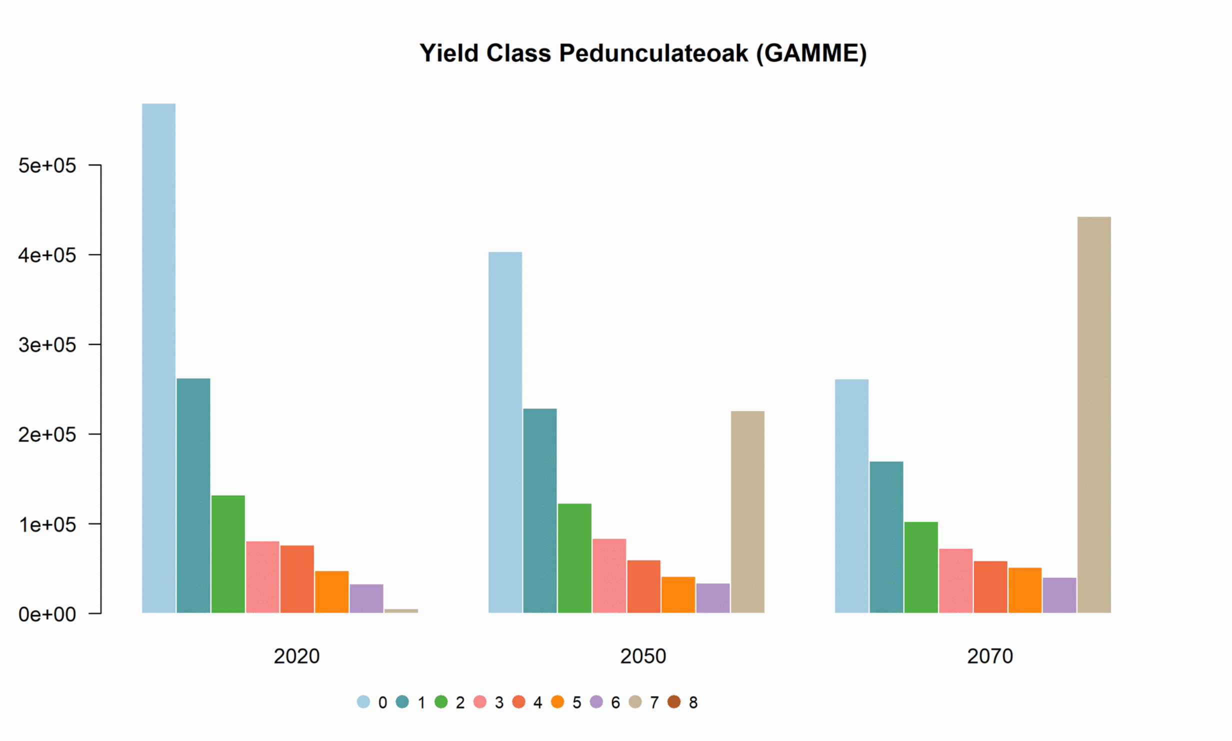

Downybirch_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
8
6
4
2
0
-2
-4

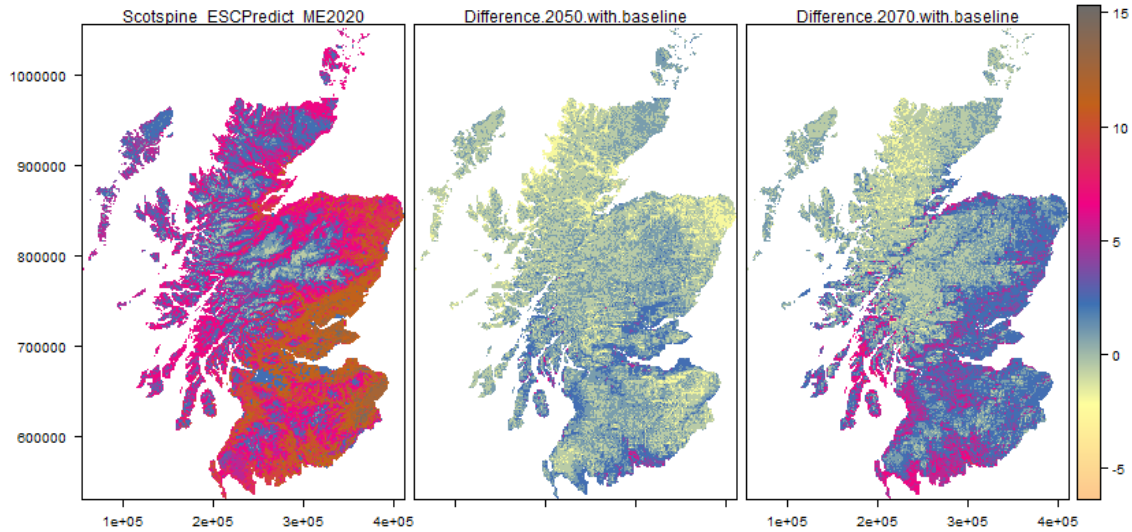

Yield Class Downybirch (GAMME)
250000
200000
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8

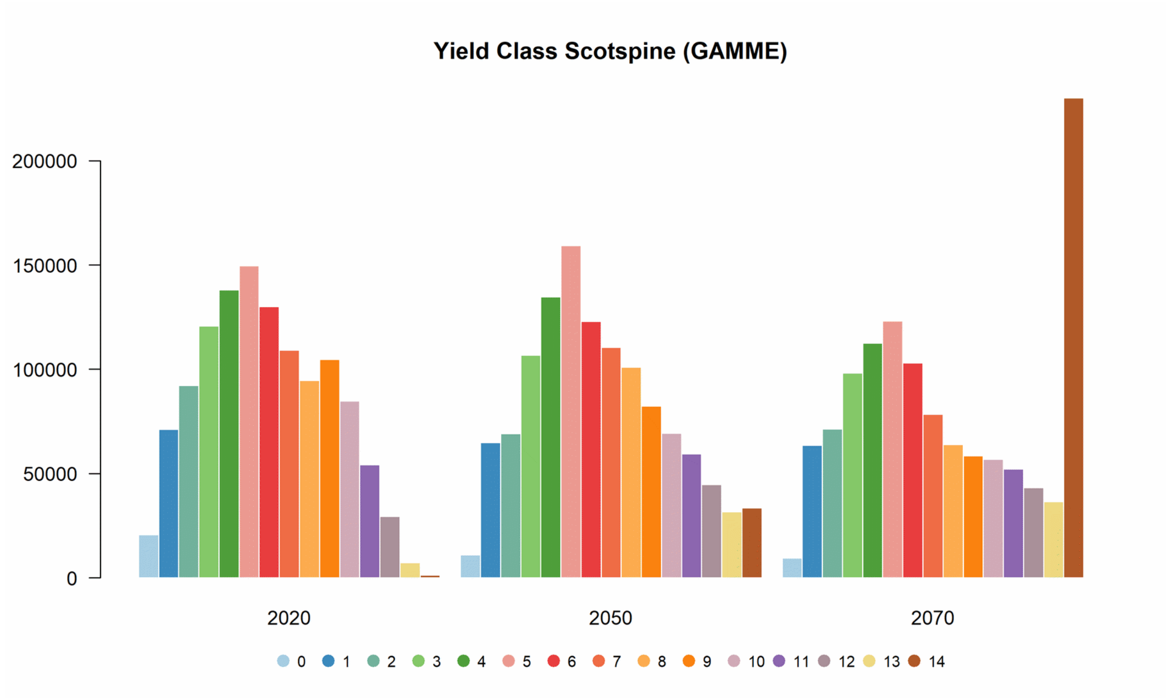

Lodgepolepine_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
15
10
5
0
-5

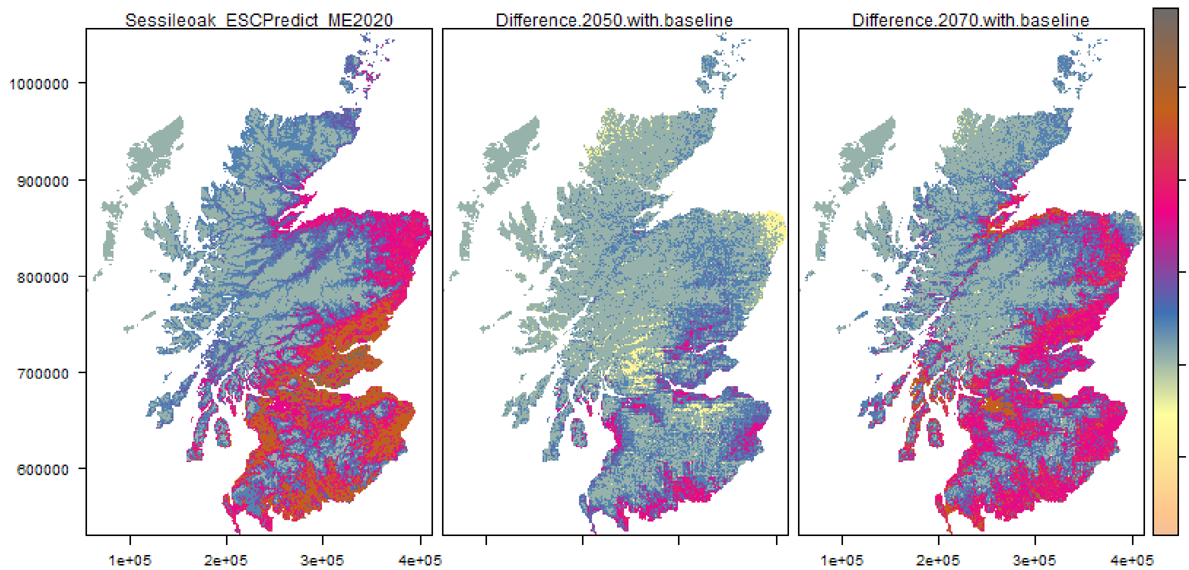

Yield Class Lodgepolepine (GAMME)
200000
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14

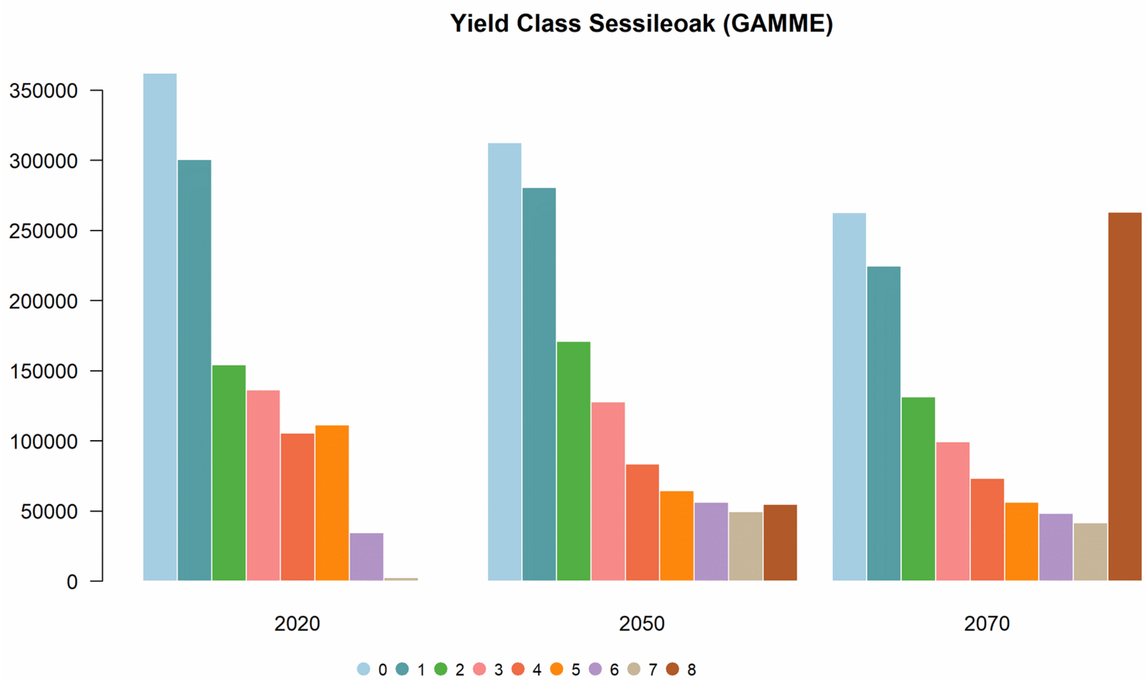

Pedunculateoak_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
8
6
4
2
0
-2

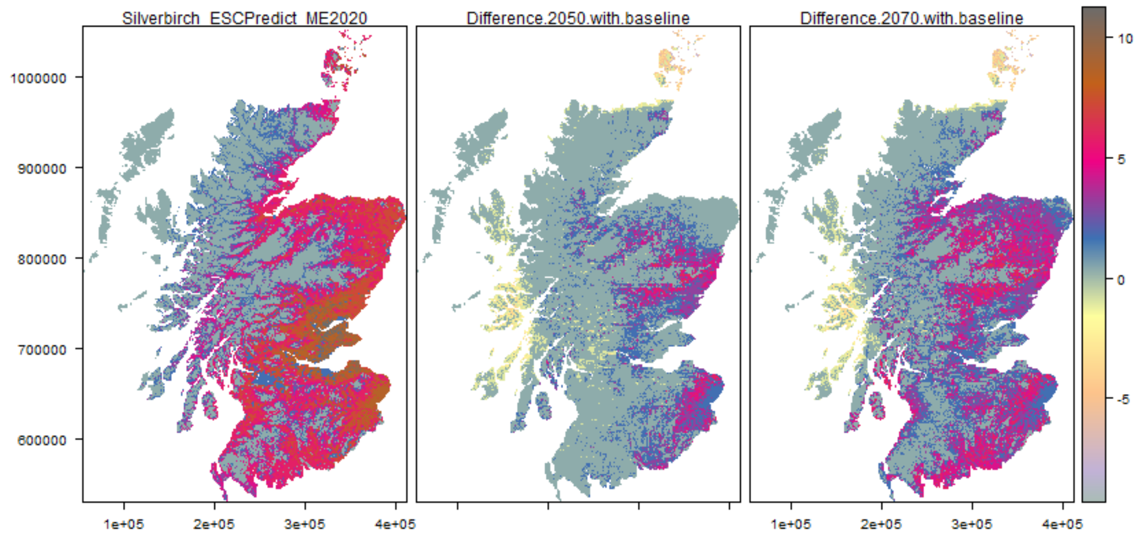

Yield Class Pedunculateoak (GAMME)
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7 8


Scotspine_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
15
10
5
0
-5

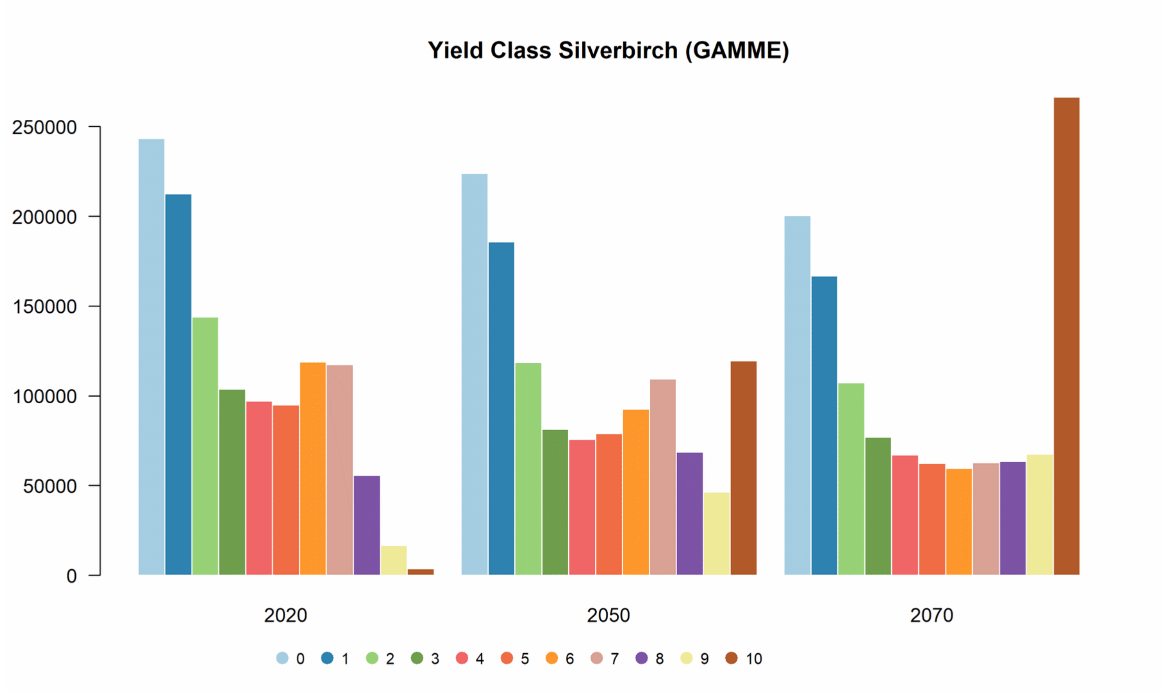

Yield Class Scotspine (GAMME)
200000
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14


Sessileoak_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
6
4
2
0
-2

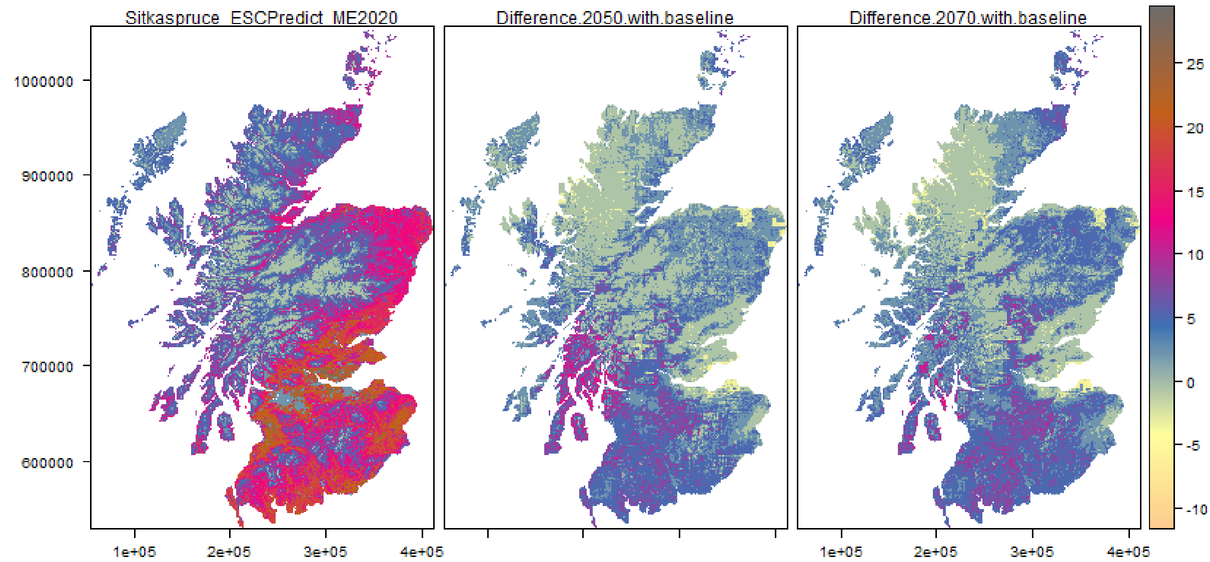

Yield Class Sessileoak (GAMME)
350000
300000
250000
200000
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8

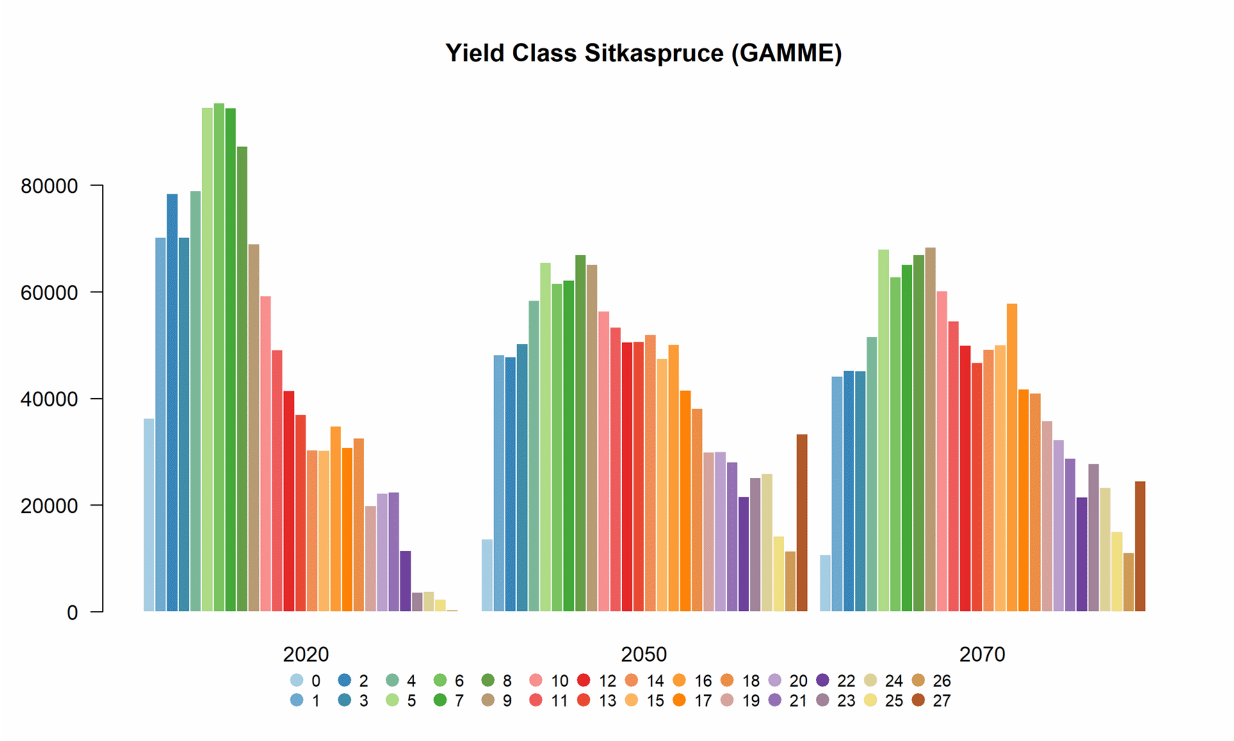


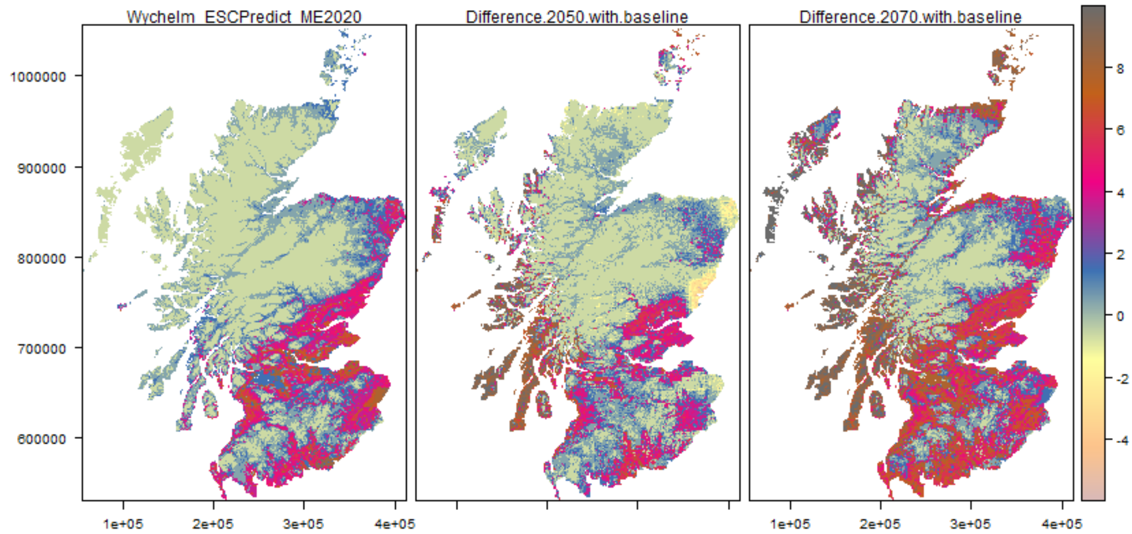
Silverbirch_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
10
5
0
-5

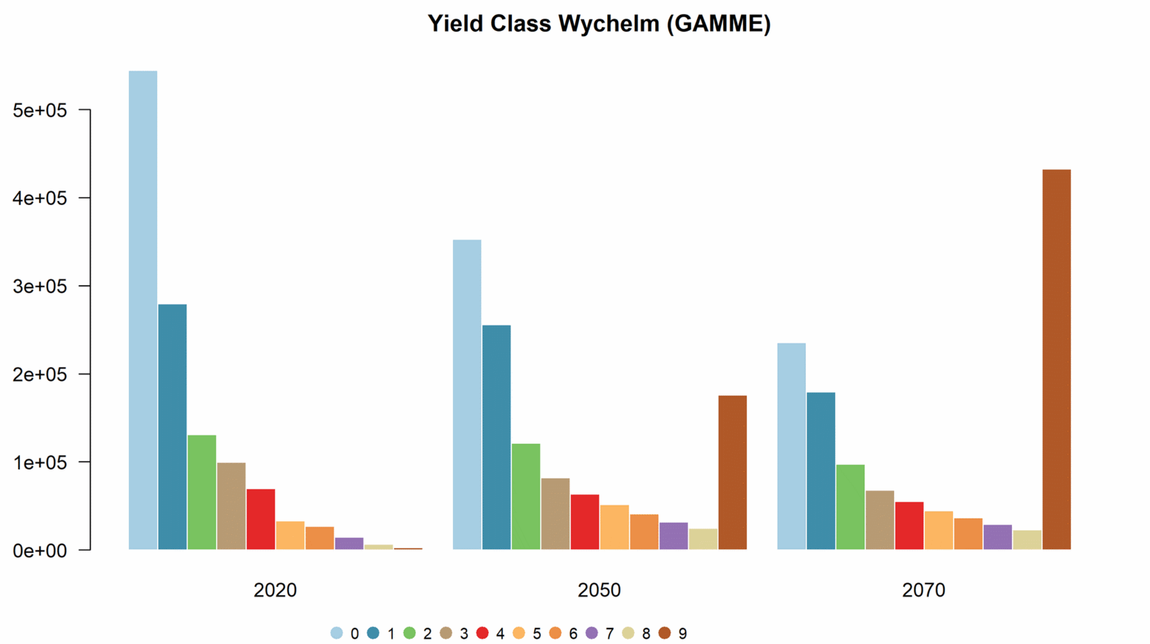


Yield Class Silverbirch (GAMME)
250000
200000
150000
100000
50000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10


Sitkaspruce_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
25
20
15
10
5
0
-5
-10

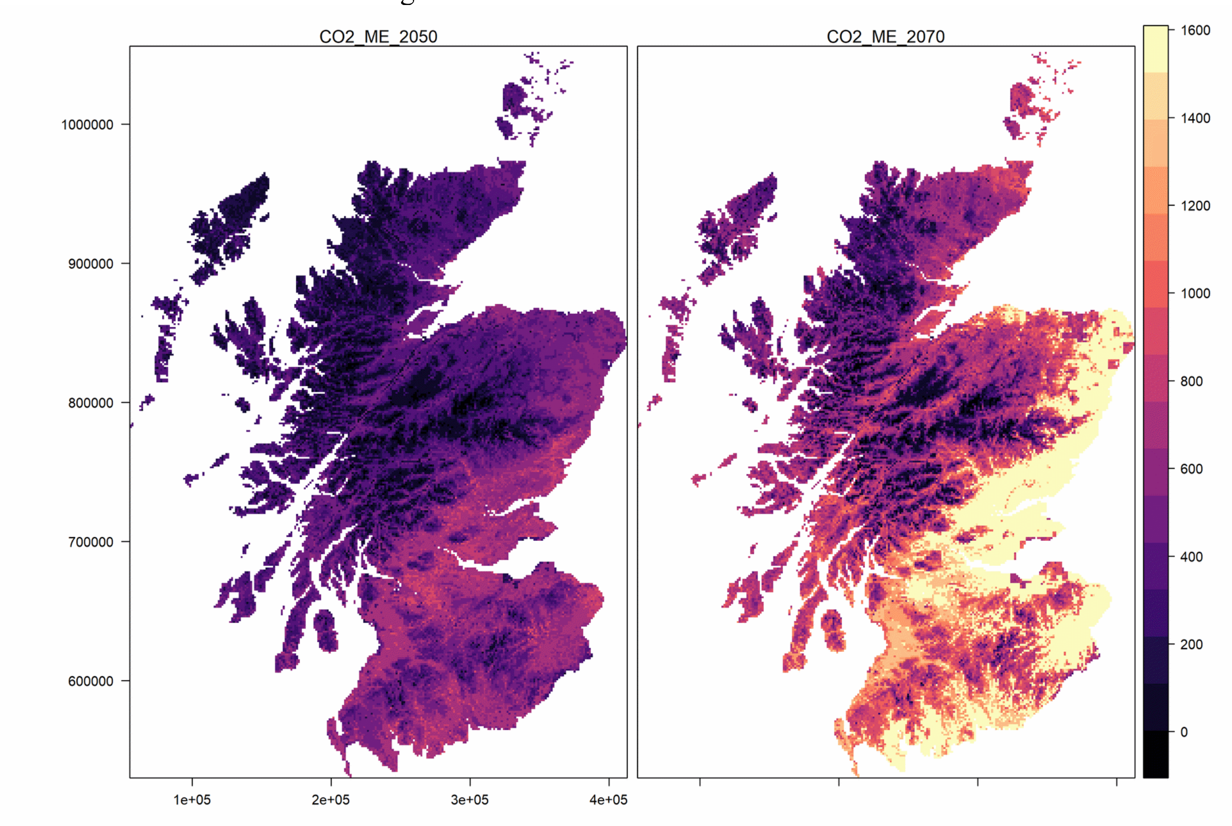

Yield Class Sitkaspruce (GAMME)
80000
60000
40000
20000
0
2020
2050
2070
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27

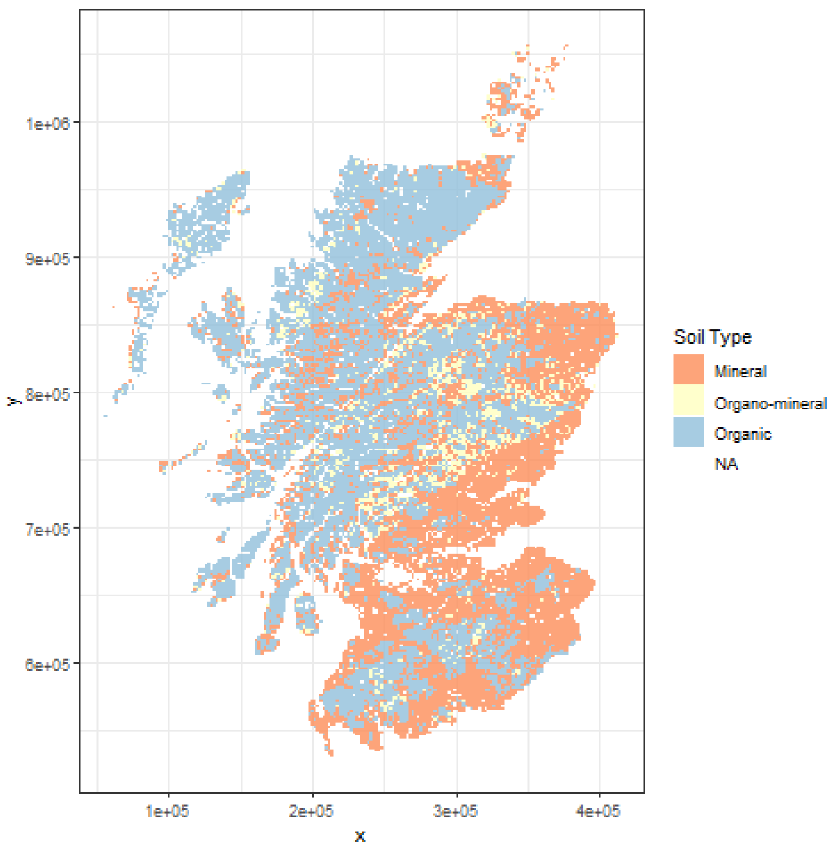

Wychelm_ESCPredict_ME2020
Difference.2050.with.baseline
Difference.2070.with.baseline
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
8
6
4
2
0
-2
-4


Yield Class Wychelm (GAMME)
5e+05
4e+05
3e+05
2e+05
1e+05
0e+00
2020
2050
2070
0 1 2 3 4 5 6 7 8 9

Appendix C

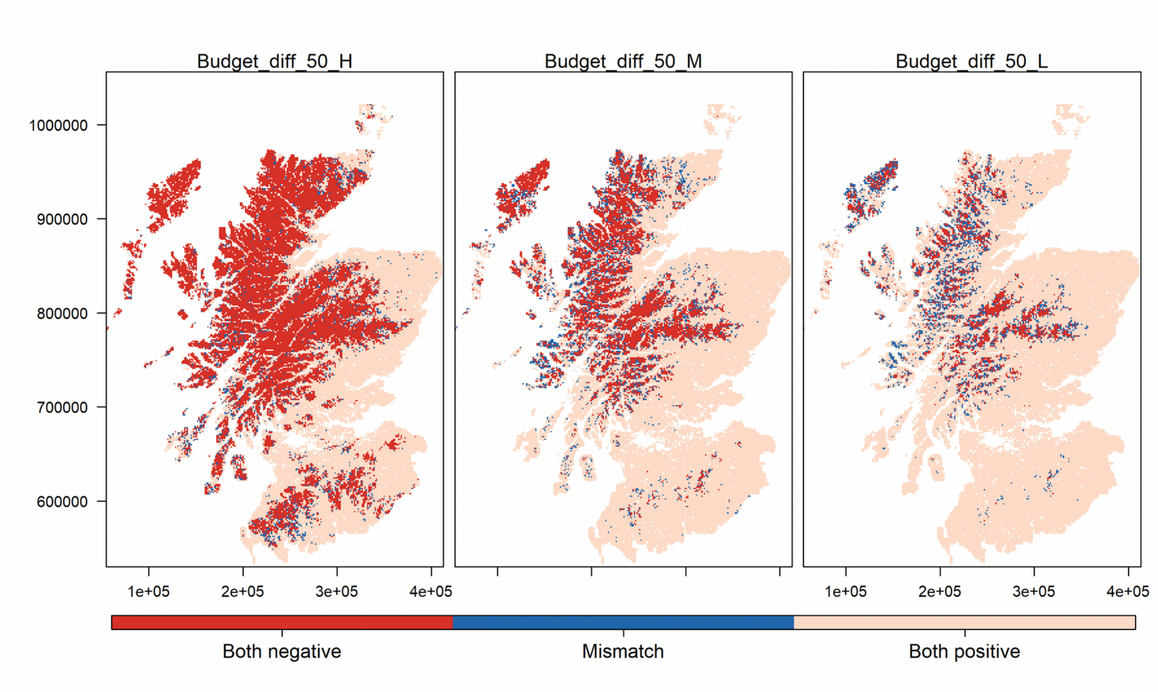
This appendix contains maps of the estimated biomass (tCO2-eq/ha) using the highest YC possible for 2050 and 2070 corrected with the growth curve method.

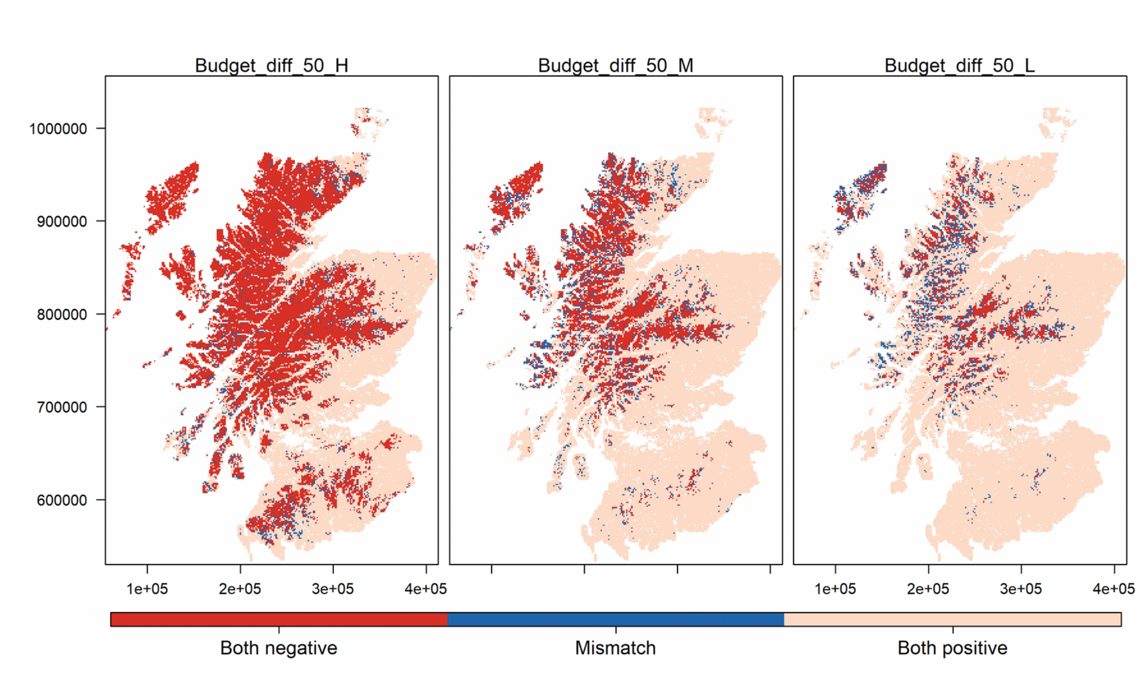




Appendix D

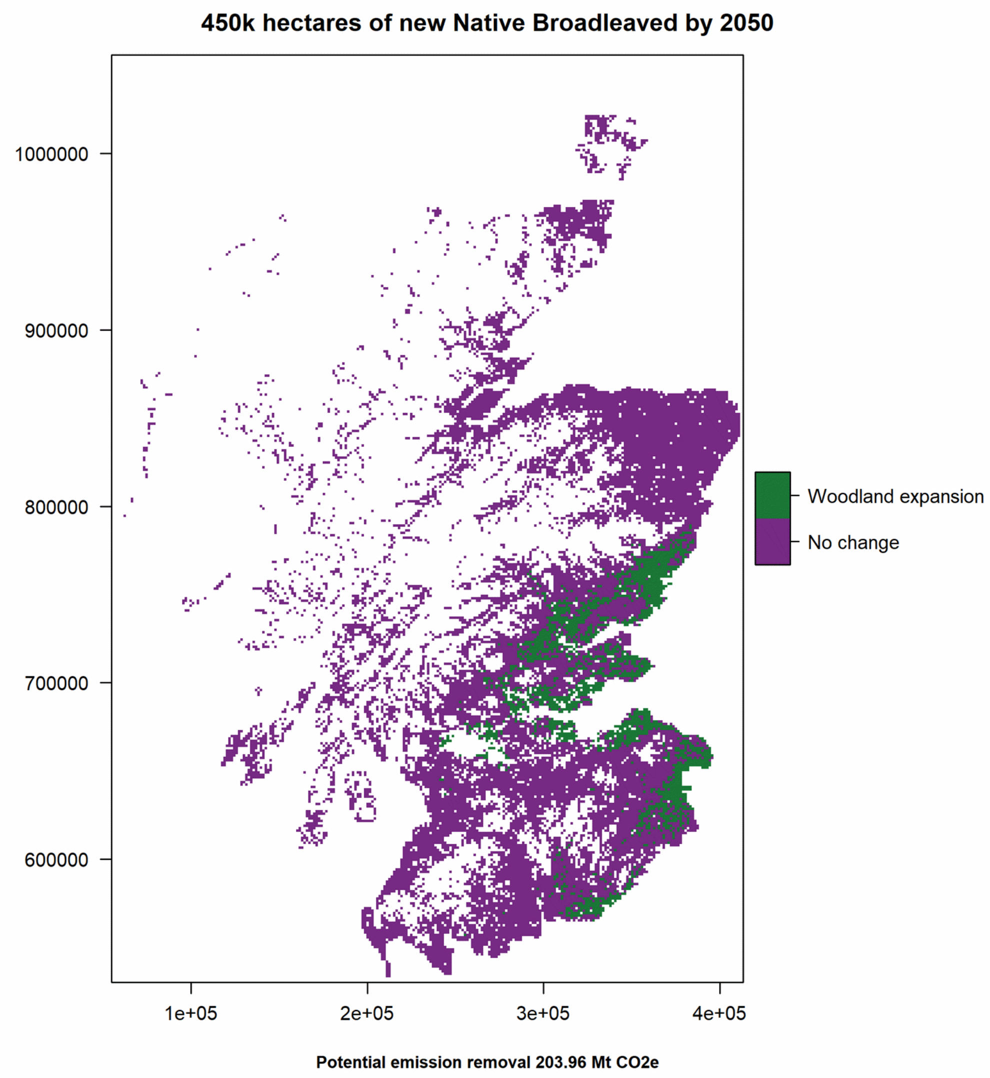
This appendix contains maps of the estimated carbon balance between soil and tree biomass for the two timesteps. The comparison illustrates the three soil disturbance levels where the two methods (lower-end and median) both predict to have a positive balance (gain), both predict a negative balance (loss) and where there is no coincidence in the direction of the balance (uncertainty).

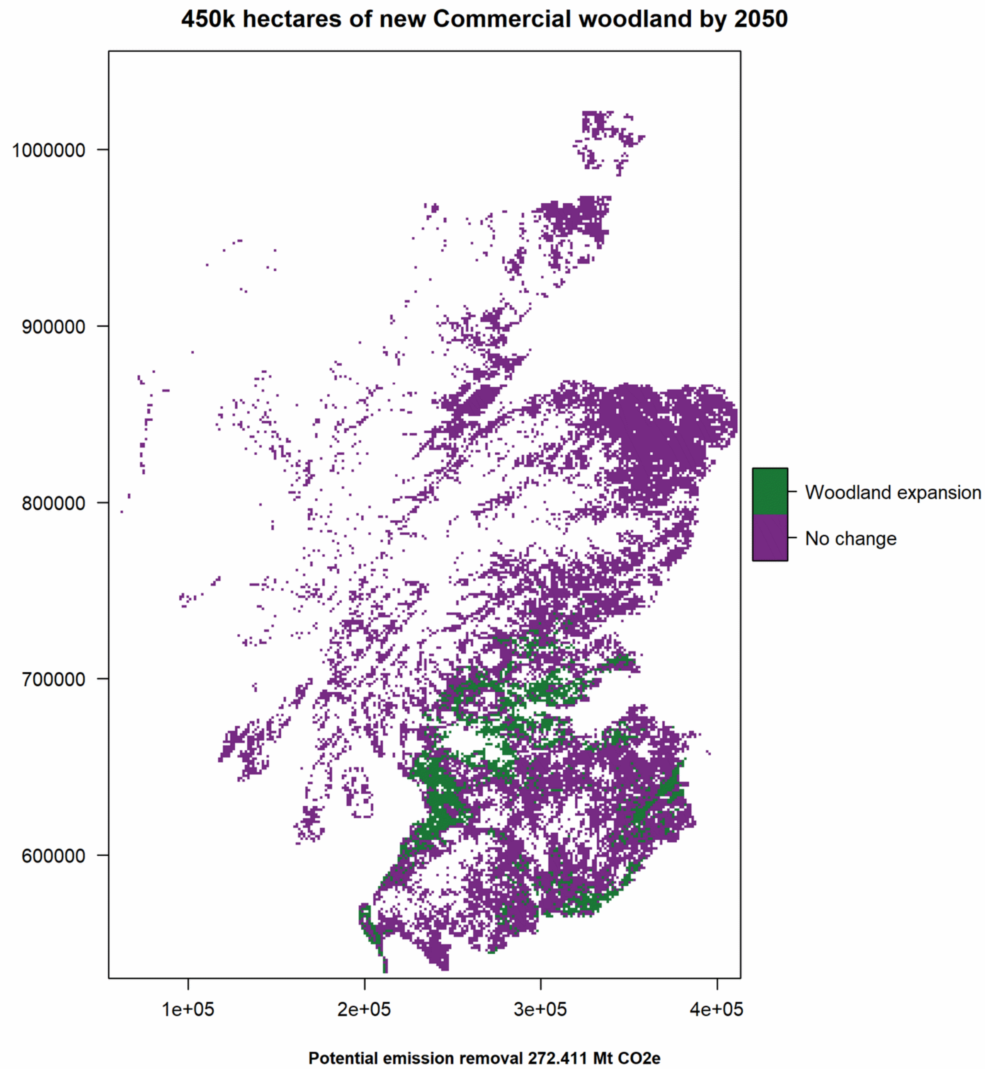




Appendix E

These maps summarised the hypothetical woodland expansion for 450k hectares in Scotland inserting commercial conifer (Sitka spruce) and native broadleaved (Ash and Birch) as indicated in *Tailwinds* CCC scenario. The scenario considered an expansion of the species with the highest YC for 2050 using locations ranked by positive carbon balance (gain) in order to maximize the potential for carbon stock. This also explains the different spatial patterns.




450k hectares of new Commercial woodland by 2050
1000000
900000
800000
700000
600000
1e+05
2e+05
3e+05
4e+05
Woodland expansion
No change
Potential emission removal 272.411 Mt CO2e

Appendix F

Timber growth and tree carbon sequestration models

Tree biomass carbon stock estimations are based on timber yield tables taken from the Forest Yield Model (FYM) (Matthews et al., 2016), considering all the yield classes available in the FYM tables for non-thinning or intermediate thinning management regimes and initial planting distance between 2 and 2.5 m. Timber volume growth or timber yield is estimated by forest species as nonlinear functions that depend on yield class (YC) and age (Table S1). It is to mention, that most yield tables do not account for trees older than 150 years (and only 80 years in case of Douglas fir), therefore timber volume prediction function are restricted to these age ranges.

Tree biomass in branchwood and roots is estimated applying an expansion factor (φ that converts standing timber stock (in cubic meters (m^3) ha^{-1} y^{-1}) to carbon stock (in t CO_2 m^{-3}):

$$\varphi = \rho \cdot \phi \cdot EF \cdot \kappa \qquad [1]$$

Where, ρ represents the density of timber assuming a relative humidity of 65%, which gives an equilibrium moisture content (MC) at 20°C of approximately 12% (Morison et al., 2012); ϕ represents the carbon content of oven dry biomass; EF the expansion factor that indicates the total volume of aboveground tree biomass and roots in relation to the standing timber stock, and κ the ratio of molecular weight to convert C to CO_2 (κ =3.667). The parameters ρ, ϕ and EF differ between tree species. In this paper we consider Morison et al., 2012 timber density estimates, a carbon content that varies between 0.42 and 1.46 (Milne and Brown (1997), and expansion factors from Levy et al. 2012. For Lodgepole pine we use the same conversion factors as Sitka spruce, while we use Birch conversion factor for Ash, all Birch species and common Alder (see Table S2).

Table S1 Timber volume nonlinear models and regression parameters by species

Function:	$v = K \cdot (t \cdot YC^2) \cdot e^{-at}$				$v = K \cdot (t \cdot YC^2) \cdot e^{-at^2}$
Variables	European larch	Beech & Wych elm	Scot pine	Douglas fir	SAB species§
K	0.6836 ***	0.6475 ***	0.6952 ***	0.7365 ***	0.5222 ***
	(0.0858)	(0.0445)	(0.0677)	(0.0769)	(0.0442)
b	1.0585 ***	1.0596 ***	1.0249 ***	1.0050 ***	1.1090 ***
	(0.0223)	(0.011)	(0.0154)	(0.0159)	(0.014)

a	-1.54E-03 **	-3.71E-04 **	9.72E-04 *	1.23E-04	-3.86E-05 ***
	(4.97E-04)	(1.42E-04)	(4.54E-04)	(3.41E-04)	(2.77E-06)
R^2	0.990	0.990	0.972	0.974	0.990
N obs.	320	600	601	720	400

Function: $$v = K \cdot (t \cdot YC^2) \cdot e^{-at}$$

Variables	Sitka spruce	All Oak species	Norway spruce	Lodgepole Pine	
K		0.0418 ***		0.710 ***	
		(0.0020)		(0.051)	
b	0.6607 ***	0.9696 ***	1.0585 ***	1.008 ***	
	(0.0007)	(0.0052)	(0.0223)	(0.012)	
a	7.35E-03 ***	1.12E-02 ***	-1.54E-03 ***	1.88E-03 ***	
	(8.21E-05)	(1.16E-04)	(4.97E-04)	(3.67E-04)	
R^2	0.966	0.999	0.990	0.992	
N obs.	1,800	450	320	480	

Notes: *p < 0.1, ** p < 0.01; *** p < 0.001. Robust standard error in parenthesis. § SAB species include Sycamore-Ash-Birch, and common Alder.

Source: Ovando (2020) for most of species except Lodgepole pine. These estimations are based on Forest Yield Model annual production tables for all yield classes available for each species type.

Table S2 Biomass to carbon conversion parameters for forest species in Great Britain

Species	Density of timber (ρ) in t m^{-3} (MC 12%[a])	Carbon content (ϕ) in tC t^{-1} of timber [b]	Timber to above ground biomass expansion factor (*EF*) [c]	Root ratio (Mroot/Mab oveground)[c]	Expansion factor from timber to total biomass (*EF*) [c]	Conversion from standing timber to carbon (φ) (tCO_2 m^{-3})
Beech*	0.689	0.46	2.226		2.226	2.587
Oak	0.689	0.46	2.226		2.226	2.587
Birch (SAB)	0.673	0.46	2.226		2.226	2.527
Douglas fir	0.497	0.42	2.230	0.260	2.490	1.906

Sitka spruce	0.384	0.42	2.230	0.410	2.640	1.561
Scot pine	0.513	0.42	2.230	0.300	2.530	1.999

Notes: *It is assumed that beech density is similar to oak.

Source: Ovando (2020) based on (a) Morison et al. (2012); (b) Milne and Brown (1997) and (c) Levy et al. (2012)

Carbon sequestration in woodland soil debris following tree planting is estimated on the basis of the Woodland Carbon Code (WCC) look up tables (West, 2018). Where the WCC data are not available for the yield classes predicted, YC is round up to the nearest even number of the WCC look up tables (Table S3).

Following the WCC procedures we only consider that only 80% (assuming a 20% WCC buffer) of tree and soil carbon sequestration as carbon that can be issued as carbon offsets. The carbon sequestration in soil debris already cred the 20% buffer.

Table S3. Estimated carbon sequestration/release in soil debris by species and yield class (in t CO_2/ha and year)

Max age	Oak			Birch			Beech	Scot pine					
	YC4	YC6	YC8	YC4	YC6	YC8	YC4	YC4	YC6	YC8	YC10	YC12	YC14
5	0.23	0.37	0.41	0.30	0.35	0.44	0.23	0.06	0.08	0.10	0.11	0.05	0.24
10	0.27	0.46	0.50	0.37	0.43	0.54	0.29	0.07	0.10	0.12	0.14	0.17	0.24
15	0.28	0.46	0.51	0.37	0.98	1.22	0.29	0.07	0.10	0.12	0.14	0.21	0.40
20	0.28	0.46	4.74	1.00	0.70	0.88	0.29	0.07	0.10	0.12	0.14	2.30	1.13
25	0.28	6.30	11.22	0.54	0.31	0.39	0.29	0.07	0.10	0.12	0.38	0.92	2.24
30	0.28	6.64	3.71	0.18	5.11	6.39	0.29	0.07	0.10	0.45	1.79	0.94	2.89
35	2.74	2.58	-1.78	2.41	7.03	8.79	0.84	0.07	0.25	1.30	0.20	2.82	2.62
40	2.74	-0.62	-1.35	5.09	1.81	2.26	1.40	0.07	0.40	0.08	0.83	2.37	1.11
45	0.75	-0.82	-0.18	1.83	-0.02	-0.03	0.55	1.70	0.71	0.79	2.82	0.95	0.44
50	0.53	-0.60	-0.86	-0.05	-0.94	-1.18	0.06	0.38	0.33	1.54	1.51	0.16	0.00
55	0.08	-1.46	-0.62	-0.80	-1.26	-1.58	-0.33	0.52	0.20	1.99	0.94	-0.28	-0.36
60	-0.34	-0.34	-1.43	-0.98	-1.34	-1.68	-1.02	0.35	1.28	0.56	0.08	-0.50	-0.46
65	-0.59	-0.11	-1.21	-0.76	-0.78	-0.97	-1.06	0.15	0.95	0.17	-0.26	-0.51	-0.48
70	-0.61	-0.16	-0.72	-0.85	-0.98	-1.22	-0.60	0.09	0.24	-0.20	-0.46	-0.34	-0.45
75	-0.39	0.11	-0.46	-0.85	-1.34	-1.68	-0.42	-0.07	0.20	-0.25	-0.43	-0.54	-0.47
80	-0.31	-1.17	-0.29	-0.85	-1.34	-1.68	-0.82	0.36	-0.42	-0.43	-0.52	-0.56	-0.51

Table S.3 continuation…

Max age	Douglas fir					Sitka spruce					Lodgepole pine		
	YC8	YC10	YC12	YC14	YC16	YC8	YC10	YC12	YC14	YC16	YC4	YC6	YC8
5	0.96	1.07	1.26	1.63	2.03	0.11	0.13	0.14	0.16	0.18	0.13	0.17	0.21
10	1.18	1.32	1.56	1.80	2.87	0.14	0.16	0.18	0.20	0.22	0.15	0.21	0.27
15	1.19	1.33	1.57	2.68	2.66	0.14	0.16	0.18	0.20	0.22	0.16	0.21	0.27
20	1.40	2.31	2.76	2.28	5.36	0.14	0.16	0.18	0.36	0.50	0.16	0.21	0.27
25	2.50	6.22	4.05	0.52	1.31	0.14	0.42	3.50	2.66	4.20	0.16	0.21	0.79
30	3.72	-0.08	0.58	3.40	1.74	2.21	1.41	0.73	0.54	0.46	0.16	0.76	1.366
35	-0.60	2.32	2.94	2.05	0.78	0.97	0.28	0.76	1.19	1.99	0.16	0.58	3.63
40	2.00	2.37	1.87	0.97	-0.65	0.92	0.74	0.49	0.86	0.65	0.84	2.93	1.66
45	1.59	0.99	0.57	0.06	-0.85	0.73	0.55	0.74	3.34	0.07	0.35	1.68	0.77
50	0.91	0.67	-0.05	-0.67	-0.96	0.34	-0.02	2.58	1.86	0.03	0.53	0.48	0.55
55	-0.04	-0.86	-0.53	-0.96	-1.19	-0.41	2.47	0.70	0.00	-0.90	1.84	0.07	-0.47
60	-0.32	-1.20	-1.27	-0.86	-1.02	-0.40	1.47	-0.30	-0.63	-0.88	0.87	-0.11	-0.5
65	-0.61	-1.49	-1.15	-0.66	-0.76	0.72	0.57	-0.61	-0.76	-0.70	0.35	-0.14	-0.35
70	-0.66	-0.96	-0.82	-0.52	-0.78	0.39	-0.35	-0.68	-0.88	-0.50	0.074	-0.112	-0.33
75	-0.70	-0.86	-0.58	-0.42	-0.56	0.00	-0.55	-0.68	-0.86	-0.44	-0.03	0.28	-0.05
80	-0.64	-0.60	-0.38	-0.43	-0.40	-0.17	-0.58	-0.59	-0.58	-0.37	0.58	-0.2	0.0

Source own elaboration based on Woodland carbon Code look up tables (West 2018).

References

Levy, P.E., Hale, S.E., Nicoll, B.C., 2012. Biomass expansion factors and root : shoot ratios for coniferous tree species in Great Britain. *Forestry* 77, 421–430.

Matthews, R.W., Jenkins, T.A.R., Mackie, E.D., Dick, E.C., 2016. *Forest Yield. A handbook on forest growth and yield tables for British forestry*. Forestry Commision, Edinburgh.

Milne, R., Brown, T.A., 1997. Carbon in the Vegetation and Soils of Great Britain. *J. Environ. Manage.* 49, 413–433.

Morison, J., Matthews, R., Miller, G., Perks, M., Randle, T., Vanguelova, E., White, M., Yamulki, S., 2012. *Understanding the carbon and greenhouse gas balance of forests in Britain*. Forest Research.

Ovando (2020) *Application of the Natural Capital Protocol at Glensaugh farm*. The James Hutton Institute. (available https://www.hutton.ac.uk/research/projects/integrating-natural-capital-thinking-land-based-decision-making)

West, V., 2018. Lookup Table Guidance Using the WCC Carbon Calculation Spreadsheet Version 2.0 1–21.

Lookup table to convert YC to Carbon stock in vegetation (includes above and below ground biomass) (t CO2/ha)

Species	**CodeSP**	**Yield class**	**Code**	Central scenario (average carbon sequestration in t CO2/ha) in tree biomass															
				5	10	15	20	25	30	35	40	45	50	55	60	65	70	75	80
Oak (all species)	**1**	1	11	1.85	6.72	13.94	23.03	33.55	45.17	57.58	70.52	83.77	97.14	110.47	123.63	136.49	148.97	160.98	172.47
Oak (all species)	**2**	2	22	3.63	13.16	27.31	45.10	65.71	88.46	112.76	138.10	164.05	190.23	216.34	242.10	267.29	291.72	315.25	337.74
Oak (all species)	**3**	3	33	5.38	19.50	40.46	66.81	97.36	131.07	167.07	204.61	243.06	281.85	320.53	358.70	396.02	432.22	467.08	500.40
Oak (all species)	**4**	4	44	7.11	25.77	53.47	88.31	128.68	173.24	220.82	270.44	321.26	372.53	423.65	474.10	523.43	571.28	617.35	661.39
Oak (all species)	**5**	5	55	8.83	31.99	66.39	109.64	159.76	215.08	274.16	335.77	398.85	462.51	525.98	588.61	649.86	709.27	766.46	821.15
Oak (all species)	**6**	6	66	10.53	38.18	79.23	130.84	190.66	256.67	327.17	400.70	475.98	551.95	627.69	702.43	775.52	846.42	914.67	979.94
Oak (all species)	**7**	7	77	12.23	44.33	92.00	151.93	221.39	298.05	379.92	465.29	552.71	640.93	728.88	815.67	900.54	982.87	1062.13	1137.91
Oak (all species)	**8**	8	88	13.92	50.46	104.72	172.94	252.00	339.25	432.43	529.61	629.11	729.53	829.63	928.42	1025.02	1118.73	1208.95	1295.20
SAB (all species)	**3**	1	31	6.28	13.52	21.09	28.82	36.59	44.32	51.92	59.35	66.53	73.41	79.96	86.13	91.88	97.19	102.02	106.36
SAB (all species)	**3**	2	32	13.56	29.15	45.49	62.16	78.93	95.59	112.00	128.01	143.50	158.36	172.48	185.78	198.19	209.63	220.06	229.42
SAB (all species)	**3**	3	33	21.25	45.71	71.31	97.45	123.74	149.87	175.59	200.69	224.98	248.27	270.41	291.27	310.72	328.66	345.01	359.68
SAB (all species)	**3**	4	34	29.24	62.88	98.12	134.08	170.24	206.19	241.58	276.12	309.53	341.57	372.04	400.74	427.50	452.18	474.66	494.86
SAB (all species)	**3**	5	35	37.45	80.54	125.66	171.73	218.04	264.09	309.42	353.65	396.44	437.48	476.50	513.26	547.53	579.14	607.94	633.80
SAB (all species)	**3**	6	36	45.84	98.59	153.82	210.21	266.90	323.27	378.75	432.90	485.28	535.52	583.28	628.27	670.23	708.93	744.18	775.84
SAB (all species)	**3**	7	37	54.39	116.97	182.50	249.40	316.67	383.54	449.37	513.61	575.75	635.36	692.03	745.41	795.18	841.10	882.92	920.48
SAB (all species)	**3**	8	38	63.07	135.64	211.63	289.21	367.21	444.75	521.10	595.59	667.65	736.77	802.49	864.39	922.11	975.35	1023.85	1067.41
SAB (all species)	**3**	9	39	71.87	154.57	241.17	329.57	418.45	506.82	593.81	678.70	760.82	839.58	914.47	985.00	1050.78	1111.45	1166.72	1216.35
SAB (all species)	**3**	10	310	80.77	173.73	271.06	370.41	470.32	569.63	667.41	762.82	855.12	943.65	1027.81	1107.09	1181.02	1249.21	1311.33	1367.12
SAB (all species)	**3**	11	311	89.78	193.10	301.28	411.71	522.75	633.14	741.82	847.87	950.46	1048.85	1142.40	1230.52	1312.69	1388.49	1457.53	1519.54
SAB (all species)	**3**	12	312	98.88	212.66	331.80	453.42	575.71	697.28	816.97	933.76	1046.74	1155.11	1258.14	1355.18	1445.68	1529.15	1605.19	1673.47
Scot pine	**4**	1	41	5.76	11.66	17.58	23.49	29.38	35.25	41.08	46.88	52.64	58.36	64.03	69.67	75.25	80.80	86.30	91.75

Scot pine	**4**	2	42	11.71	23.72	35.77	47.80	59.79	71.73	83.59	95.39	107.11	118.74	130.29	141.75	153.13	164.41	175.60	186.70
Scot pine	**4**	3	43	17.75	35.94	54.19	72.43	90.59	108.68	126.66	144.53	162.29	179.92	197.42	214.78	232.01	249.11	266.07	282.88
Scot pine	**4**	4	44	23.84	48.27	72.78	97.26	121.66	145.94	170.09	194.09	217.94	241.61	265.11	288.43	311.57	334.53	357.30	379.88
Scot pine	**4**	5	45	29.96	60.67	91.48	122.25	152.92	183.45	213.80	243.97	273.93	303.69	333.23	362.55	391.63	420.49	449.11	477.50
Scot pine	**4**	6	46	36.12	73.13	110.27	147.37	184.34	221.13	257.73	294.09	330.21	366.09	401.69	437.03	472.09	506.88	541.38	575.60
Scot pine	**4**	7	47	42.30	85.65	129.15	172.59	215.89	258.98	301.83	344.42	386.73	428.74	470.44	511.83	552.89	593.63	634.04	674.11
Scot pine	**4**	8	48	48.50	98.21	148.09	197.90	247.55	296.96	346.10	394.94	443.45	491.62	539.43	586.89	633.98	680.69	727.02	772.97
Scot pine	**4**	9	49	54.72	110.81	167.09	223.29	279.31	335.06	390.51	445.61	500.34	554.69	608.64	662.19	715.32	768.02	820.30	872.14
Scot pine	**4**	10	410	60.96	123.45	186.14	248.75	311.16	373.27	435.03	496.42	557.39	617.94	678.04	737.69	796.88	855.59	913.83	971.59
Scot pine	**4**	11	411	67.22	136.11	205.24	274.28	343.09	411.57	479.67	547.35	614.59	681.35	747.62	813.39	878.65	943.39	1007.60	1071.28
Scot pine	**4**	12	412	73.49	148.81	224.38	299.86	375.09	449.96	524.41	598.41	671.91	744.90	817.35	889.25	960.60	1031.38	1101.58	1171.20
Scot pine	**4**	13	413	79.77	161.53	243.56	325.50	407.15	488.42	569.24	649.56	729.35	808.58	887.22	965.28	1042.72	1119.55	1195.76	1271.33
Scot pine	**4**	14	414	86.07	174.28	262.78	351.18	439.28	526.96	614.16	700.82	786.90	872.38	957.23	1041.45	1125.00	1207.89	1290.11	1371.65
Sitka spruce	**6**	1	61	10.10	24.33	40.07	56.49	73.13	89.69	105.99	121.88	137.27	152.09	166.28	179.81	192.66	204.81	216.26	227.02
Sitka spruce	**6**	2	62	15.96	38.46	63.35	89.31	115.61	141.80	167.56	192.69	217.01	240.43	262.86	284.25	304.57	323.78	341.89	358.89
Sitka spruce	**6**	3	63	20.87	50.28	82.81	116.75	151.13	185.36	219.04	251.88	283.69	314.30	343.62	371.58	398.14	423.26	446.93	469.16
Sitka spruce	**6**	4	64	25.24	60.80	100.15	141.19	182.77	224.17	264.90	304.62	343.08	380.10	415.56	449.38	481.49	511.87	540.49	567.37
Sitka spruce	**6**	5	65	29.25	70.46	116.06	163.62	211.80	259.78	306.98	353.01	397.58	440.48	481.58	520.77	557.98	593.18	626.36	657.51
Sitka spruce	**6**	6	66	32.99	79.48	130.92	184.56	238.92	293.04	346.28	398.20	448.48	496.87	543.23	587.44	629.41	669.12	706.55	741.69
Sitka spruce	**6**	7	67	36.53	88.00	144.96	204.35	264.53	324.46	383.41	440.90	496.56	550.15	601.48	650.42	696.90	740.87	782.31	821.21
Sitka spruce	**6**	8	68	39.90	96.12	158.33	223.20	288.93	354.38	418.78	481.57	542.37	600.90	656.96	710.42	761.18	809.21	854.46	896.96
Sitka spruce	**6**	9	69	43.13	103.90	171.14	241.26	312.32	383.07	452.67	520.54	586.26	649.53	710.13	767.91	822.79	874.70	923.62	969.55
Sitka spruce	**6**	10	610	46.24	111.39	183.48	258.66	334.84	410.68	485.30	558.07	628.53	696.36	761.32	823.28	882.11	937.76	990.21	1039.45
Sitka spruce	**6**	11	611	49.24	118.63	195.40	275.47	356.60	437.38	516.85	594.34	669.38	741.62	810.81	876.79	939.44	998.71	1054.57	1107.02
Sitka spruce	**6**	12	612	52.16	125.65	206.97	291.77	377.70	463.26	547.44	629.52	708.99	785.51	858.79	928.67	995.04	1057.82	1116.98	1172.53

Sitka spruce	**6**	13	613	54.99	132.47	218.21	307.62	398.22	488.42	577.17	663.71	747.50	828.17	905.43	979.11	1049.08	1115.27	1177.64	1236.21
Sitka spruce	**6**	14	614	57.75	139.12	229.16	323.06	418.20	512.93	606.13	697.01	785.01	869.73	950.87	1028.25	1101.73	1171.24	1236.74	1298.25
Sitka spruce	**6**	15	615	60.44	145.61	239.85	338.13	437.71	536.86	634.40	729.52	821.63	910.29	995.22	1076.21	1153.11	1225.86	1294.43	1358.80
Sitka spruce	**6**	16	616	63.08	151.96	250.30	352.86	456.77	560.24	662.04	761.31	857.42	949.95	1038.58	1123.09	1203.35	1279.27	1350.82	1418.00
Sitka spruce	**6**	17	617	65.65	158.17	260.53	367.28	475.44	583.14	689.10	792.42	892.47	988.78	1081.03	1168.99	1252.53	1331.55	1406.03	1475.95
Sitka spruce	**6**	18	618	68.18	164.25	270.55	381.41	493.74	605.59	715.62	822.92	926.82	1026.83	1122.63	1213.99	1300.74	1382.80	1460.14	1532.76
Sitka spruce	**6**	19	619	70.66	170.23	280.39	395.29	511.70	627.61	741.65	852.85	960.52	1064.18	1163.46	1258.14	1348.05	1433.10	1513.25	1588.50
Sitka spruce	**6**	20	620	73.10	176.10	290.06	408.91	529.34	649.25	767.22	882.25	993.64	1100.86	1203.57	1301.51	1394.52	1482.50	1565.41	1643.26
Sitka spruce	**6**	21	621	75.49	181.86	299.56	422.31	546.68	670.52	792.35	911.15	1026.19	1136.93	1243.00	1344.15	1440.21	1531.07	1616.70	1697.10
Sitka spruce	**6**	22	622	77.85	187.54	308.91	435.49	563.75	691.45	817.08	939.60	1058.22	1172.42	1281.81	1386.11	1485.16	1578.86	1667.17	1750.08
Sitka spruce	**6**	23	623	80.17	193.13	318.12	448.47	580.55	712.06	841.44	967.60	1089.76	1207.37	1320.01	1427.43	1529.43	1625.92	1716.86	1802.24
Sitka spruce	**6**	24	624	82.46	198.64	327.19	461.26	597.11	732.37	865.44	995.20	1120.84	1241.80	1357.66	1468.14	1573.05	1672.29	1765.82	1853.64
Sitka spruce	**6**	25	625	84.71	204.07	336.14	473.87	613.43	752.39	889.10	1022.41	1151.49	1275.75	1394.78	1508.27	1616.06	1718.01	1814.10	1904.32
Sitka spruce	**6**	26	626	86.93	209.43	344.96	486.32	629.54	772.14	912.44	1049.25	1181.72	1309.24	1431.39	1547.87	1658.48	1763.12	1861.73	1954.31
Sitka spruce	**6**	27	627	89.13	214.72	353.68	498.59	645.43	791.64	935.48	1075.74	1211.56	1342.30	1467.54	1586.96	1700.36	1807.64	1908.74	2003.66
Douglas fir	**5**	1	51	5.66	11.35	17.04	22.74	28.44	34.14	39.84	45.53	51.22	56.91	62.59	68.27	73.94	79.61	85.27	90.93
Douglas fir	**5**	2	52	11.35	22.77	34.20	45.64	57.08	68.52	79.95	91.38	102.80	114.22	125.62	137.02	148.40	159.78	171.15	182.50
Douglas fir	**5**	3	53	17.06	34.23	51.41	68.61	85.80	102.99	120.18	137.35	154.52	171.67	188.81	205.94	223.06	240.16	257.24	274.31
Douglas fir	**5**	4	54	22.78	45.70	68.65	91.61	114.57	137.52	160.47	183.40	206.32	229.23	252.12	274.99	297.84	320.67	343.49	366.28
Douglas fir	**5**	5	55	28.51	57.19	85.91	114.64	143.37	172.10	200.81	229.51	258.19	286.86	315.50	344.12	372.72	401.29	429.84	458.36
Douglas fir	**5**	6	56	34.25	68.69	103.18	137.69	172.20	206.70	241.19	275.67	310.12	344.54	378.95	413.32	447.67	481.99	516.28	550.54
Douglas fir	**5**	7	57	39.99	80.20	120.47	160.76	201.06	241.34	281.61	321.86	362.08	402.28	442.45	482.59	522.69	562.76	602.80	642.80
Douglas fir	**5**	8	58	45.73	91.72	137.78	183.85	229.93	276.00	322.06	368.09	414.09	460.06	506.00	551.90	597.76	643.59	689.37	735.12
Douglas fir	**5**	9	59	51.47	103.25	155.09	206.96	258.83	310.69	362.53	414.34	466.12	517.87	569.58	621.25	672.88	724.47	776.01	827.50

Douglas fir	**5**	10	510	57.22	114.78	172.41	230.08	287.74	345.39	403.02	460.62	518.19	575.72	633.21	690.65	748.04	805.39	862.69	919.93
Douglas fir	**5**	11	511	62.98	126.32	189.75	253.20	316.67	380.12	443.54	506.93	570.28	633.60	696.86	760.08	823.24	886.35	949.41	1012.41
Douglas fir	**5**	12	512	68.73	137.86	207.09	276.34	345.61	414.85	484.07	553.26	622.40	691.50	760.54	829.54	898.47	967.35	1036.17	1104.93
Douglas fir	**5**	13	513	74.49	149.41	224.43	299.49	374.56	449.60	524.62	599.60	674.54	749.42	824.25	899.03	973.74	1048.39	1122.97	1197.49
Douglas fir	**5**	14	514	80.25	160.96	241.79	322.65	403.52	484.37	565.19	645.97	726.70	807.37	887.99	968.54	1049.03	1129.46	1209.81	1290.09
Douglas fir	**5**	15	515	86.01	172.52	259.15	345.82	432.49	519.15	605.77	692.35	778.87	865.34	951.75	1038.09	1124.36	1210.55	1296.67	1382.72
Douglas fir	**5**	16	516	91.78	184.08	276.51	368.99	461.48	553.94	646.36	738.74	831.07	923.33	1015.53	1107.65	1199.70	1291.67	1383.57	1475.38
Douglas fir	**5**	17	517	97.54	195.65	293.89	392.17	490.47	588.74	686.97	785.16	883.28	981.34	1079.33	1177.24	1275.07	1372.82	1470.49	1568.06
Douglas fir	**5**	18	518	103.31	207.21	311.26	415.36	519.47	623.55	727.59	831.58	935.51	1039.36	1143.15	1246.85	1350.46	1453.99	1557.43	1660.78
Douglas fir	**5**	19	519	109.08	218.78	328.64	438.56	548.48	658.37	768.22	878.02	987.75	1097.41	1206.98	1316.47	1425.88	1535.19	1644.41	1753.52
Douglas fir	**5**	20	520	114.85	230.36	346.03	461.76	577.49	693.20	808.86	924.47	1040.00	1155.46	1270.84	1386.12	1501.31	1616.41	1731.40	1846.29
Douglas fir	**5**	21	521	120.62	241.94	363.42	484.97	606.52	728.04	849.52	970.93	1092.27	1213.53	1334.71	1455.78	1576.76	1697.64	1818.42	1939.08
Douglas fir	**5**	22	522	126.39	253.52	380.82	508.18	635.55	762.89	890.18	1017.40	1144.55	1271.62	1398.59	1525.46	1652.24	1778.90	1905.45	2031.89
Douglas fir	**5**	23	523	132.17	265.10	398.22	531.40	664.58	797.74	930.85	1063.89	1196.85	1329.72	1462.49	1595.16	1727.72	1860.18	1992.51	2124.73
Douglas fir	**5**	24	524	137.95	276.68	415.62	554.62	693.63	832.60	971.53	1110.38	1249.15	1387.83	1526.40	1664.87	1803.23	1941.47	2079.59	2217.58
Douglas fir	**5**	25	525	143.72	288.27	433.03	577.85	722.68	867.47	1012.22	1156.88	1301.47	1445.95	1590.33	1734.60	1878.75	2022.78	2166.68	2310.46
Douglas fir	**5**	26	526	149.50	299.86	450.43	601.08	751.73	902.35	1052.91	1203.40	1353.79	1504.08	1654.27	1804.34	1954.28	2104.10	2253.79	2403.35
Lodgepole pine	**9**	1	91	1.45	5.10	10.27	16.48	23.33	30.54	37.84	45.06	52.04	58.69	64.90	70.63	75.83	80.49	84.60	88.15
Lodgepole pine	**9**	2	92	2.83	9.96	20.08	32.22	45.62	59.70	73.99	88.10	101.76	114.74	126.89	138.10	148.27	157.38	165.41	172.35
Lodgepole pine	**9**	3	93	4.19	14.75	29.72	47.69	67.54	88.38	109.52	130.42	150.63	169.85	187.84	204.42	219.49	232.97	244.85	255.12
Lodgepole pine	**9**	4	94	5.54	19.48	39.26	62.99	89.21	116.74	144.67	172.26	198.97	224.36	248.11	270.01	289.91	307.72	323.41	336.99
Lodgepole pine	**9**	5	95	6.88	24.17	48.72	78.17	110.70	144.86	179.52	213.77	246.91	278.41	307.89	335.07	359.76	381.86	401.33	418.18
Lodgepole pine	**9**	6	96	8.20	28.84	58.11	93.24	132.05	172.81	214.15	255.00	294.53	332.11	367.28	399.70	429.15	455.52	478.74	498.83
Lodgepole pine	**9**	7	97	9.52	33.47	67.46	108.24	153.29	200.60	248.58	296.00	341.89	385.52	426.34	463.97	498.16	528.77	555.73	579.05

Lodgepole pine	**9**	8	98	10.83	38.09	76.76	123.16	174.42	228.26	282.86	336.82	389.03	438.67	485.12	527.95	566.85	601.68	632.36	658.90
Lodgepole pine	**9**	9	99	12.14	42.69	86.02	138.03	195.47	255.80	317.00	377.47	435.98	491.62	543.67	591.66	635.26	674.29	708.67	738.42
Lodgepole pine	**9**	10	910	13.44	47.27	95.25	152.84	216.44	283.25	351.01	417.97	482.76	544.36	602.00	655.14	703.43	746.64	784.71	817.65
Lodgepole pine	**9**	11	911	14.74	51.83	104.45	167.60	237.35	310.61	384.91	458.34	529.39	596.94	660.15	718.42	771.37	818.76	860.50	896.62
Lodgepole pine	**9**	12	912	16.04	56.38	113.63	182.32	258.19	337.88	418.71	498.59	575.88	649.36	718.12	781.51	839.10	890.66	936.07	975.35
Lodgepole pine	**9**	13	913	17.33	60.92	122.77	196.99	278.98	365.08	452.42	538.73	622.24	701.64	775.93	844.43	906.66	962.36	1011.43	1053.88
Lodgepole pine	**9**	14	914	18.62	65.45	131.90	211.63	299.71	392.22	486.04	578.76	668.49	753.79	833.60	907.18	974.04	1033.88	1086.60	1132.20

Appendix G
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Technical report: Maps of land use data and ecosystem services for catchment level case studies in Scotland: examples applied to the National Parks and Aberdeenshire River Dee

RESAS1.4.3 Objective D [Deliverable: D3]: Mapping ESS and benefits to illustrate adaptive and integrated catchment management

Authors: and Richard Hewitt*, Kit (C.J.A.) Macleod, Andrea Baggio Compagnucci, Marie Castellazzi, David G. Miller and Alessandro Gimona, James Hutton Institute, Aberdeen, UK.

*Corresponding author: richard.hewitt@hutton.ac.uk

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Contents

Executive summary

The purpose of this technical report is to provide an update on work in the Strategic Research Programme (Work package 1.4.3) that is collating maps/spatial datasets on land use and ecosystem services for case study locations of Scotland's two National Parks and Aberdeenshire River Dee. These maps have been produced through work on agricultural land use and its impacts (e.g. Hewitt et al 2018) and on mapping ecosystem service indicators (e.g. Gimona et al 2018), and are intended for analysis at the whole river catchment scale or smaller.

This report provides the bridge between this earlier work of data compilation and integration in year one and scheduled tasks for 2019, as follows:

1 Rapid analysis of broad tendencies of land use and land cover change at Scotland scale. (Deliverable 4a)

2 Catchment-scale analysis of recent land change developments in forestry and agriculture, e.g. for selected catchments or comparable-scale case study areas of interest to key stakeholders. (Deliverable 4b)

We provide a brief structured description of the maps/spatial datasets presented in this report: comprising general description, methods used to produce them, their recommended use, and their principal limitations. The intention is that these descriptions should serve as a guide to stakeholders, interested in using them to aid management of Scotland's natural assets. All the datasets presented in this report have been developed for all of Scotland. However, three study areas have been selected to illustrate these datasets at the landscape scale.

These data form a useful basis for spatial analysis to mitigate negative land use and ecosystems service impacts, and providing support to land managers and other policy stakeholders. However, the datasets described do have some limitations, which we summarize below. It should be noted that all datasets have errors, and detailed description of the limitations provided in this report should not be taken as an indicator of poor quality relative to other sources. Rather, good practice requires that limitations should be properly described and documented.

With respect to the integrated land use datasets described in the first part of the report, the main limitations relate to the use of mixed data from multiple sources or data not originally intended for that purpose. The IACS data are not ideally suited for use as a land use time series, since land parcels record only the use claimed under the agricultural payments system, so that cessation of claims results in the disappearance of a land use from one date to the next in a way that does not reflect land use in reality. In addition, they provide a poor record of land use outside of the most important agricultural areas, due to the lesser importance of these areas in the payments system.

With respect to the ecosystem services dataset described in the second part of the report, main limitations relate to their reliance on published, rather than directly measured data, together with insufficient resources available for acquisition of proprietary, third party data, e.g. on forest biomass, British Trust for Ornithology bird atlas data, Land Cover Map etc.

We provide recommendations relating to future development of these integrated datasets.

1. Introduction

This technical report is part of the Strategic Research Programme (SRP) on land use change and Ecosystem Services funded by the Scottish Government. The Natural Assets Theme of the Scottish Government's Strategic Research Programme 2016-21 (hereafter Theme) is concerned with identification, quantification and valuation of Scotland's environmental assets, biodiversity and ecosystem services. Modelling and mapping of land use change and key indicators of ecosystem services is an essential component of this Theme. Mapped indicators could support decision-making across land use policy priorities (such as a low carbon economy, sustainable food production and water management) by allowing spatially explicit visioning of the land use change and ecosystem services trade-offs. For example, they can highlight areas in which landscapes provide multiple services and benefits, which could be protected if necessary, and areas where intervention through a variety of policy instruments could be needed.

Modelling and mapping of land use change and indicators of ecosystem services requires adequate spatial data on land and its resources. Though land use, agriculture and forestry data are developed and maintained by a wide range of scientific and public bodies, including Scottish Government, the Centre for Ecology and Hydrology (CEH) and the Forestry Commission (FC) data are not always obtained in a form that is directly appropriate for the relevant analyses. For this reason, significant resources have been allocated under the Natural Assets Theme for the systematization, harmonisation and integration of large-scale spatial land use datasets from a range of sources.

The purpose of this technical report is to provide an update on research in Work Package 1.4.3 that is collating maps/spatial datasets on land use and ecosystem services for case study locations of Scotland's two National Parks and Aberdeenshire River Dee. These maps have been produced through work on agricultural land use and its impacts (e.g. Hewitt et al 2018) and on mapping ecosystem service indicators (e.g. Gimona et al 2018) and are intended for analysis at the whole river catchment scale or smaller.

This report therefore provides the bridge between this earlier work of data compilation and integration in year one and scheduled tasks for 2019, as follows:

3 Rapid analysis of broad tendencies of land use and land cover change at Scotland scale. (Deliverable 4a)

4 Catchment-scale analysis of recent land change developments in forestry and agriculture, e.g. for selected catchments or comparable-scale case study areas of interest to key stakeholders. (Deliverable 4b)

2. Description of datasets produced

The purpose of these sections is to provide a brief structured description of the maps/spatial datasets presented in this report: comprising of a general description, methods used to produce them, their recommended use, and their principal limitations. The intention is that these descriptions should serve as a guide to available data to inform evidence-based policy on management of Scotland's Natural Assets. All the datasets presented in this report have been developed for all of Scotland. However, three study areas have been selected to illustrate these datasets at the landscape scale. These case study areas were chosen as areas of key interest to stakeholders following national and regional level stakeholder engagement in year one. These areas are shown in Figure 1. Maps referred to throughout the text are provided for these three study areas in Appendices to this report.

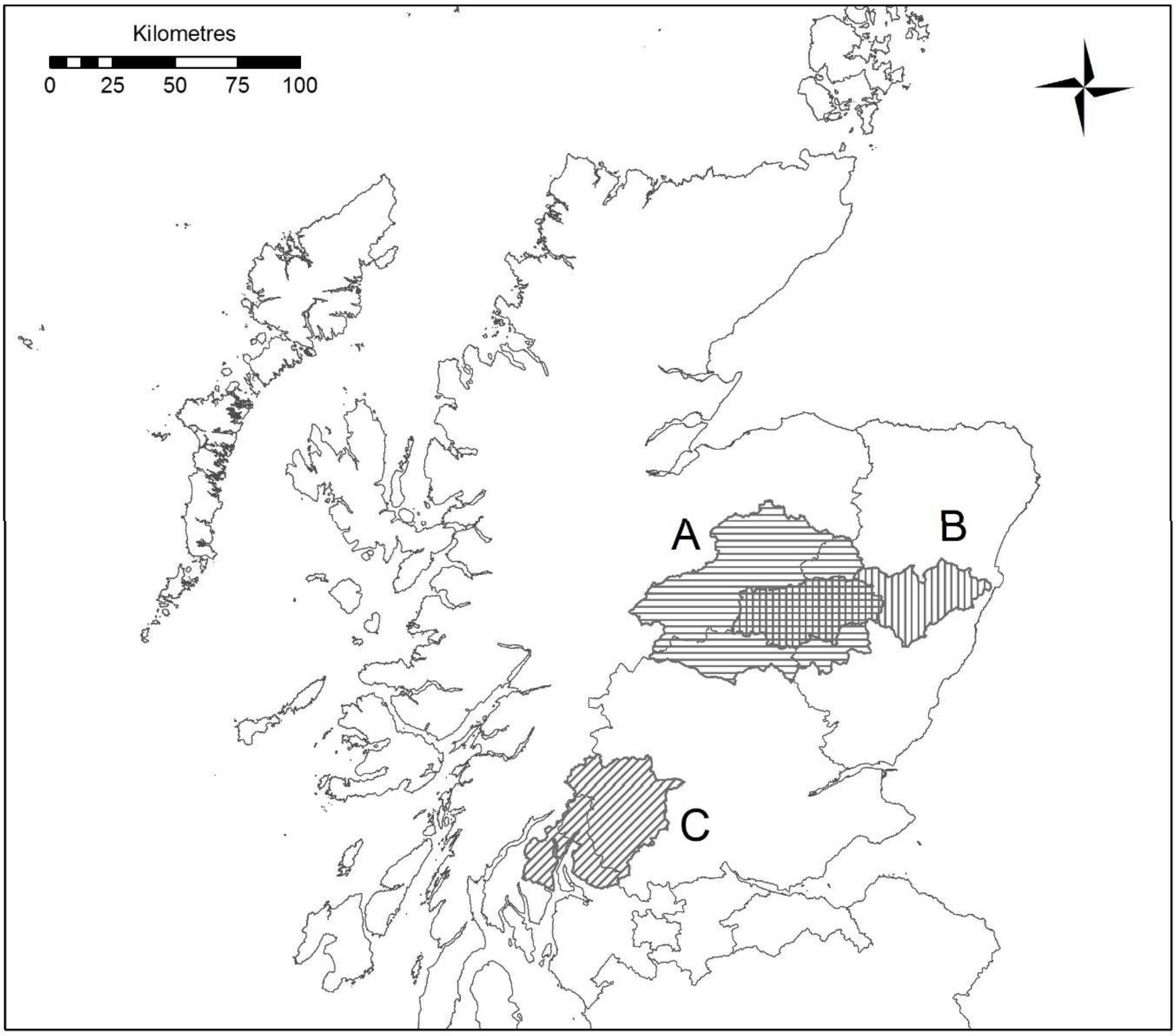

Figure 1: Mainland Scotland and three study areas chosen as exemplars for display of spatial datasets. A: Cairngorms National Park; B: Grampian River Dee; C: Loch Lomond and Trossachs National Park.

2.1 Integrated spatial land use datasets

2.1.1 Land use datasets and rationale for 2019 work programme

Since earth surface cover is a key factor in controlling erosion, water supply and climate, ecosystems and the services they provide are highly vulnerable to land use and land cover (LUC) change (Metzger et al. 2006, Turner et al. 2007). LUC monitoring therefore plays a vital role in understanding these change processes and analysing, reporting, and managing their impacts on the ecosystems that are necessary for human survival. The development of accurate, large scale LUC datasets is an essential pre-requisite for this work.

In order to assess the potential of existing land use and land cover datasets to respond to this necessity, a rapid survey of available datasets was carried out (Table 1). The survey (Table 1) shows that there is a lack of large-scale (high spatial resolution) data on land use or land cover data for a series of consecutive historical dates for Scotland. Without such a resource, the scope, nature, and extent of analysis on land change and ecosystem services is seriously constrained. For instance, while both CORINE land cover and Agcensus[1] allow a national scale understanding of agricultural land change, and are useful for highlighting areas of concern, the information is not detailed enough to be able to identify individual crop types, or fully evaluate the impacts of land use change on ecosystem services. At the same time, Forestry and Woodland Inventories are available for a range of dates at a highly detailed scale but lack accompanying data for other land use types. This makes it difficult to obtain a good understanding of the evolution of forestry in relation to other types of land use.

Table 1: Available data on land use/land cover for Scotland

Dataset	Creator	Dates available	Key limitations	URL
The Land Cover of Scotland 1988 (LCS88)	Macaulay land research institute	1988	Only one date available.	https://www.hutton.ac.uk/learning/exploringscotland/landcover-scotland-1988
Land Cover Map (LCM) series	Centre for Ecology and Hydrology (CEH)	1990,2000, 2007, 2015	Not freely available (must be purchased). JHI do not have latest map (2015), there are no plans to obtain it due to its high cost. Not recommended for comparison of different map dates due to different classification criteria at each date.	https://www.ceh.ac.uk/services/land-cover-map-2007
Forestry commission forestry surveys (various), e.g. Native Woodlands Survey for Scotland 2014, National Forestry Inventory for Scotland 2015, National Forest Estate Legal Boundary for Scotland 2016.	Forestry Commission	2010-15, earlier dates also may be available, e.g. National inventory of woodland and trees	Forest/Woodland land uses only.	https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/

		(1995-99)		
Coordination of Information on the Environment (CORINE)	European Environment Agency	2000, 2006,2012	Small scale (1: 100,000 max). Classification not well adapted to local land cover types.	https://land.copernicus.eu/pan-

[1] http://agcensus.edina.ac.uk/

	and local partners			european/corine-land-cover
Agricultural census data for Scotland (agcensus)	Scottish Executive: SEERAD and (from 2007) Environment Directorate.	Annually from 1969	Coarse grained (1km max) Aggregated to parish scale, Agricultural land only.	http://agcensus.edina.ac.uk/
Habitat Map of Scotland (HabMoS)	Scottish Natural Heritage (SNH)	Nominal date of 2015	Land cover information (EUNIS Land Cover Scotland) is an amalgamation of many existing sources, e.g. LCS88, LCM 2000, LCM 2007, National Forest Inventory etc. Multiple dates in a single map not useful for change monitoring, and likely to be very unreliable for this purpose.	http://gateway.snh.gov.uk/natural-spaces/dataset.jsp?dsid=HABMOS https://www.spatialdata.gov.scot/geonetwork/srv/eng/catalog.search#/metadata/08d85469-bc12-4e67-819e-b41ae47b0392

To respond to these limitations in the baseline datasets available (Table 1), a range of new spatial datasets were created by Hutton staff, either by combining information from different sources (Land Cover Map (LCM), Forest Inventory), or by using information from other sources (e.g. Integrated Administration and Control System (IACS) dataset) to create new spatial datasets. These are listed in Table 2 and are described in the following sections. We describe these data as “integrated spatial datasets” because their creation involves a process of systematic unification (=integration) of the information in each dataset. Clearly, such a process makes the output dataset more useful for the required objectives (analysis of catchment level natural assets), but also introduces some limitations. These are described in detail in the following sections.

Table 2. List of integrated spatial land use datasets

No.	Name	Scale/ resolution	Time periods available	Accessible	Description/sources	Type	Filename	Format	Created by / contact
1	**IACS predominant land use 2008-15**	From 1:5000 (lowlands) to 1:50000 (uplands)	2008-15	Restricted access, contact creator	IACS surveyed land parcels with area claimed under CAP payments system, with predominant land uses assigned according to the simple classification (see documentation).	Land use information (spatial)	f[year]PREDOM.shp (e.g. f10PREDOM.shp)	ESRI Shape file	Richard Hewitt Richard.hewitt@hutton.ac.uk
2	**IACS predominant land use, extended crops classification**	From 1:5000 (lowlands) to 1:50000 (uplands). Minimum mapped unit c. 0.2ha	2010, 2015 (can create any other date between 2008 and 2015 as required)	Restricted access, contact creator	IACS surveyed land parcels with area claimed under CAP payments system, with predominant land uses assigned according to the extended classification (see documentation).	Land use information (spatial)	f10PREDOM_detailed.shp, f15PREDOM_detailed.shp	ESRI Shape file	Richard Hewitt

3	**LCM2007 integrated with Forestry Commission woodland inventory data (LCM2007w2 and LCM2007w3)**	LCM states minimum mappable unit 0.5ha, though some woodland parcels may be smaller	2007 with 2015 woodland	Restricted access, contact creator	LCM2007 (produced by CEH), merged with Native Woodlands Survey for Scotland 2014, National Forestry Inventory for Scotland 2015, National Forest Estate Legal Boundary for Scotland 2016.	Land use information (spatial)	LCM07v6s2_010217.gdb, LCM07v6s2_WoodR2id_raster_25m, LCM07v6s2_Wood	ESRI Shape file, 25m raster	Marie Castellazzi Marie.castellazzi@hutton.ac.uk

					2 reclassifications: WoodR3id: all as LCM INTCODE classes; WoodR2id : LCM INTCODES + 8 woodland subcategories.		R3id_raster_25m		
4	**IACS_LCM07w_raster**	As LCM2007	2007/2010, 2007/2015	Restricted access, contact creator	IACS and LCM2007w3 merged using the ArcGIS MOSAIC tool, giving overlay priority to IACS.	Land use information (spatial)	Integrated IACS simple classification (see 1, above) and LCM 2007 with forestry dataset (see 3 above)	25m raster	Richard Hewitt
5	**IACSextended_LCM07w_raster**	As LCM2007	2007/2010, 2007/2015	Restricted access, contact creator	IACS and LCM2007w3 merged using the ArcGIS MOSAIC tool, giving overlay priority to IACS.	Land use information (spatial)	Integrated IACS extended classification (see 2, above) and LCM 2007	25m raster	Richard Hewitt

								with forestry dataset (see 3 above)		

2.1.2 IACS predominant land use 2008-15

2.1.2.1 General description

The Integrated Administration and Control System (IACS) dataset, available under restricted licensing conditions due to the sensitivity of the data, contains information on land use at the level of the land parcel from land use declarations made by land managers as part of the requirement to receive payments under either Pillar 1 or Pillar 2 of the Common Agricultural Policy (CAP). Maps are shown in Appendix 1.1.

2.1.2.2 Methods

By joining land claims information (hectares of crop claimed in each parcel) to the corresponding parcels in the spatial database using the Field_ID, it was possible to obtain a highly detailed map of agricultural land uses claimed for each year since the first spatial database became available in 2001 until the most recent complete dataset available (2015). However, development of the spatial database was incremental, with improvements made continuously every year, and full, high quality coverage was not achieved until ca. 2008. To obtain a series of snapshots or “time slices” suitable for the study of the spatial evolution of land claims over this period, maps were generated for 3 dates, 2008, 2010 and 2015. The maps were generated by summing the total land claims per parcel and automatically assigning the predominant claim to the whole land parcel. Since the claims database contains over 100 crop types, in the first instance data were aggregated into 10 simple classes (Appendix 3.1).

The claims assigned to these 10 classes were checked by summing the total hectare amounts for the new aggregate classes and comparing with the field "TOTAL_AREA". Thus 2008 and 2009 totals were found not to match the total in the "TOTAL_AREA" column, since some of the parcels had land classified as Land Let Out" (LLO), representing a land use unknown to or undeclared by the claimant. Thus by adding the LLO amounts in each case, the totals have been corrected. Thus all totals listed are correct and checked. LLO appears only in years 2008 and 2009, and in 2008 and 2009 classes for Water, Inland_Rock and Urban are empty, as all area quantities were documented in the Unclassified category. As of 2010, these data appear in their correct classes and Unclassified is empty.

2.1.2.3 Recommended use

The recommended use of this dataset is to provide large-scale (i.e. detailed) spatial information on basic agricultural land use in Scotland and its evolution over time; this could be used to show more detailed estimates of ecosystem services and multiple benefits from Scotland’s agricultural land.

2.1.2.4 Principal limitations

In terms of the usefulness of IACS as a land use dataset, there are two key limitations that need to be taken into account. These are: 1) Errors in GIS mapping quality; 2) assigning a single land use to parcels containing multiple uses; 3) the source of the land use information. These are briefly discussed as follows:

2.1.2.4.1 Errors in GIS mapping quality.

The accuracy of the spatial data is dependent on the quality of the original IACS dataset, which contained significant errors. These errors have gradually been corrected, and from 2010 onwards datasets no longer contain significant errors. Detailed technical description of these errors is given in Appendix 4.1.

2.1.2.4.2 Assigning a single land use to parcels containing multiple uses.

Another potential source of error relates to the fact that land parcels often contain more than one land use. However, since the spatial distribution of multiple land uses within each parcel was unknown, it was necessary to choose the predominant land use in order to make the time series maps for the three snapshot years (2008, 2010, 2015). This is described in more detail in Appendix 4.2.

2.1.2.4.3 The source of the land use information

One further key limitation with this dataset relates to the origin of the information used to classify land use at the scale of an individual land parcel, which are claims submitted to the Rural Payments and Services division of Scottish government under Pillar 1 of the CAP. The presence of land use on the maps is therefore an indicator of land use, rather than an objective measurement, such as would be obtained by classification of remotely sensed data or orthophotographic mapping. For instance, the appearance of many new woodland areas between 2008 and 2010 is not an indicator of woodland growth, but rather, it reflects the full incorporation of woodland payments data into the claims database after 2008. Conversely, if a claim for a particular land use class made in one year is discontinued in subsequent years, it disappears from the map. For this reason, IACS is a rather unreliable source for year-on-year monitoring. The problems are likely to be most severe for non-agricultural land use classes. Since established agricultural land is likely to remain eligible across dates under various cropping regimes, these areas are likely to be more reliable.

2.1.3 IACS predominant land use, extended crops classification

2.1.3.1 General description

The simple crops classification adopted from the IACS simple groupings described above, e.g. arable, temporary grassland, permanent grassland etc are too broad for many types of analysis, for example, to understand differential nutrient export using the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model (Sharp et al 2014). For this reason a different grouping of the IACS individual land use codes was undertaken for this specific purpose. Maps are shown in Appendix 1.2.

2.1.3.2 Method

This dataset was created from the IACS land claims database, in the same way as for the previous dataset (see 2.1.1.1). As for the preceding dataset, the maps were generated by summing the total land claims per parcel and automatically assigning the predominant claim to the whole land parcel. To provide more detail on crop type than in the previous dataset, land claims data were aggregated into 17 classes (Appendix 3.2).

2.1.3.3 Recommended use

This dataset provides large scale spatial information on agricultural land use in Scotland and its evolution over time, allowing for more detailed mapping of ecosystem services. The crop categories chosen are of particular interest for understanding nutrient retention/export.

2.1.3.4 Principal limitations

Since these data are derived from the same source as 2.1.1, the same issues noted above are also applicable to this dataset.

2.1.4 LCM2007 integrated with Forestry Commission woodland inventory data

2.1.4.1 General description

These data (LCM07v6s2_WoodR2n3.shp, or .gdb) were created in January 2017 by M. Castellazzi, with the aim of improving the representation of woodland in the LCM2007 land cover map (Appendix 1.3a, Appendix 1.3b). The integrated dataset incorporated the latest version (as of January 2017) of 3 Forestry Commission datasets:

- Native Woodlands Survey for Scotland 2014 (NWSS),
- National Forestry Inventory for Scotland 2015 (NFIS),
- National Forest Estate Legal Boundary for Scotland 2016.

Maps are shown in Appendix 1.3.

2.1.4.2 Method

Two reclassifications were carried out: WoodR2 & WoodR3; both combines in order of priority: native woodlands from NWSS + non-native woodlands from NFIS + LCM07 classes.
In WoodR2, woodlands are subdivided as broadleaved, coniferous, woodland (unspecified type) and clear fell (includes Failed and Windthrow categories). Note that shrubs, scrubs and most PAWS (Planted Ancient Woodland Sites), are not included in this reclassification.
In WoodR3, all woodlands are kept in only 2 categories to fit with the original LCM07 'INTCODE' attributes: broadleaved and coniferous. Integration of the datasets was carried out in GIS software. To limit the occurrence of small artefact polygons (slivers) when overlaying the datasets, a 10m tolerance was used.

2.1.4.3 Recommended use

The resulting aggregate map is used to all intents and purposes as a replacement for the standard LCM2007 spatial dataset for analyses of ecosystem service provision that are dependent on land use inputs.

2.1.4.4 Principal limitations

The use of a 10m tolerance when integrating the data has introduced small spatial discrepancies (<0.2% of the landscape when comparing NFIS15 woodland areas between the original dataset and the aggregated dataset). The combination of the land use classes from the different input datasets was designed for ecosystem services (ESS) models (e.g. InVEST), which needed to identify mature woodlands. Further reclassification rules could be implemented to fit requirements of other studies, e.g. taking into account shrubs or young trees.

2.1.5 IACS predominant land use integrated with LCM 2007 woodland dataset

2.1.5.1 General description

Given the limitations of the IACS data for non-agricultural land cover classes, a combined dataset was created in which agricultural land classes (arable, temporary grassland, improved grassland) from IACS simple classification (Section 2.1.1) were combined with LCM07w3 (extended LCM07 with Woodland inventory 2015 classification no.3) (Section 2.1.3). This was carried out for two IACS periods: 2010 and 2015 (Appendix 1.4). This integrated dataset allowed changes in agricultural land (from IACS) to be monitored while at the same time incorporating accurately mapped non-agricultural land use data from LCM and the woodland inventory. Maps are shown in Appendix 1.4.

2.1.5.1 Recommended use

This dataset has a wide range of uses including land use and land cover change analysis and ecosystem services analysis and monitoring. For example, it has been used to provide land use inputs for the InVEST model (Sharp et al 2014) for the analysis of sediment and nutrient output (see, e.g. Hewitt et al 2018), and will likely form the base dataset used for land use modelling work in WP

1.4. It is recommended to review carefully the limitations of this dataset before using it.

2.1.5.2 Principal limitations

In addition to the limitations previously discussed for the IACS dataset (Section 2.1.1.3), one further limitation is that comparable land uses in each dataset do not precisely spatially coincide. The merge operation assumes IACS to be a superior measure of agricultural land use, for this reason the three agricultural categories from IACS take precedence over LCM categories which they overlap. However, LCM seems to show a larger area of agricultural land than IACS, these areas will be added to the new merged arable land category. This problem has no easy solution, since IACS is less reliable outside of agricultural land areas, so cannot serve as a replacement, but simply removing the non-coincident areas from LCM would create holes in the dataset. An idea of the extent of the problem can be obtained through a cross tabulation of LCM07 and IACS 2010 (Appendix 5.1, Appendix 5.1). The main areas of error relate to arable and grassland which are not clearly coincident across the two datasets.

2.1.6 IACS predominant land use extended classification integrated with LCM 2007 woodland dataset

2.1.6.1 General description

This dataset refers to the combined dataset created as for the previously described datasets, except that the IACS extended classification was used. Maps are shown in Appendix 1.5.

2.1.6.2 Recommended use

As for the previous dataset, this dataset was created to provide input for the InVEST nutrient and sediment model. The extended classification, in which key crop types with known nutrient loads are disaggregated, is more useful than the simple classification, since it allows different arable cropping regimes with correspondingly different nutrient loads to be separately modelled. In other words, rather than broadly estimating nutrient and sediment output for generic “arable” land, local scale differences in nutrient export associated with individual cropping regimes can be distinguished.

2.1.6.3 Principal limitations

The main limitations of this dataset relate to the different criteria used for mapping similar classes between the three datasets used, with the result that overlap between apparently similar thematic categories is not exact. See the discussion for the previous dataset (Section 2.1.4.3).

2.2 Ecosystems Services maps

The ecosystem services framework is a commonly-adopted measure of the benefits that nature provides to human well-being and quality of life (e.g. Ehrlich and Mooney 1983, Constanza and Daly 1992, Constanza et al 1997). Quantification of ecosystem services is a first step for their inclusion in policy and decision making. Ecosystem services are commonly classified according to a hierarchical framework that relates services to how they contribute to human well-being, known as the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potchin 2012). This framework, which is a refinement of the one proposed by the Millennium Ecosystems Assessment (MEA 2005), is the one chosen by the European Union and is the classification system followed in the work that we report on here.

Ecosystem services maps were commissioned by Scottish Government (in RD1.4.2 Gimona et al) and developed from available data sources under the Strategic Research Programme. Maps were prepared following the CICES classification (Haines-Young and Potchin 2012), which separates ecosystem services into three types, provisioning services, regulating services and cultural services, described in detail in the following sections. The maps produced are shown in Table 3.

Several of these spatial datasets/maps were produced using the InVEST suite of models (Sharp et al 2014). These are a set of openly available models that have been widely used to provide estimates of ecosystem services worldwide, including in the UK (e.g. Nelson et al 2009, Zhou et al 2010, Redhead et al 2016).

Table 3: Ecosystems services maps and related spatial indicators

No.	Name	Scale/resolution	Description/sources	Documentation	Created by*
Provisioning Services					
1	Water Supply	25m raster, aggregated to sub-catchment scale	The map, obtained using the InVEST 'water yield' model, ranks Scottish sub-catchments based on the total annual runoff form land.	http://www.arcgis.com/apps/MapSeries/index.html?appid=a1c9afe0f8594c3da68654f8124632fa	AG & ABC
2	Suitability for crop Production	1km	Indicator of crop production, correlated with crop yield for a range of crops but is not itself a quantification of yield of individual crops. The map was produced by integrating data from the Land Capability for Agriculture (LCA) analysis and 12 years of weekly time series of MODIS satellite data that provide a measure of plant productivity.	As above	AG
3	Cattle Density	2km gridded data on 25m raster	Gridded Agricultural Census data for cattle from EDINA (2 km resolution) were downscaled by redistributing recording cattle numbers at 2 km resolution onto 25 m grid cells of the land cover map.	As above	AG & ABC
4	Sheep Density	2km gridded data on 25m resolution raster	Data from Gridded Agricultural Census data from EDINA (2 km resolution) were down-scaled by redistributing sheep on grasslands derived from LCM 2007 and on moorland habitat that supports their grazing.	As above	AG & ABC
Regulating Services					
5	Water purification - nutrients	25m raster, aggregated to sub-catchment scale	The map, obtained using the InVEST nitrogen retention model, ranks Scottish catchments based on the total amount of nitrogen that runs off from the land but is retained before reaching the streams.	As above	AG & ABC
6	Soil retention	25m raster, aggregated to sub-catchment scale	The map, obtained using the InVEST soil and sediment retention model, ranks Scottish catchments based on the total amount of soil that is retained before reaching the streams, including soil that might be initially transported but is deposited later.	As above	AG & ABC
7	Soil Organic Carbon	1km resolution	The map, at 1 km resolution, is based on estimates of soil organic carbon stocks to up to 1 m depth. The estimates were obtained by relating field data contained in the National Soil Inventory	As above	LP & AG

	Stocks		of Scotland (NSIS) data base, to a range of environmental variables using Digital Soil Mapping methods. For example, topography and satellite data were used to produce the estimates for un-sampled locations.		
8	Pollination	100m resolution	The map shows an index of pollination service rescaled between 1 (highest) and 0 (lowest). The index is based on 6 species of bumble bee, namely *Bombus lapidarius, B. lucorum, B. muscorum, B. pascuorum, B. pratorum, and B. terrestris.* For each species the model had 4 main components:	As above	LP ,AG, RB, RP, ES

			a floral resources component (276 species), a nesting habitat component, a spatial component (to account for flight distance) and a time component (to account for flowering of floral resources and queen emergence). Flowering times of the species considered were obtained from several data bases, namely Bioflor, EcoFlora, LEDA. The species geographical distributions were taken from the Atlas of the British and Irish Flora and downscaled to 100 m (see the species richness section for more details). Each bumble bee species contributes to the service to flowering crops if these are within the species' maximum flight distance. The latter were mapped using agricultural census data from 2015. The service is defined only in proximity of flowering crops. The latter were mapped using agricultural census data.		
Cultural Services					
9	Recreation and Amenity	1km resolution (partial)	This map uses geo-referenced, crowd-sourced photographs as a synthetic indicator for intermediate cultural services such as Amenity, Aesthetics and Cultural Importance. We have mapped the number of unique submitters to Panoramio in each 1 km square as a (partial) indicator of the recreation service. Values (between 0 and 560 per per Km^2) are rescaled between 1 and 0 as in the other maps. The white areas did not have any uploaded photos.	As above	MC & AG
10	Plant Species Richness	1km resolution	The plant species richness map was obtained by down-scaling the distribution of all native flowering species (from the Atlas of the British and Irish Flora) to 1 km. For each 10 km square of the Atlas where a species was reported present, the down-scaling was carried out as follows: we attributed presence of the species to the broad habitats of the 25-m land cover map (LCM2007) in which it can live. Results were aggregated to 1 km. For each 1km square the number of present species was counted. The values were then rescaled between 1 (highest richness) and 0 (lowest).	As above	LP & AG
11	Floral Distinctivene ss	10km resolution	Local species richness is not a sufficient criterion to highlight areas that are important for the provision of plant diversity. It is also important to identify areas that have a distinctive species composition and, when taken together, provide a good overall representation of the species present in Scotland. The map, with values rescaled between 1 and 0, was obtained by using the 'Zonation' algorithm which ranks the cells in terms of their importance based on the 'global' (in this case at the scale of Scotland) loss of species suffered if a cell is removed.	As above	AG

*ABC =Andrea Baggio Compagnucci; AG = Alessandro Gimona; LP = Laura Poggio; MC = Marie Castellazzi; RB = Rob Brooker; RP = Robin Pakeman, ES = Enrico Simonetti

2.2.1 Provisioning Services

Provisioning services mainly comprise water, and food and fibres from the land. Functioning ecosystems are necessary to support the production of material goods that can be consumed directly, used for manufacturing other products or traded. The mapped indicators can be separated into two categories:

1 *Water*

Fresh water is used in homes and businesses, in agriculture and in power generation. The food and drink industry in Scotland crucially depends on water availability and quality, and the hydrological cycle sustains terrestrial and water ecosystems including rivers, lakes, and wetlands.

2 *Food and fibres*

One of the most long standing human activities, often connected to identity and culture, is the transformation of both lowland and upland ecosystems to provide food and fibres through farming. Food production creates wealth, and has impacts on health and the condition of landscapes and ecosystems. Livestock are a source of food and fibres. In Scotland cattle and hill and upland sheep farming plays an important role in the balance of multiple benefits derived from the land.

2.2.1.1 Water Supply

2.2.1.1.1 General description

Water Supply –runoff. See:

http://www.arcgis.com/apps/MapSeries/index.html?appid=a1c9afe0f8594c3da68654f8124632fa

The map, obtained using the InVEST 'water yield' model[2], ranks Scottish sub-catchments based on the total annual runoff form land. We estimated how each sub-catchment contributes annually to runoff production.

To produce this map, data were gathered from the literature, or generated our own spatial estimates, of average annual precipitation, how much water is lost (transpired) by different vegetation types, soil depth, soil water content available to plants, land use and land cover, and elevation. All original values (between 670 and 6400 m^3 per ha) were re-scaled between 0 and 1: the closer the values are to 1 the higher the runoff. Values close to 0 are at the lower end of the scale, but they don't mean that no run off occurs. Values close to 0 are at the lower end of the relative scale, but they don't mean that no run off occurs. There is a clear East-West gradient, reflecting topography and climate.

2.2.1.1.2 Recommended use

The main purpose of this dataset is to provide input data for analyses of water provision, shortages and needs.

[2] http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/reservoirhydropowerproduction.html

2.2.1.1.3 Principal limitations

The maps are based on average precipitation values taken over multiple years, and individual annual variability is not accounted for.

2.2.1.2 Suitability for Crop Production

2.2.1.2.1 General description

Suitability for crop production. See:
http://www.arcgis.com/apps/MapSeries/index.html?appid=a1c9afe0f8594c3da68654f8124632fa

This map depicts an indicator of suitability for crop production, which is correlated with crop yield for a range of crops but is not itself a quantification of yield of individual crops. The map was produced by integrating data from the Land Capability for Agriculture (LCA)[3] analysis and 12 years of weekly time series of MODIS satellite data that provide a measure of plant productivity. High LCA class labels indicate low potential for production (e.g. 7 is the lowest level of production). To produce this map, areas with land capability scores of 3.1 and below (i.e. with better potential for crop production) were classified as 'High' potential if they also had consistently high MODIS productivity over the 12 years (i.e. if their potential productivity was being realised); otherwise they were classified as 'Medium'.

Areas with land capability between 3.2 and 4.2 were classified as 'Medium' if they had high MODIS productivity otherwise they were classified as 'Low'. All areas with land capability poorer than 4.2 were classified as having 'extremely low' crop production potential. In this map, a wide range of crops expecting good yields can be cultivated commercially on high potential areas, while a more restricted range of crops can be cultivated commercially on areas of low potential.

2.2.1.2.2 Recommended use

As an indicator of suitability for crop production to be used in future analyses.

2.2.1.2.3 Principal limitations

The indicator relates to suitability only, and does not provide figures for actual yield.

2.2.1.3 Cattle Density

2.2.1.3.1 General description

Cattle density**.** See: http://www.arcgis.com/apps/MapSeries/index.html?

appid=a1c9afe0f8594c3da68654f8124632fa

The map is based on gridded Agricultural Census data, and can be used to provide a broad scale impression of the pattern of production. The Agricultural Census is conducted in June each year by the Scottish government. Each farmer declares the agricultural activity on the land via a postal questionnaire. One of the products derived from this census is a gridded data set produced by the Edinburgh University Data Library (EDINA)[4].

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[3] http://www.hutton.ac.uk/learning/exploringscotland/land-capability-agriculture-scotland

[4] http://agcensus.edina.ac.uk/

Values ranged between 0 and 4 per ha. The map shows values of cattle per ha rescaled between 0 (very low density) and 1 (highest density). Gridded Agricultural Census data for cattle from EDINA (2 km resolution) were downscaled by redistributing recording cattle numbers at 2 km resolution onto 25 m grid cells of the land cover map. The land cover map used was LCM 2007.

The greatest density of cattle is in Dumfries & Galloway, with high density also in Ayrshire, some areas of Grampian and of the Highlands. Grasslands used by cattle on farmland tend to occur where crop cultivation is limited by climate, slope, or wetness.

2.2.1.3.2 Recommended use

To provide a broad scale impression of the pattern of livestock production for cattle, to help identify areas at risk of suffering negative impacts from livestock concentrations.

2.2.1.3.3 Principal limitations

The main problem relates to the low spatial resolution of the original data, which were aggregated to 2 km resolution for data protection purposes.

2.2.1.4 Sheep Density

2.2.1.4.1 General description

Sheep density. See: http://www.arcgis.com/apps/MapSeries/index.html?

appid=a1c9afe0f8594c3da68654f8124632fa There are ca 6.8 million sheep in Scotland with Scottish

annual meat production around 61,000 tons.

Values ranged between 0 and 220 sheep per squared km (2.2/ha). The map shows values of sheep per ha, rescaled between 0 (very low density) and 1 (highest density). As for cattle, data from Gridded Agricultural Census data from EDINA (2 km resolution) were down-scaled by redistributing sheep on grasslands derived from LCM 2007 and on moorland habitat that supports their grazing.

The highest sheep density is in the Scottish Borders, Dumfries and Galloway, Aberdeenshire, and some areas of the Highlands.

2.2.1.4.2 Recommended use

To provide a broad scale impression of the pattern of livestock production for sheep, to help identify areas at risk of suffering negative impacts from livestock concentrations.

2.2.1.4.3 Principal limitations

As for cattle density, above (Section 2.2.1.3.3).

2.2.2 Regulating Services

Regulating Services refers to the beneficial regulatory functions carried out by ecosystems. Functioning ecosystems undertake processes that are beneficial for society; for example, regulation of water and soil quality through natural purification, pollination, climate regulation, disease and pest regulation. These benefits are generated through the interactions among living and non living elements of the ecosystems: for example water purification derives from soil organisms' activity and from the mechanical ability of soil and vegetation to trap and transform nutrients, pollutants and/or and pathogens.

2.2.2.1 Water purification - nutrients

2.2.2.1.1 General description

Water purification **-** Nitrogen Retention**.** See: http://www.arcgis.com/apps/MapSeries/index.html?appid=a1c9afe0f8594c3da68654f8124632fa

The map, obtained using the InVEST nitrogen retention model[5] , ranks Scottish catchments based on the total amount of nitrogen that runs off from the land but is retained before reaching the streams. The model uses the amount of nitrogen loaded on each land use type, calculates the annual average water runoff, and then it computes the quantity of nitrogen retained by each pixel based on the land use efficiency (expressed as the percentage of load that will be retained) and on how the water is routed through the landscape. By the routing process the model calculates how much of the nitrogen loaded on land reaches stream and how much is retained. It then aggregates the values to the sub-watershed level. As in the case of Water Supply, the values in the map were re-scaled between 0 and 1 and the same interpretation applies, with 0 being interpreted as a value indicating lowest relative nitrogen retention.

The map shows more nitrogen was added to agricultural areas, compared to non-agricultural areas, leading to greater levels retained and exported.

2.2.2.1.2 Recommended use

This indicator is principally useful for understanding the retention of nutrients at the sub-catchment scale.

2.2.2.1.3 Principal limitations

Interaction with groundwater level, transformation during the routing made by soil, bacteria or the interaction of the water with biophysical processes were not considered.

Nutrient loads were based on tables published by DEFRA, not measured in the field. Clearly, future work should consider obtaining more accurate estimates using field measurements. Since the data were aggregated to sub-catchments the provide no information on individual variation within the sub-catchment itself.

2.2.2.2 Soil retention

2.2.2.2.1 General description

Soil Retention**.** See: http://www.arcgis.com/apps/MapSeries/index.html?appid=a1c9afe0f8594c3da68654f8124632fa

Soil is associated with a wide range of essential functions, such as plant and crop growth, regulating the amount of water flowing into rivers, storing carbon. Vegetation provides a vital service by retaining soil. This benefits both terrestrial and aquatic systems. The map, obtained using the InVEST soil and sediment retention model[6], ranks Scottish catchments based on the total amount of soil

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[5] http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/ndr.html
[6] http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/sdr.html

that is retained before reaching the streams, including soil that might be initially transported but is deposited later.

The retention service provided by vegetation cover is higher where topography and climate pose more risk of erosion. Before estimating retention, the model uses the Universal Soil Loss Equation (USLE), which integrates information on vegetation cover, soil properties, topography, rainfall and climate data to estimate soil erosion from a grid cell.

2.2.2.2.2 Recommended use

To highlight differences in soil retention provision across Scotland and areas at greatest risk of erosion, allowing potential mitigation option to be considered (e.g. tree planting).

2.2.2.2.3 Principal limitations

The soil retention service mapping is not very reliable for peat soils. Slope is the main factor which influences the soil formation and the quantity of material available to be moved from original areas and transported downstream.

2.2.2.3 Soil Organic Carbon Stocks

2.2.2.3.1 General description

Soil Organic Carbon Stocks**.** See: http://www.arcgis.com/apps/MapSeries/index.html?

appid=a1c9afe0f8594c3da68654f8124632fa

Soil is an important carbon sink, and globally soil stores two to three times more carbon than the atmosphere. In Scotland, there is often two to four times more carbon in the soil than in the vegetation. By sequestering carbon that would otherwise contribute to greenhouse gases, soil organic carbon contributes to mitigation of climate change. The map, at 1 km resolution, is based on estimates of soil organic carbon stocks to up to 1 m depth. The estimates were obtained by relating field data contained in the National Soil Inventory of Scotland (NSIS)[7] data base, to a range of environmental variables using Digital Soil Mapping methods (Poggio and Gimona 2014). For example, topography and satellite data were used to produce the estimates for un-sampled locations. The values, ranging between 60 and 1500 tons per ha, are re-scaled between 1 (highest) and 0 (lowest). The highest values occur on peatlands in the Highlands and the Hebrides. The total carbon stocks estimated for Scottish soils were around 3000 Mt.

2.2.2.3.2 Recommended use

This indicator offers a useful approximation of the total carbon sequestration capability of Scotland's soils.

2.2.2.3.3 Principal limitations

Stocks were only measured down to a depth of 1 m. Peatland soils in many areas are much deeper, so the 3000 Mt figure is certainly too low.

[7] http://www.hutton.ac.uk/about/facilities/national-soils-archive/resampling-soils-inventory

2.2.2.4 Pollination

2.2.2.4.1 General description

Pollination**.** See: http://www.arcgis.com/apps/MapSeries/index.html?

appid=a1c9afe0f8594c3da68654f8124632fa

Healthy populations of pollinators are important for food security and for the reproduction of numerous species of wild plants. Some pollinators, especially bees, are declining, either because they lack specific resources, such as flowers and nesting habitat, and/or because multiple risk factors, including pesticides and climate change, are reducing their numbers. The map shows an index of pollination service rescaled between 1 (highest) and 0 (lowest). The index is based on 6 species of bumble bee, namely *Bombus lapidarius, B. lucorum, B. muscorum, B. pascuorum, B. pratorum, and B. terrestris.* For each species the model had 4 main components: a floral resources component (276 species), a nesting habitat component, a spatial component (to account for flight distance) and a time component (to account for flowering of floral resources and queen emergence). Flowering times of the species considered were obtained from several data bases, namely Biolflor[8], EcoFlora[9], and LEDA[10]. The species geographical distributions were taken from the Atlas of the British and Irish Flora[11] and downscaled to 100 m (see the species richness section for more details). Each bumble bee species contributes to the service to flowering crops if these are within the species' maximum flight distance. The latter were mapped using agricultural census data from 2015. The service is defined only in proximity of flowering crops. The latter were mapped using agricultural census data. High levels are predicted in areas like the Spey valley and in the upland-lowland transition, where flowering and nesting resources are more available, while many lowland areas, more intensely farmed, have relatively low levels of pollination service. While bumble bees are good indicator species, more pollinators could be used in the future to have a more complete picture of the service.

2.2.2.4.2 Recommended use

Provides an estimate of the degree of pollination service potentially available to agricultural areas, and a measure of the extent to which this service may be negatively impacted by farming or other land use practices.

2.2.2.4.3 Principal limitations

The main limitation of this indicator is that it does not account for other pollinators, such as butterflies and hoverflies. The value of this indicator would be increased by repeated sampling at frequent intervals, enabling a picture of pollination service change over time to be obtained.

2.2.3 Cultural Services

The precise definition of cultural ecosystem services (CES) is still being debated, and therefore it is challenging to decide what aspects of CES to map. The Millennium Ecosystem Assessment defined cultural ecosystem services as "the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences",

[8] http://www.ufz.de/index.php?en=38567
[9] http://ecoflora.org.uk/
[10] https://uol.de/en/biology/landeco/research/projects/leda/
[11] https://www.brc.ac.uk/plantatlas/

while the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) points out that a sense of cultural identity is needed for good quality of life. It is broadly agreed that CES are intangible, and linked to identity, meaning and experience, and this makes them both difficult and important to assess. Mapping all CES is not always possible or necessary. However, we have tried to map some indicators of recreation and amenity, and we have placed biodiversity among cultural services, emphasising its importance for human cultural fulfilment. This does not deny the important, but poorly understood, role of biodiversity in ecosystem functioning.

2.2.3.1 Recreation and Amenity

2.2.3.1.1 General description

Volunteered Photographs. See: http://www.arcgis.com/apps/MapSeries/index.html?

appid=a1c9afe0f8594c3da68654f8124632fa

Photo-sharing services, such as Panoramio[12] and Flickr[13] provide geo-referenced crowd-sourced photographs. A partial indicator of recreation service provision was obtained by mapping the number of unique submitters from several thousand contributors to Panoramio in each 1 km square. Values (between 0 and 560 per per Km^2) are rescaled between 1 and 0 as in the other maps. The white areas did not have any uploaded photos. The map shows the highest density along the Great Glen, the Spey valley, on the mountains of the Cairngorms National Park, and in some urban areas. While the submitters are self-selected, introducing potential bias, the high number of unique users (several thousand) is an advantage over rigorous surveys, with number of participants one or two orders of magnitude higher than the typical survey. Understanding the spatio-temporal patterns of photo contributions will allow us better to assess the suitability of these data for mapping recreation and amenity.

2.2.3.1.2 Recommended use

These photos can provide valuable information such as identifying travel routes and tourist hot spots. It can be argued that they provide a synthetic indicator for intermediate cultural services such as Amenity, Aesthetics and Cultural Importance.

2.2.3.1.3 Principal limitations

The crowd-sourced geo-referenced photographs are a self-selected sample from individuals who choose to submit photographs. It does not account for the preferences of other users who have visited these or other locations but not submitted a photograph. Factors like accessibility of the photographed locations are also influential but have not been controlled for. Work to address these limitations is ongoing (Baggio Compagnucci et al 2018).

2.2.3.2 Plant Species Richness

2.2.3.2.1 General description

Plant Species Richness: See http://www.arcgis.com/apps/MapSeries/index.html?

appid=a1c9afe0f8594c3da68654f8124632fa

[12] https://www.panoramio.com/
[13] https://www.flickr.com/

This is an indicator of biodiversity, which, as explained above, in this context is related to cultural and spiritual fulfilment. Although biodiversity in general is believed to have an important role in ecosystem processes, we have not been able, so far, to investigate this aspect at the scale of Scotland. The plant species richness map was obtained by down-scaling the distribution of all native flowering species (from the Atlas of the British and Irish Flora) to 1 km. For each 10 km square of the Atlas where a species was reported present, the down-scaling was carried out as follows: we attributed presence of the species to the broad habitats of the 25-m land cover map (LCM2007) in which it can live. Results were aggregated to 1 km. For each 1 km square the number of present species was counted. The values were then rescaled between 1 (highest richness) and 0 (lowest). On the map, a lowland-upland and a North-South gradient can be observed. The richest areas are in uplands of Dumfries and Galloway, Lothians, Perthshire, and in the Spey valley. Montane areas and bogs tend to be less diverse (but often have a distinctive flora) because fewer species can tolerate conditions there. It should be noticed that plant species richness is not necessarily correlated with the richness of other taxa. Therefore, further work is needed to produce an indicator of overall species richness.

2.2.3.2.2 Recommended use
Serves as a partial indicator of biodiversity across different areas of Scotland.

2.2.3.2.3 Principal limitations
The use of species richness as a biodiversity indicator has a number of well-known limitations, including differential sampling effort and variability in species abundance (Gotelli and Colwell 2001). Aside from these general limitations, plant species richness is poorly understood at the macro scale. At the same time, lack of information on other species (e.g. for birds), due to the high cost of obtaining the data, acts a major barrier to obtaining an overall indicator.

2.2.3.3 *Floral Distinctiveness*

2.2.3.3.1 General description
Floral Distinctiveness: See:

http://www.arcgis.com/apps/MapSeries/index.html?appid=a1c9afe0f8594c3da68654f8124632fa

This is another indicator of biodiversity, and therefore related to cultural and spiritual fulfilment. Local species richness is not a sufficient criterion to highlight areas that are important for the provision of plant diversity. It is also important to identify areas that have a distinctive species composition and, when taken together, provide a good overall representation of the species present in Scotland. The map, with values rescaled between 1 and 0, was obtained by using the Zonation algorithm (Moilanen 2007) which ranks the cells in terms of their importance based on the 'global' (in this case at the scale of Scotland) loss of species suffered if a cell is removed.

The more distinctive the contribution, the higher the importance of a map square. If a species were present in only a small area, that area would be deemed irreplaceable. The footprint of 10 km squares of the floral Atlas is still clearly visible; therefore, borders between areas of different value are sharper than in reality. Notice that the areas of high distinctiveness have to be conserved together because they have complementary species composition. Therefore, while map squares with high distinctiveness might not have a particularly high local diversity, they provide a distinctive contribution to the overall set of plant species present in Scotland.

2.2.3.3.2 Recommended use

Serves as a partial indicator of biodiversity across different areas of Scotland.

2.2.3.3.3 Principal limitations

The main limitations of this indicator relate to the use of plant species only. Further future work could include further indicators that encompass more species of vertebrates and invertebrates, and provide a more complete picture.

3 Discussion and Recommendations

The above overview has provided a concise summary of the integrated spatial datasets recently developed under RESAS WP 1.4.2 and 1.4.3 on land use and ecosystem services for the whole of Scotland, we have presented these for the two National Parks and for the Aberdeenshire River Dee. These spatial datasets have potential to support evidence-driven policy making around adaptive and integrated land management in these areas. These data form a useful basis for analysis aimed to mitigating negative land use and ecosystems service impacts, and providing support to land managers and other policy stakeholders. However, the datasets described do have some limitations, which we summarize below. It should be noted that all datasets have errors, and detailed description of the limitations provided in this report should not be taken as an indicator of poor quality relative to other sources. Rather, good practice requires that limitations should be properly described and documented.

3.1 Integrated spatial land use datasets

The integrated land use datasets suffer from a range of limitations inherent in the use of mixed data from multiple sources or data not originally intended for that purpose. The IACS data are not ideally suited for use as a land use time series, since land parcels record only the use claimed under the agricultural payments system, with the result that cessation of claims results in the disappearance of a land use from one date to the next in a way that does not reflect reality. In addition, they provide a poor record of land use outside of the most important agricultural areas due to the lesser importance of these areas in the payments system. Integration of these datasets with the Land Cover Map for 2007 is also fraught with difficulties, since broadly equivalent thematic categories in each of the datasets do not coincide spatially, meaning that combining the two datasets introduces errors from each and multiplies the level of uncertainty. Additionally, some of the data used (e.g. LCM 2007) are outdated. Clearly, integration with a more recent land cover map (LCM 2015) would be more desirable, yet this is at present unavailable to the James Hutton Institute due to its high cost.

3.2 Ecosystems Services maps

Limitations of the individual ecosystems services maps have been described above and will not be repeated here. Overall, the main limitations of the ecosystems services dataset as a whole relate to its reliance on published, rather than directly measured data, together with insufficient resources available for acquisition of proprietary, third party data, e.g. on forest biomass, BTO Bird Atlas data, Land Cover Map etc. Additionally, several indicators relate to environmental variables undergoing constant change, e.g. pollination, biodiversity; these would be more useful as part of an ongoing monitoring programme rather than as a standalone collection of maps.

3.3 Suggestions for future work to address these limitations

A number of recommendations can be made relating to the future development of these kinds of integrated datasets and acquisition of the base datasets that facilitate such development. In particular:

1 We recommend the development of a land use and land cover time series for Scotland (updated at least at decadal intervals) using a single methodology and thematic classification. Though such a task is well within the technical capabilities of the JHI, it requires a long-term funding commitment. The difficulty of securing this commitment in the past has meant that LCS88, developed by the then Macaulay Land Research Institute, has remained as a single time snapshot, severely compromising its usefulness for change monitoring.

2 In terms of ecosystems services and natural capital mapping generally, aspirations for understanding the evolution of Scotland's natural capital and services flowing from it need to be matched by appropriate data collection campaigns. Effective monitoring of Scotland's land-based natural capital would require a considerable sampling effort over many years, similar to that carried out by environment agencies on the water environment. Although it is expensive, this cannot be avoided if estimates of change are needed with a degree of uncertainty low enough to be useful for policy. In the short term, significant improvements can be made with relatively minor investment, e.g. collecting data on livestock nutrient production and spreading (on land) from Scottish catchments, instead of relying on published data from DEFRA.

4 Next steps

The datasets developed provide a springboard for a series of land-based analyses beginning in November 2018. These include (but are not limited to) the following (where these relate to specific aspects of the delivery framework, the WP number is given):

4.1 Analysis of land use and land cover change in case study areas (WP 1.4.3)
4.2 Analysis of land cover change in relation to land capability (WP 1.4.2)
4.3 Modelling land use change scenarios for agriculture and forestry in Scotland under Representative Concentration Pathways (RCPs) storylines (WP 1.4.2)
4.4 Investigation of trade-offs in natural Protected areas (e.g. Cairngorms National Park)
4.5 Identification of ecological connectivity for broadleaved and coniferous woodland
4.6 Work to improve the quality of cultural ecosystems services, e.g. integration of Flickr with other user-created photographic datasets, and improved methods for determining landscape attractiveness.

Acknowledgements

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8 References

Baggio Compagnucci, A., Gimona, A., and Castellazzi, M. (2018). A methodological framework to derive landscape attractiveness from voluntarily submitted photographs. ESP Europe 2018 Regional Conference “Ecosystem Services in a changing world: moving from theory to practice”. Abstract and oral presentation.

Costanza, R., & Daly, H. E. (1992). Natural capital and sustainable development. Conservation biology, 6(1), 37-46.

Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., ... & Raskin, R. G. (1997). The value of the world's ecosystem services and natural capital. nature, 387(6630), 253.

Ehrlich, P. R., & Mooney, H. A. (1983). Extinction, substitution, and ecosystem services. BioScience, 33(4), 248-254.

Gimona, A.; Poggio, L; Baggio Compagnucci, A.; Castellazzi, M.; Pakeman, R.J.; Brooker, R. (2018) Web story on ecosystem services in Scotland. *Web story: Indicators of Ecosystem Services in Scotland.*

Gotelli, N. J., & Colwell, R. K. (2001). Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology letters*, *4*(4), 379-391.

Haines-Young, R., & Potschin, M. (2012). Common international classification of ecosystem services (CICES, Version 4.1). European Environment Agency, 33.

Hewitt, R.J.; Baggio Compagnucci, A.; Castellazzi, M.; Matthews, K.B.; Miller, D.G.; Wardell-Johnson, D.; Gimona, A. (2018) Nitrogen and sediment export associated with four agricultural payment scenarios for 2019 using integrated land claims-land use-forestry data (IACS-LCM2007). Report.

MEA (Millennium Ecosystem Assessment) (2005). Ecosystems and human well-being: synthesis. Millennium Ecosystem Assessment Series. *World Resources Institute, Washington, DC*, 155.

Metzger, M. J., Rounsevell, M. D. A., Acosta-Michlik, L., Leemans, R., y Schröter, D. (2006). The vulnerability of ecosystem services to land use change. Agriculture, Ecosystems & Environment, 114(1), 69-85.

Moilanen, A. (2007). Landscape zonation, benefit functions and target-based planning: unifying reserve selection strategies. Biological Conservation, 134(4), 571-579.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., ... & Lonsdorf, E. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment, 7(1), 4-11.

Poggio, L., & Gimona, A. (2014). National scale 3D modelling of soil organic carbon stocks with uncertainty propagation—an example from Scotland. *Geoderma*, *232*, 284-299.

Redhead, J. W., Stratford, C., Sharps, K., Jones, L., Ziv, G., Clarke, D., ... & Bullock, J. M. (2016). Empirical validation of the InVEST water yield ecosystem service model at a national scale. Science of the Total Environment, 569, 1418-1426.

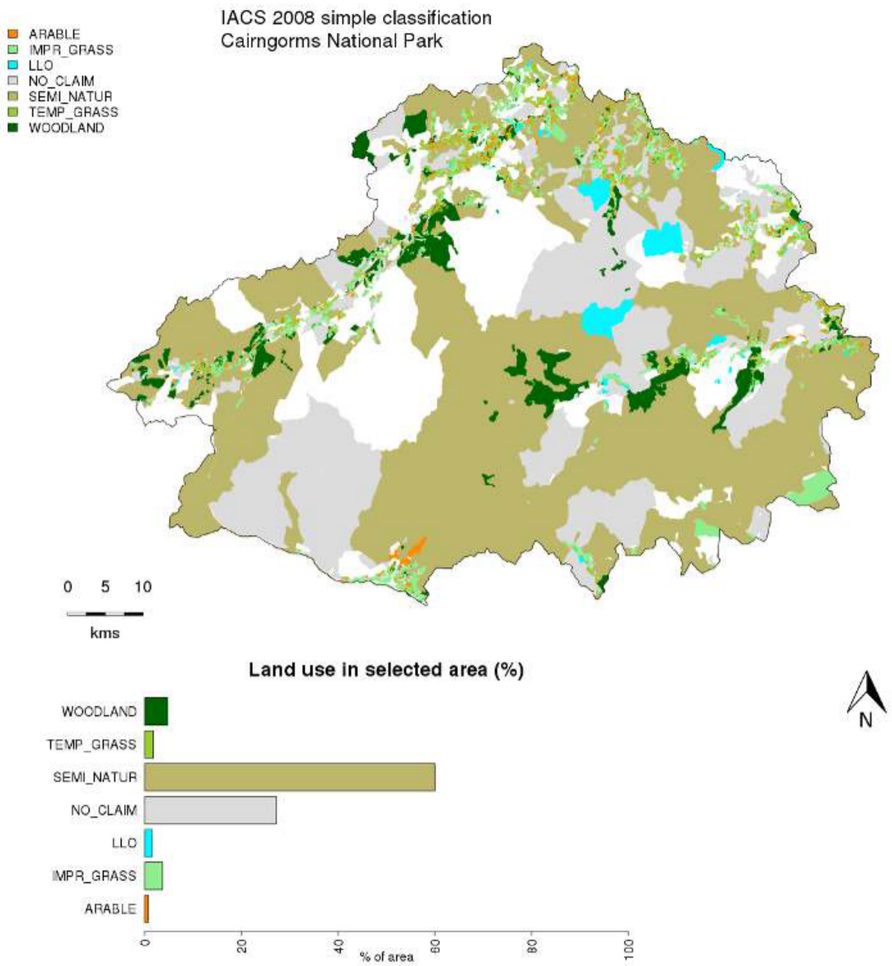
Sharp, R., Tallis, H. T., Ricketts, T., Guerry, A. D., Wood, S. A., Chaplin-Kramer, R., ... & Vigerstol, K. (2014). InVEST user's guide. The Natural Capital Project: Stanford, CA, USA.
http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/
Accessed: October 2018.

Turner B.L.II, Lambin E.F., & Reenberg A. (2007) The emergence of land change science for global environmental change and sustainability. Proc Natl Acad Sci USA 104:20666–20671.

Zhou, B., Yu, X. X., Chen, L. H., ZHANG, Z. M., Lü, X. Z., & FAN, M. R. (2010). Soil erosion simulation in mountain areas of Beijing based on InVEST Model. Research of Soil and Water Conservation, 17(6), 9-13.

9 Appendices

Appendix 1: Maps of integrated land use datasets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Land use in selected area (%)
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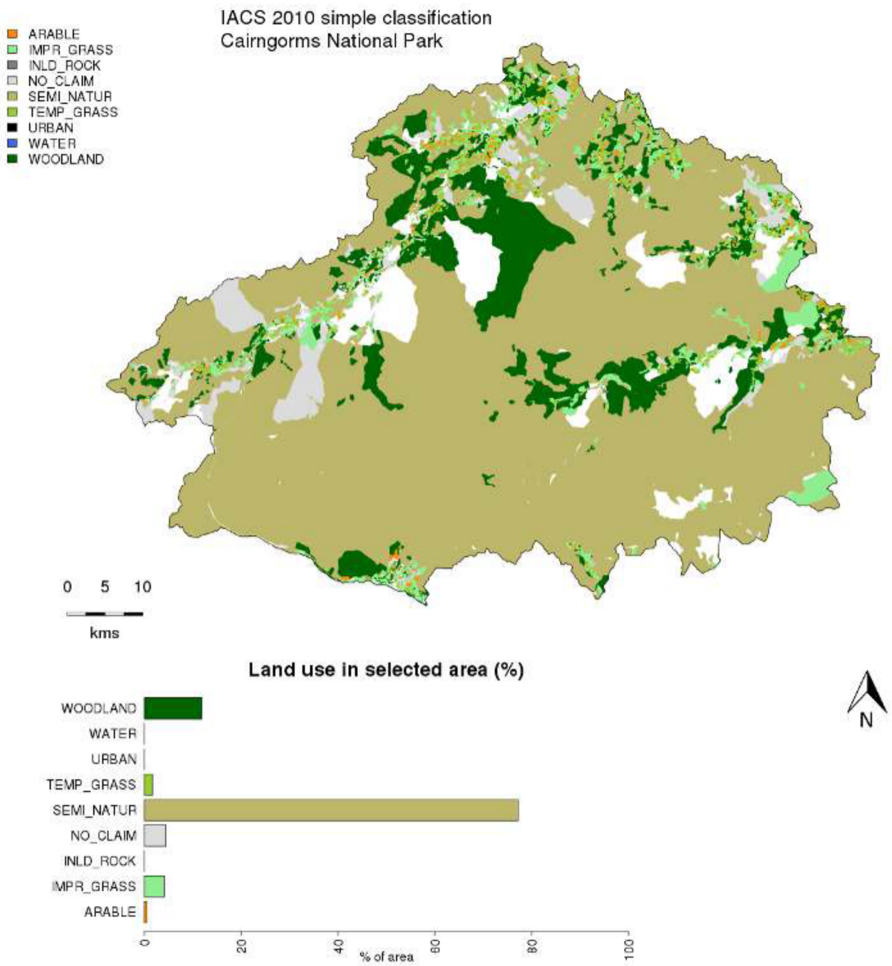
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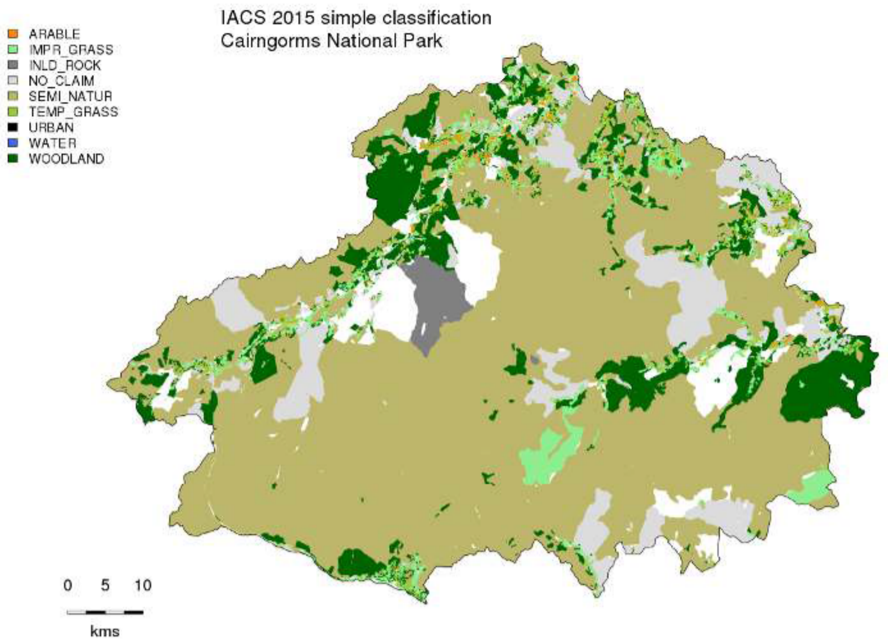
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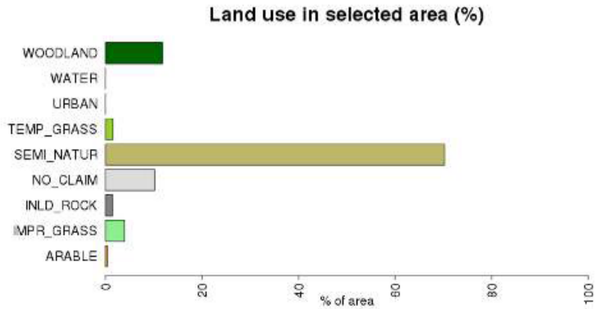
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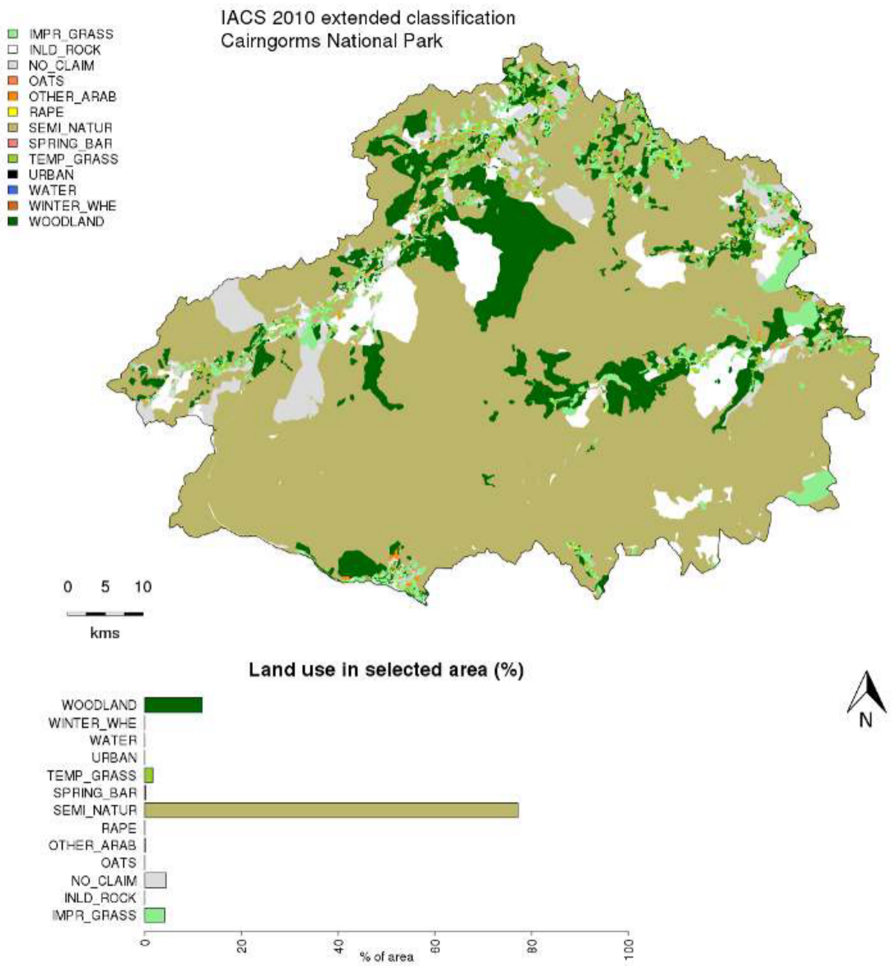
% of area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RAPE
SEMI_NATUR
SPRING_BAR
TEMP_GRASS
URBAN
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Land use in selected area (%)
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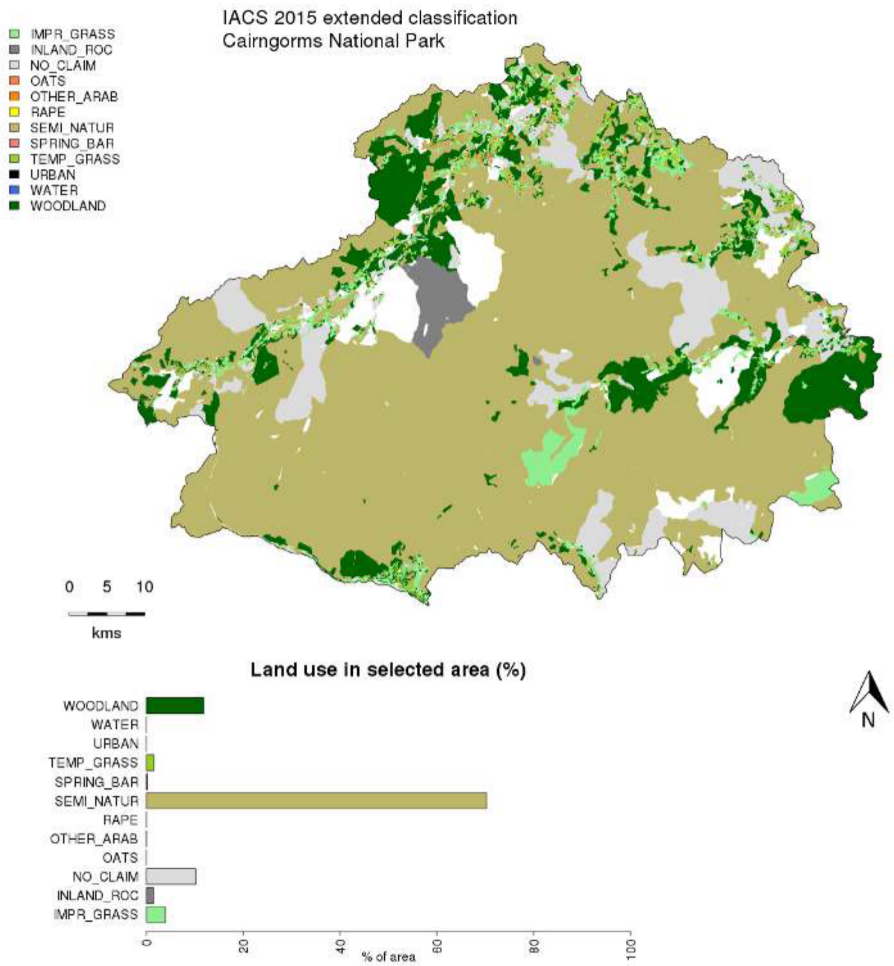
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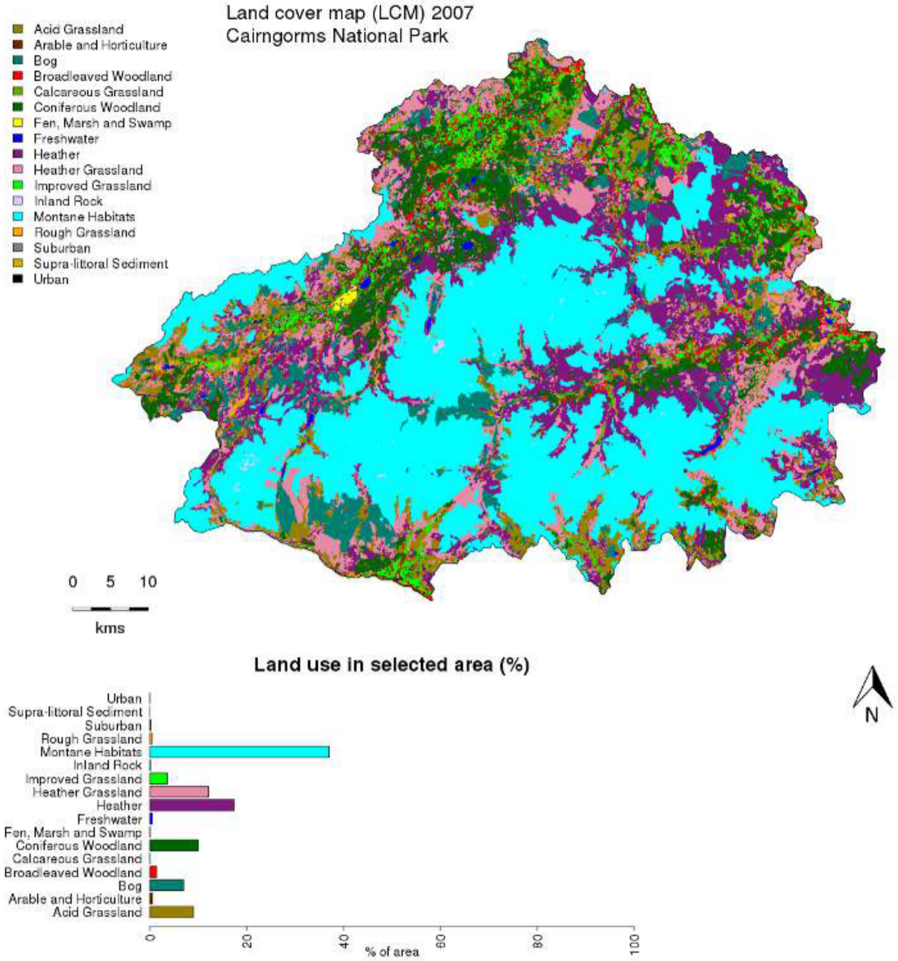
Land cover map (LCM) 2007
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Arable and Horticulture
Bog
Broadleaved Woodland
Calcareous Grassland
Coniferous Woodland
Fen, Marsh and Swamp
Freshwater
Heather
Heather Grassland
Improved Grassland
Inland Rock
Montane Habitats
Rough Grassland
Suburban
Supra-littoral Sediment
Urban
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kms
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Land use in selected area (%)
Urban
Supra-littoral Sediment
Suburban
Rough Grassland
Montane Habitats
Inland Rock
Improved Grassland
Heather Grassland
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Freshwater
Fen, Marsh and Swamp
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Arable and Horticulture
Acid Grassland
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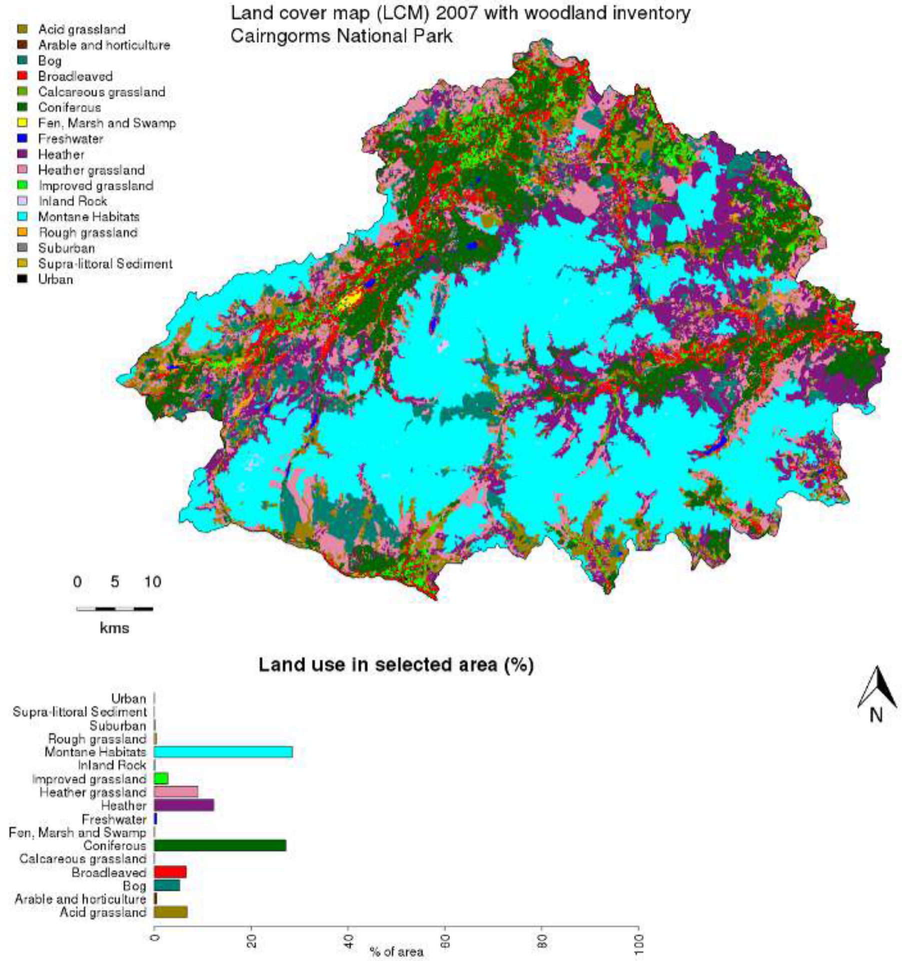
% of area


Land cover map (LCM) 2007 with woodland inventory
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Arable and horticulture
Bog
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Calcareous grassland
Coniferous
Fen, Marsh and Swamp
Freshwater
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Improved grassland
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Montane Habitats
Rough grassland
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Supra-littoral Sediment
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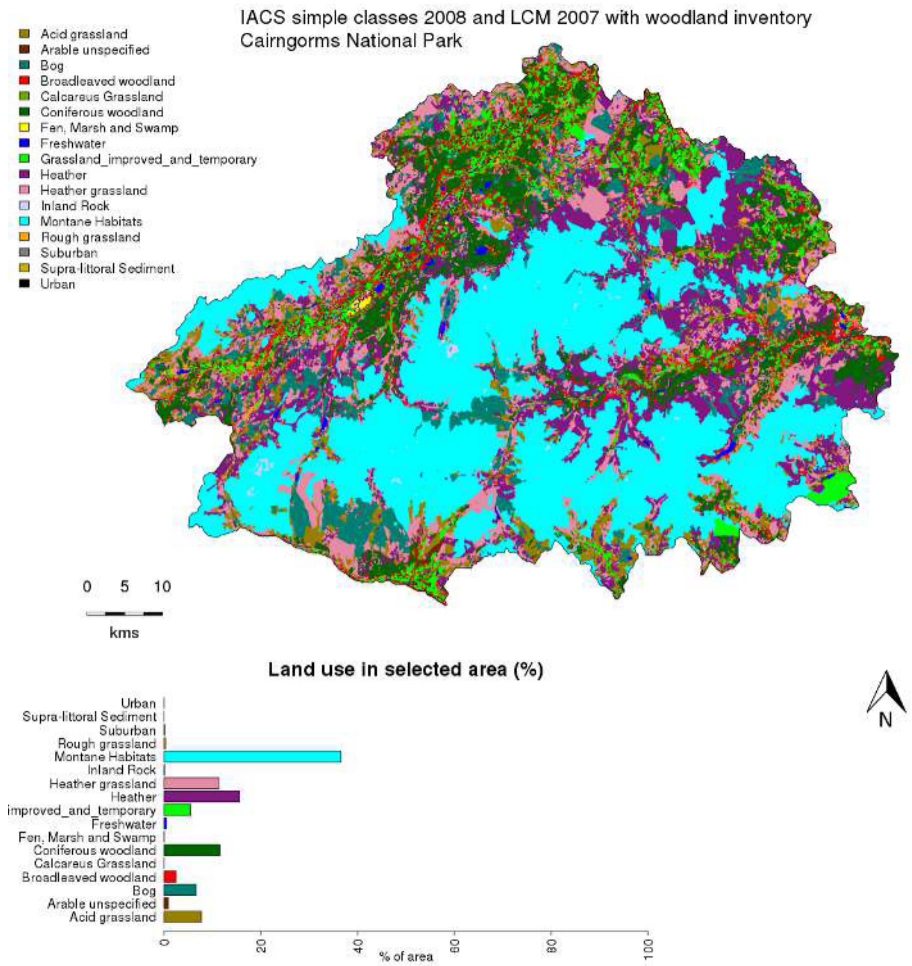
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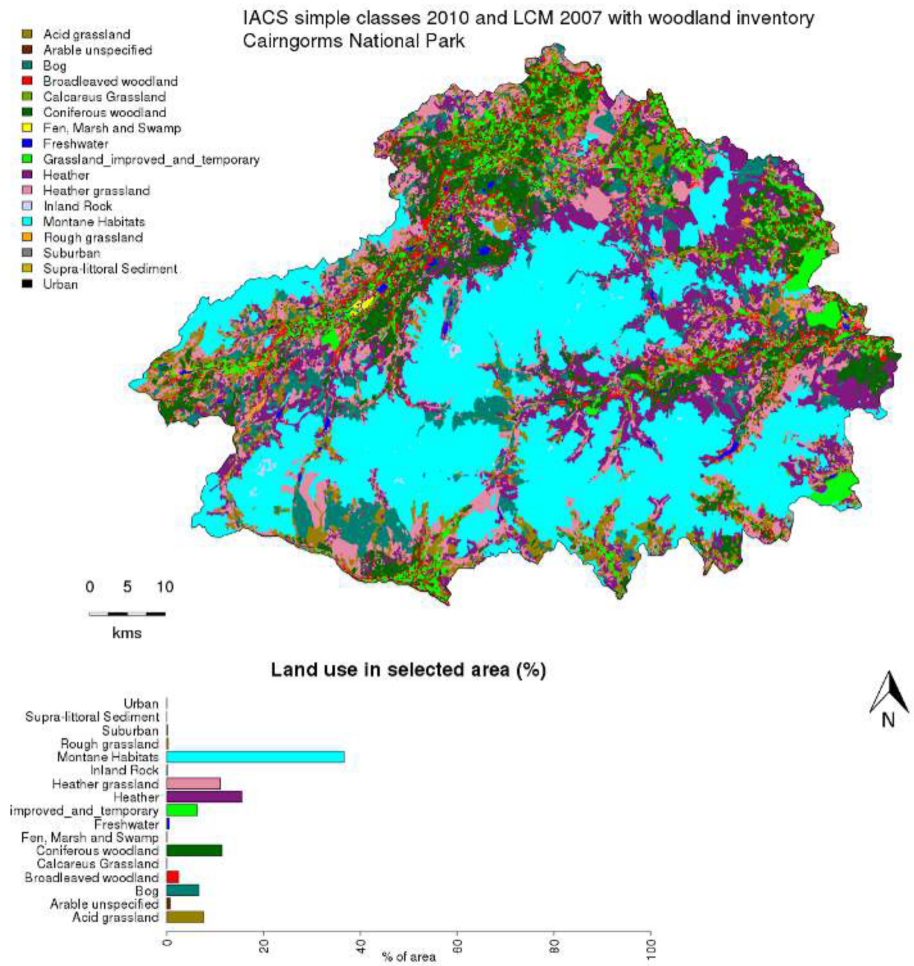
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Land use in selected area (%)
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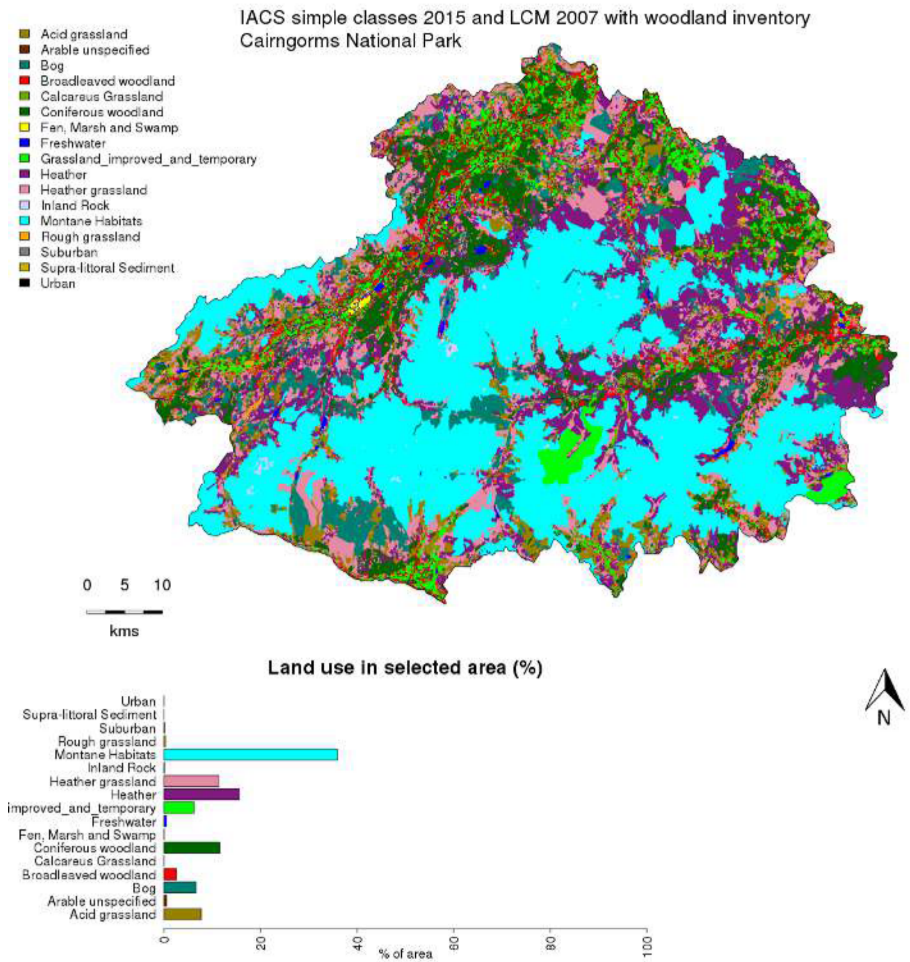
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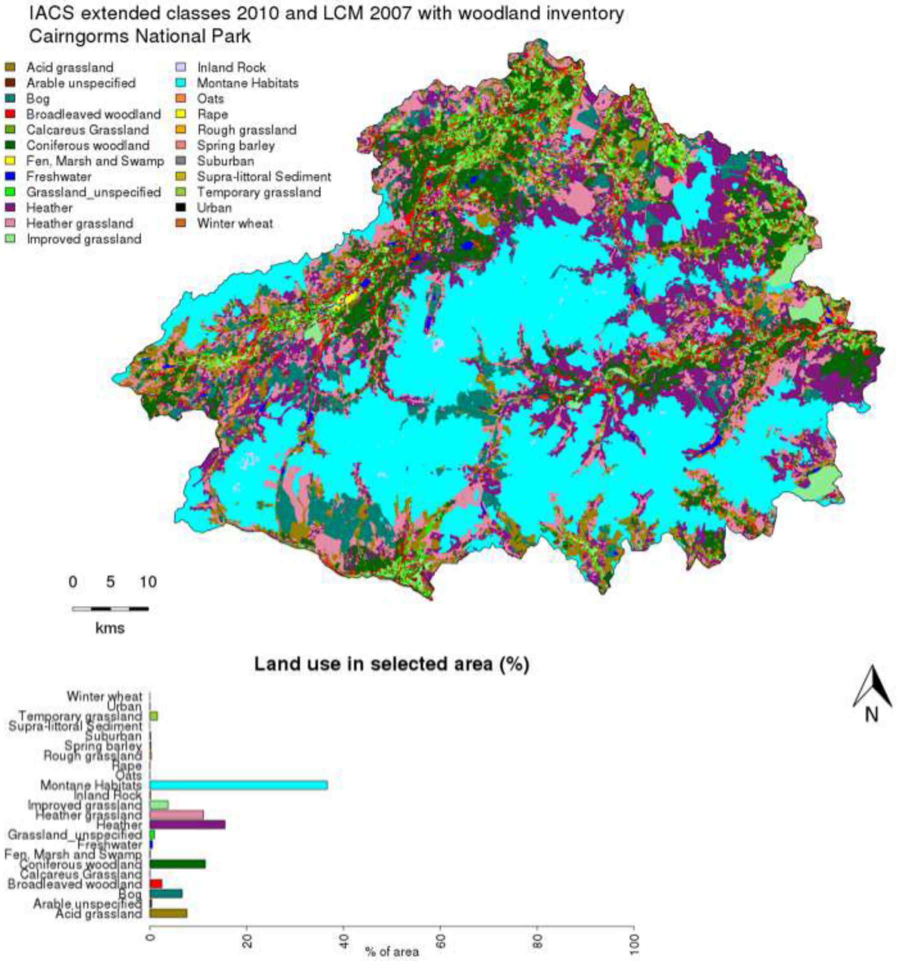
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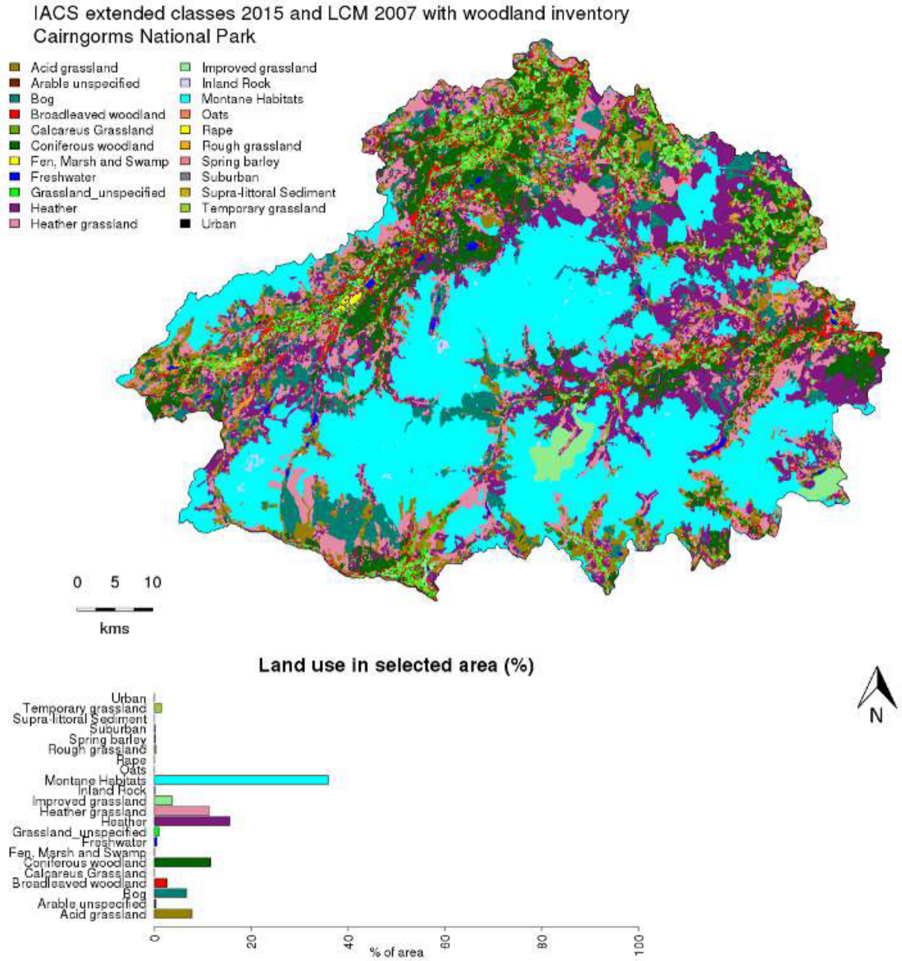
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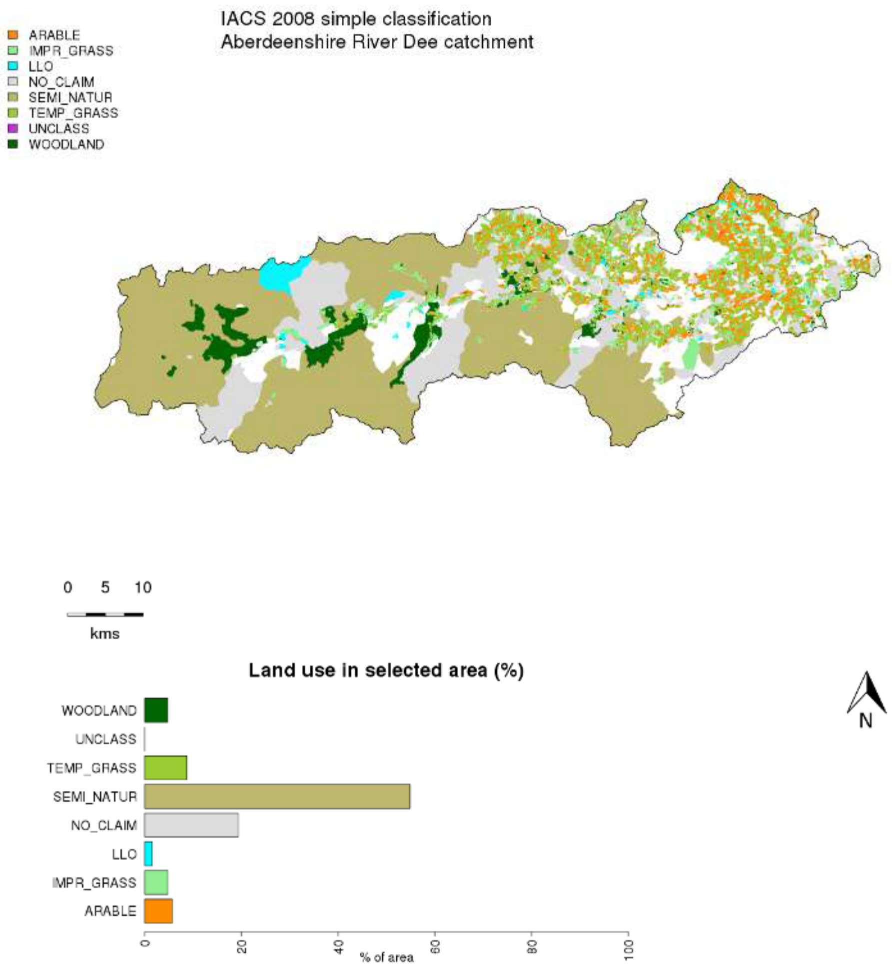
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UNCLASS
WOODLAND
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kms
Land use in selected area (%)
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% of area
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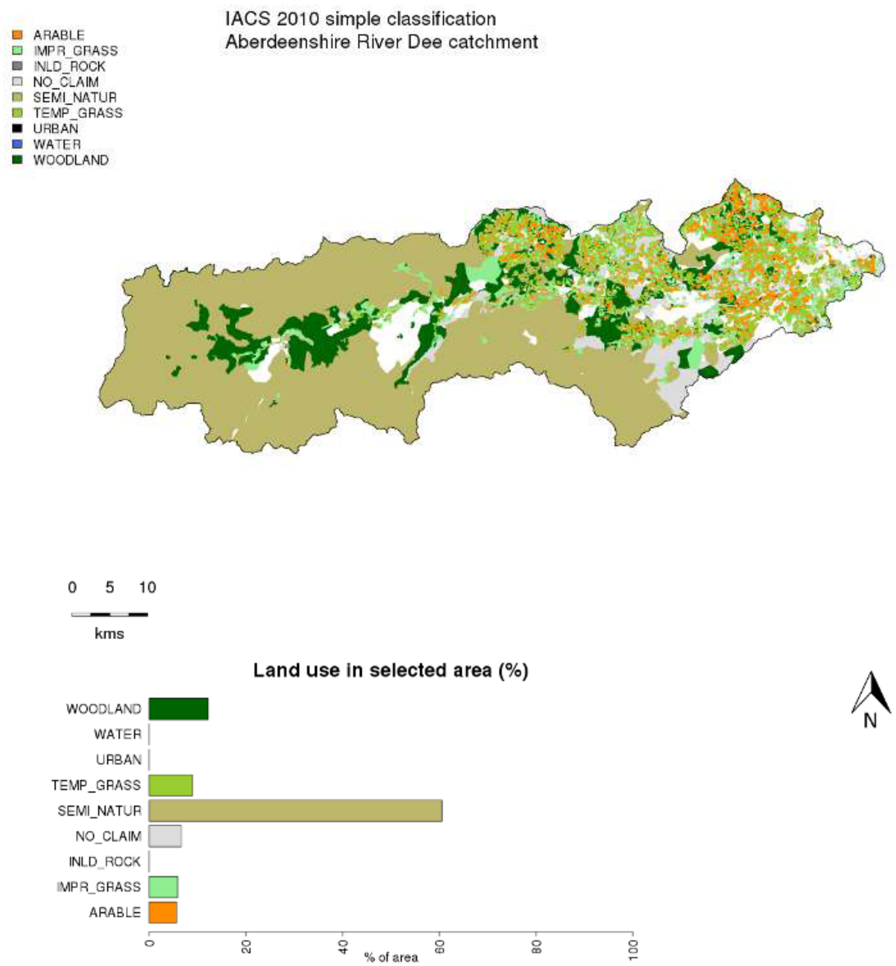
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WOODLAND
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Land use in selected area (%)
WOODLAND
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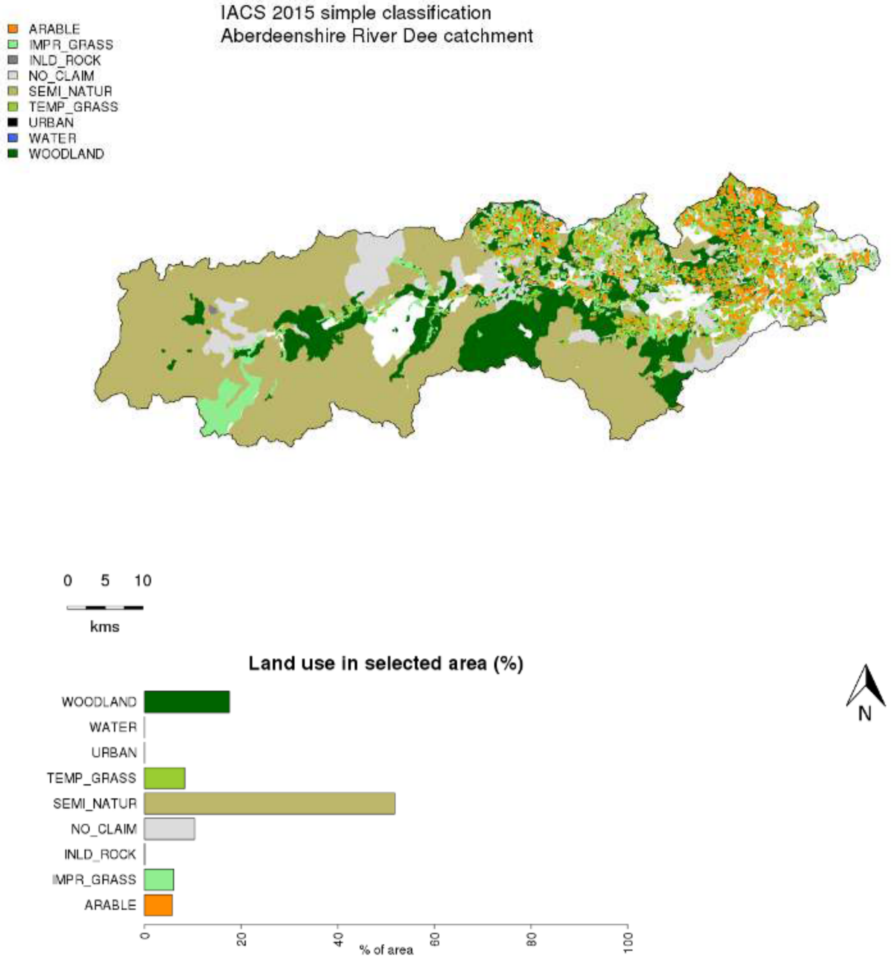
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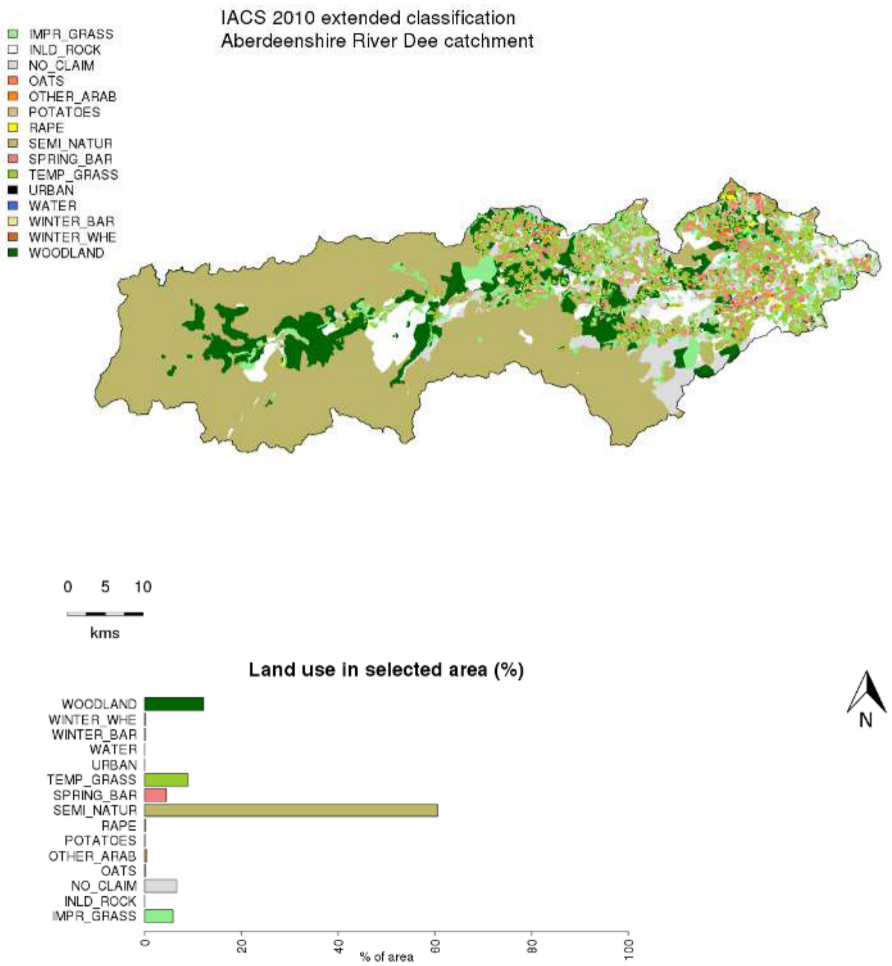
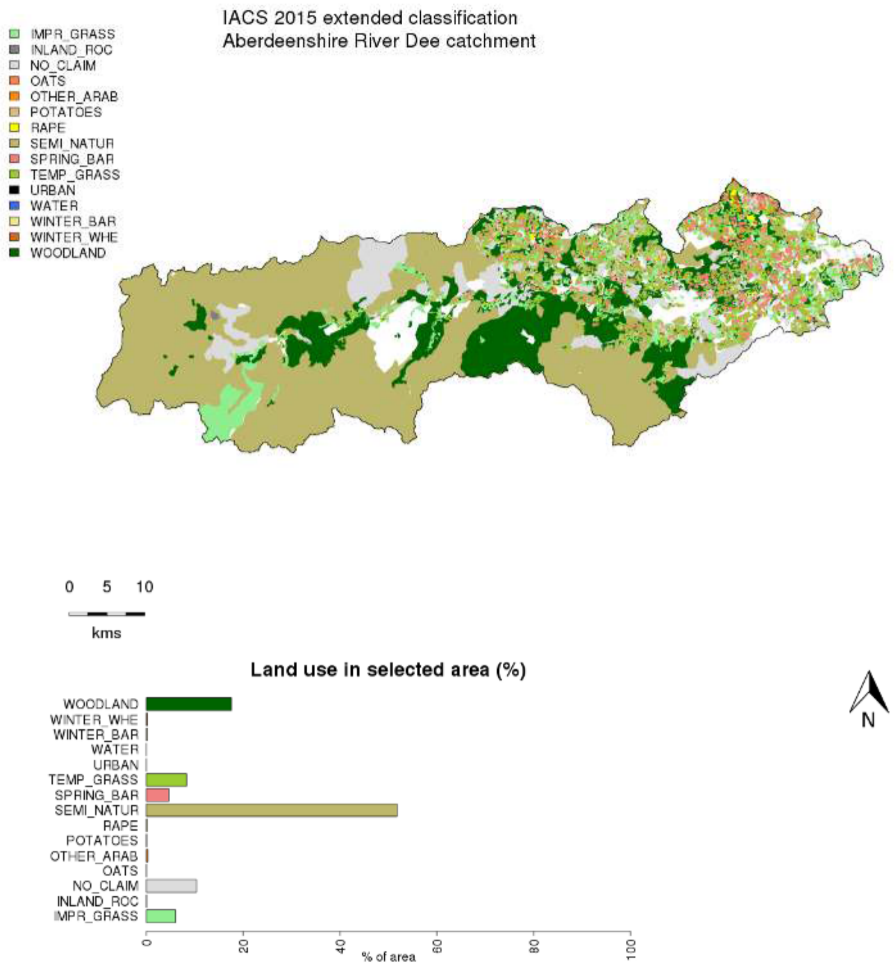
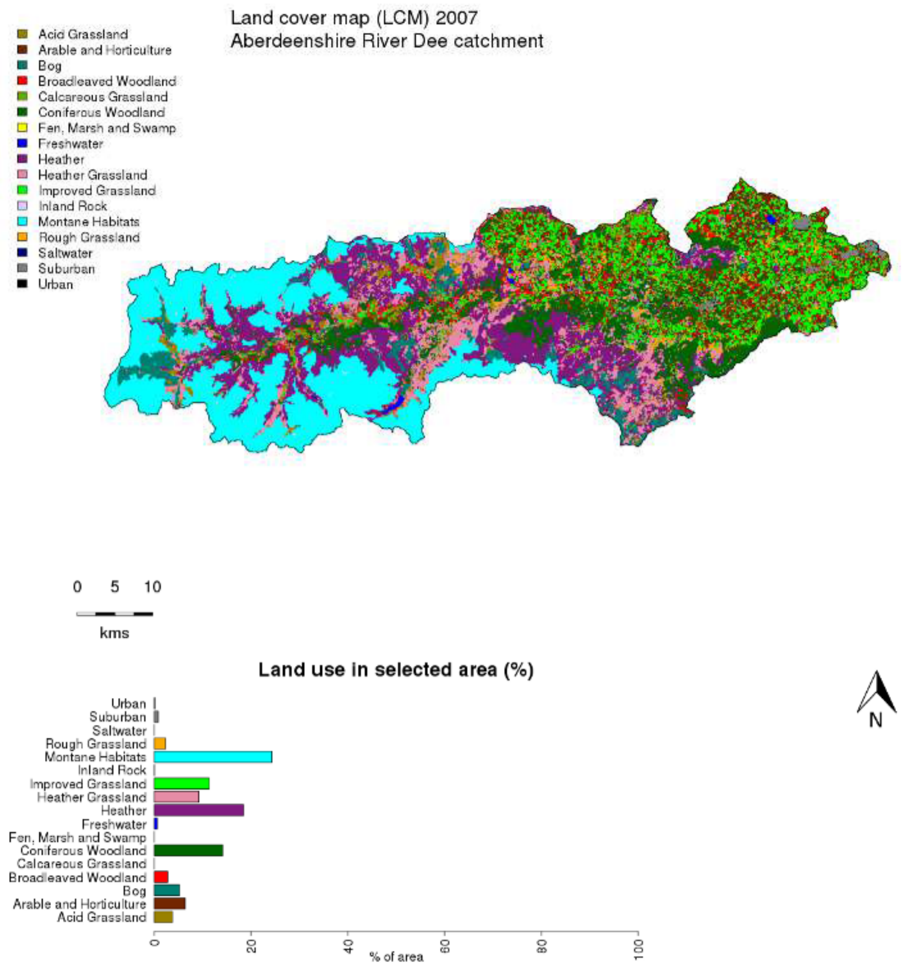
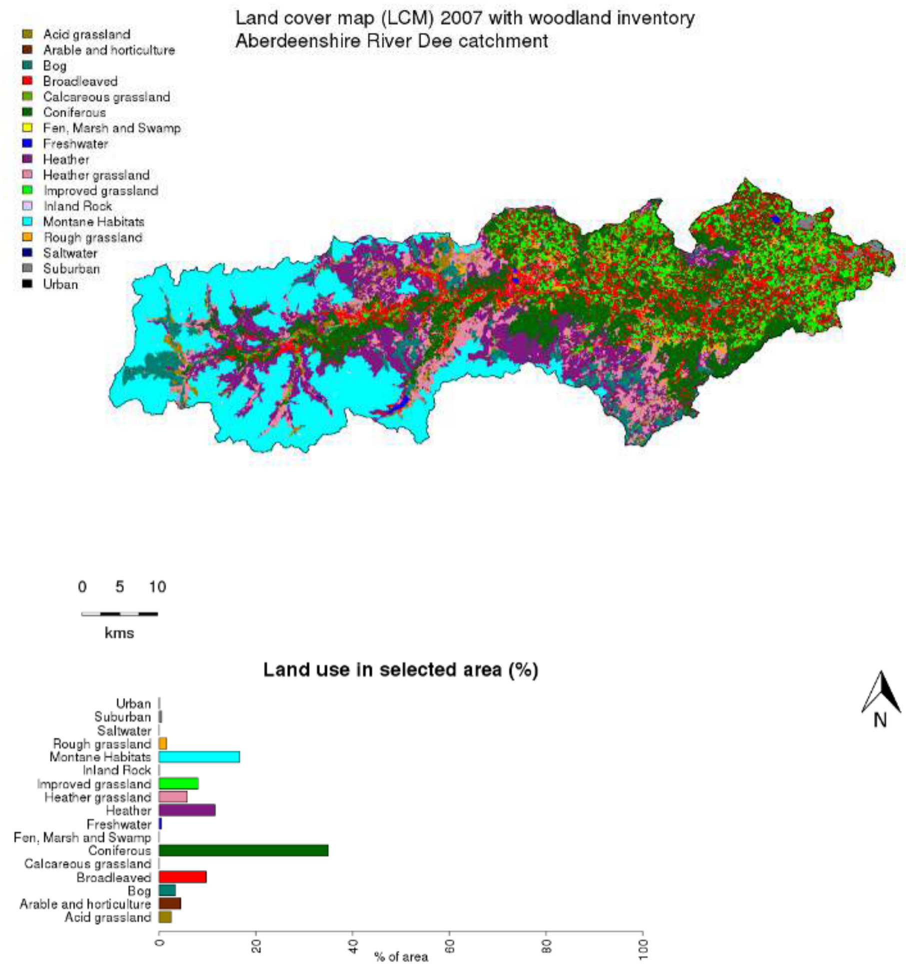

IACS 2010 extended classification
Aberdeenshire River Dee catchment
IMPR_GRASS
INLD_ROCK
NO_CLAIM
OATS
OTHER_ARAB
POTATOES
RAPE
SEMI_NATUR
SPRING_BAR
TEMP_GRASS
URBAN
WATER
WINTER_BAR
WINTER_WHE
WOODLAND
0 5 10
kms
Land use in selected area (%)
% of area
0
20
40
60
80
100
N

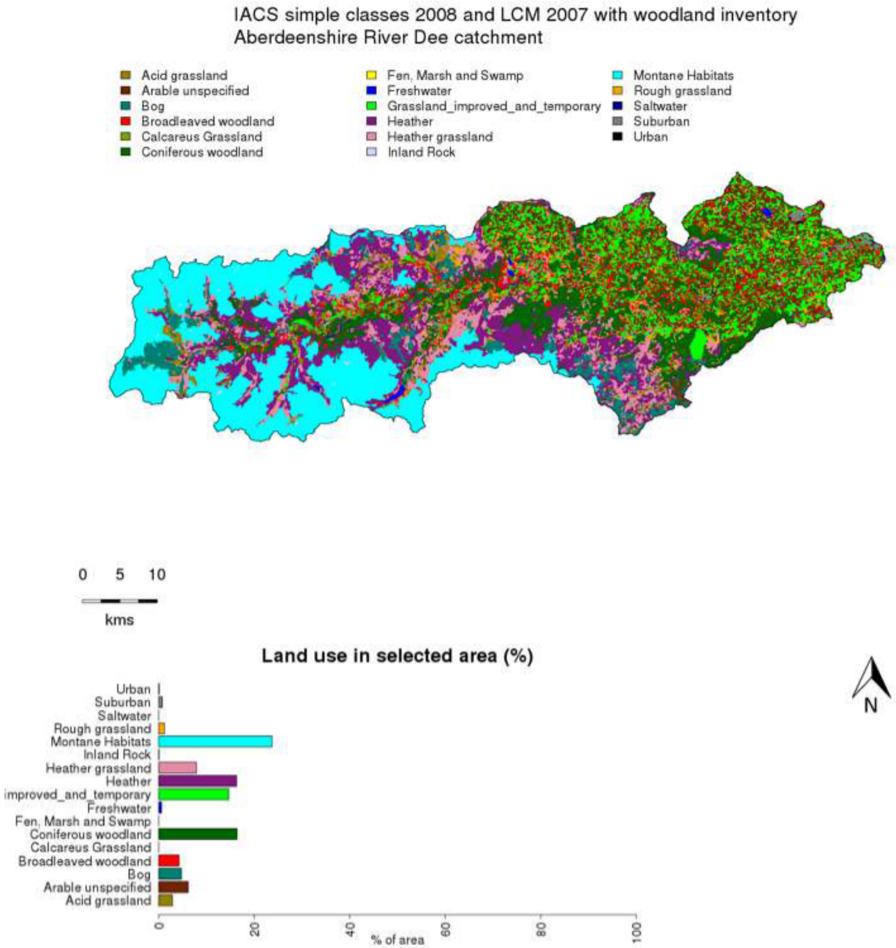

IACS 2015 extended classification
Aberdeenshire River Dee catchment
IMPR_GRASS
INLAND_ROC
NO_CLAIM
OATS
OTHER_ARAB
POTATOES
RAPE
SEMI_NATUR
SPRING_BAR
TEMP_GRASS
URBAN
WATER
WINTER_BAR
WINTER_WHE
WOODLAND
0 5 10
kms
Land use in selected area (%)
% of area
0
20
40
60
80
100
N

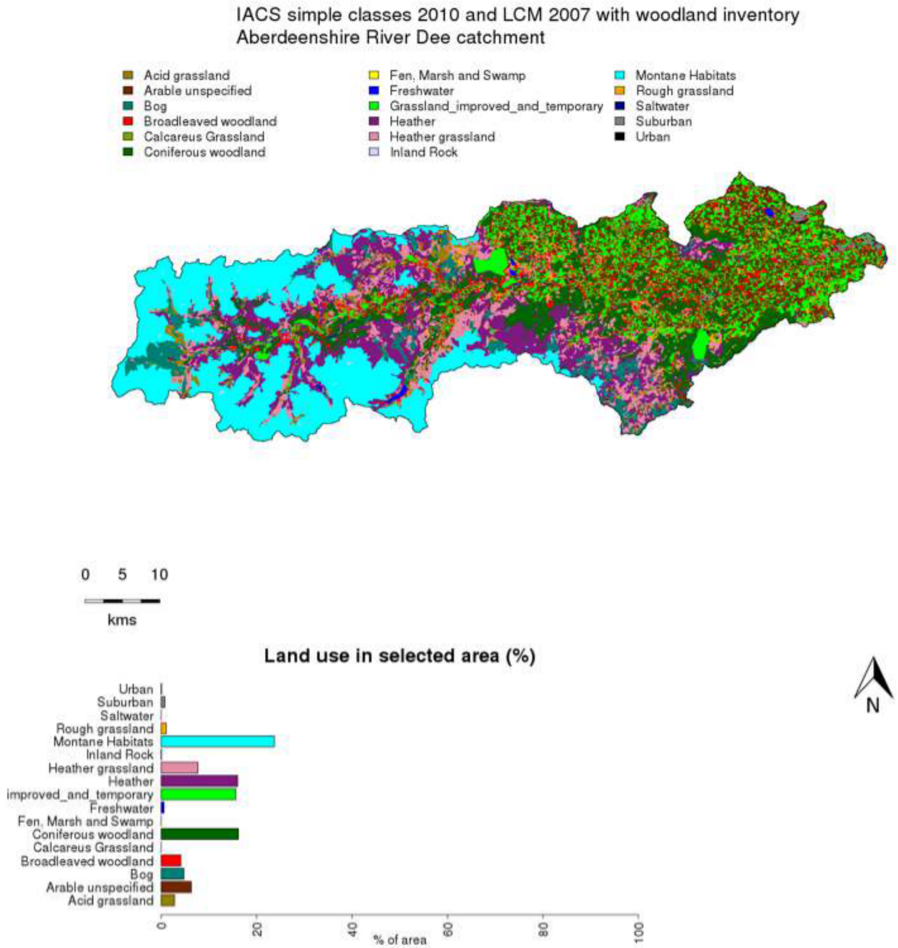


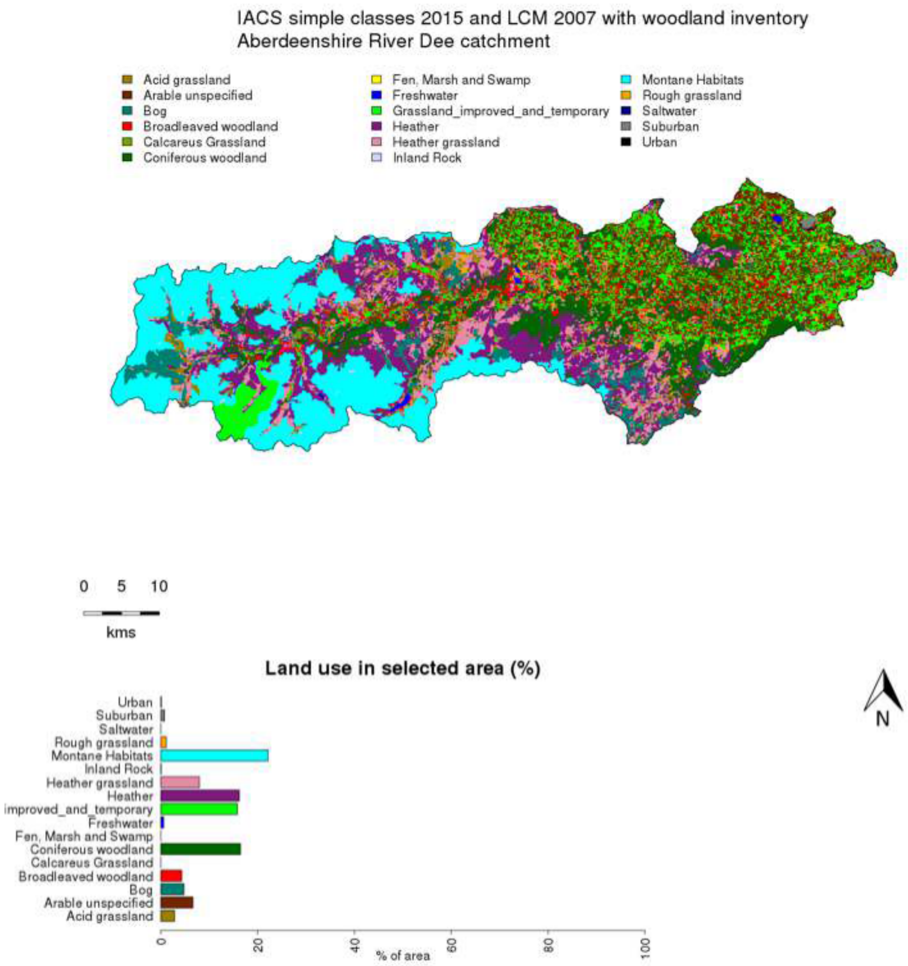
Land cover map (LCM) 2007
Aberdeenshire River Dee catchment
Acid Grassland
Arable and Horticulture
Bog
Broadleaved Woodland
Calcareous Grassland
Coniferous Woodland
Fen, Marsh and Swamp
Freshwater
Heather
Heather Grassland
Improved Grassland
Inland Rock
Montane Habitats
Rough Grassland
Saltwater
Suburban
Urban
0 5 10
kms
N
Land use in selected area (%)
Urban
Suburban
Saltwater
Rough Grassland
Montane Habitats
Inland Rock
Improved Grassland
Heather Grassland
Heather
Freshwater
Fen, Marsh and Swamp
Coniferous Woodland
Calcareous Grassland
Broadleaved Woodland
Bog
Arable and Horticulture
Acid Grassland
0 20 40 60 80 100
% of area

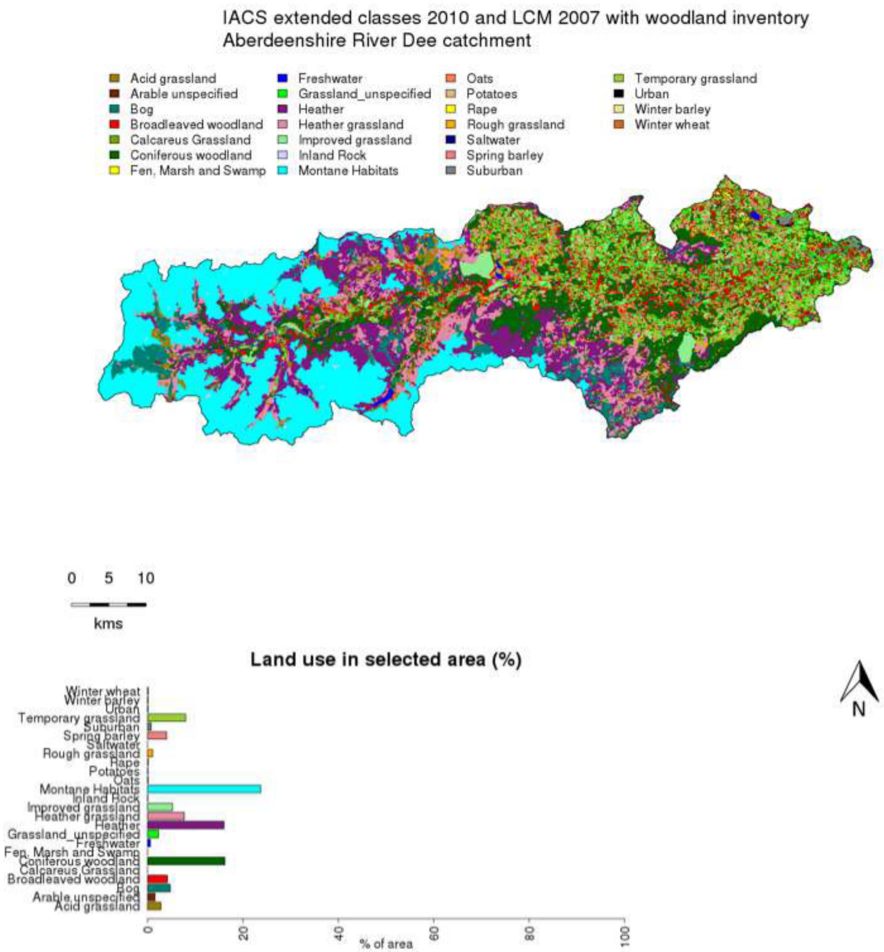


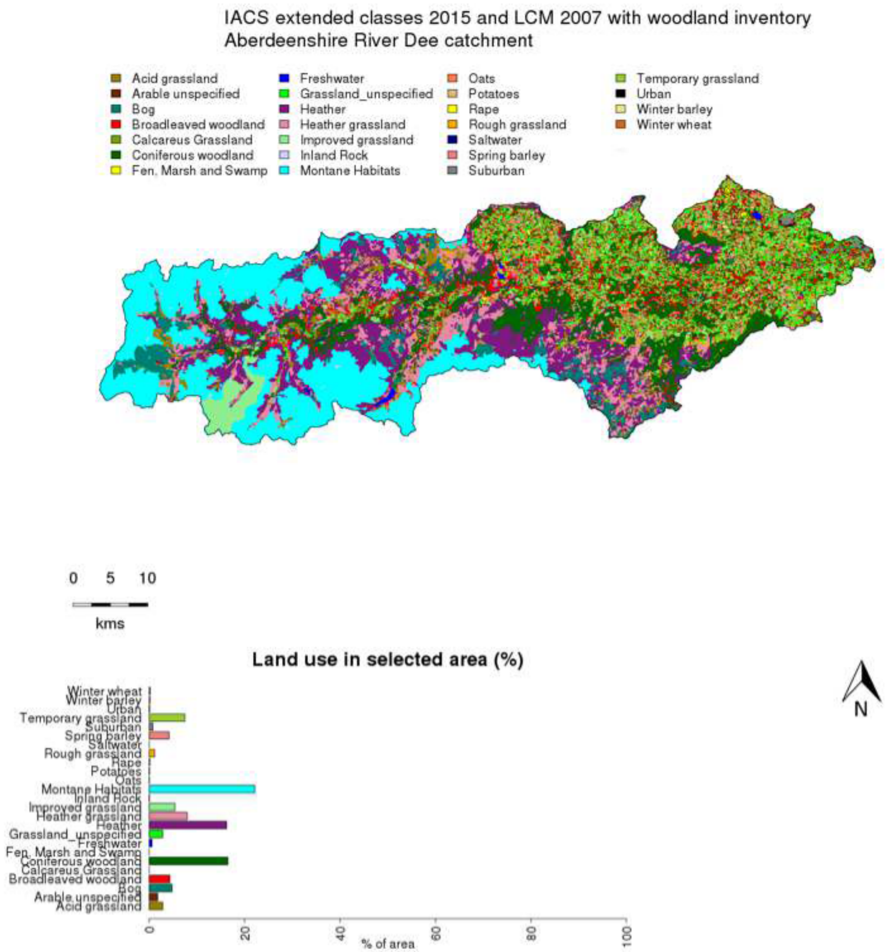
Land cover map (LCM) 2007 with woodland inventory
Aberdeenshire River Dee catchment
Acid grassland
Arable and horticulture
Bog
Broadleaved
Calcareous grassland
Coniferous
Fen, Marsh and Swamp
Freshwater
Heather
Heather grassland
Improved grassland
Inland Rock
Montane Habitats
Rough grassland
Saltwater
Suburban
Urban
0 5 10
kms
N
Land use in selected area (%)
Urban
Suburban
Saltwater
Rough grassland
Montane Habitats
Inland Rock
Improved grassland
Heather grassland
Heather
Freshwater
Fen, Marsh and Swamp
Coniferous
Calcareous grassland
Broadleaved
Bog
Arable and horticulture
Acid grassland
0 20 40 60 80 100
% of area

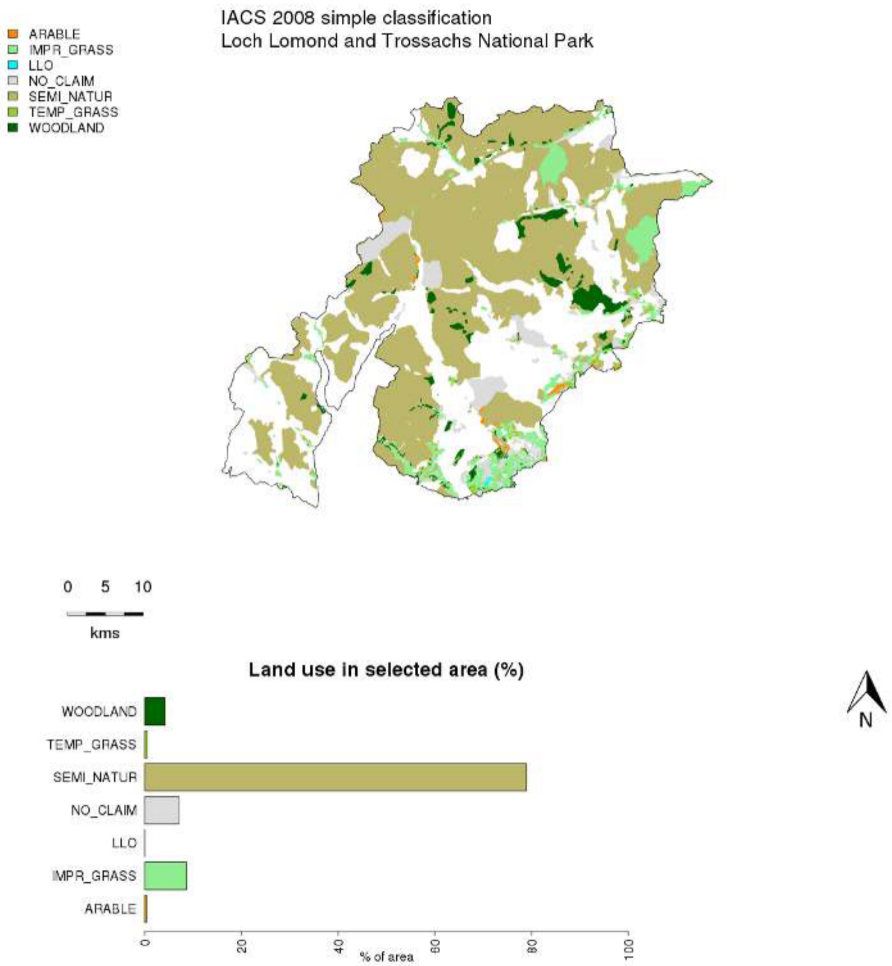

IACS simple classes 2008 and LCM 2007 with woodland inventory
Aberdeenshire River Dee catchment
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Calcareus Grassland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_improved_and_temporary
Heather
Heather grassland
Inland Rock
Montane Habitats
Rough grassland
Saltwater
Suburban
Urban
0 5 10
kms
N
Land use in selected area (%)
Urban
Suburban
Saltwater
Rough grassland
Montane Habitats
Inland Rock
Heather grassland
Heather
improved_and_temporary
Freshwater
Fen, Marsh and Swamp
Coniferous woodland
Calcareus Grassland
Broadleaved woodland
Bog
Arable unspecified
Acid grassland
0
20
40
60
80
100
% of area

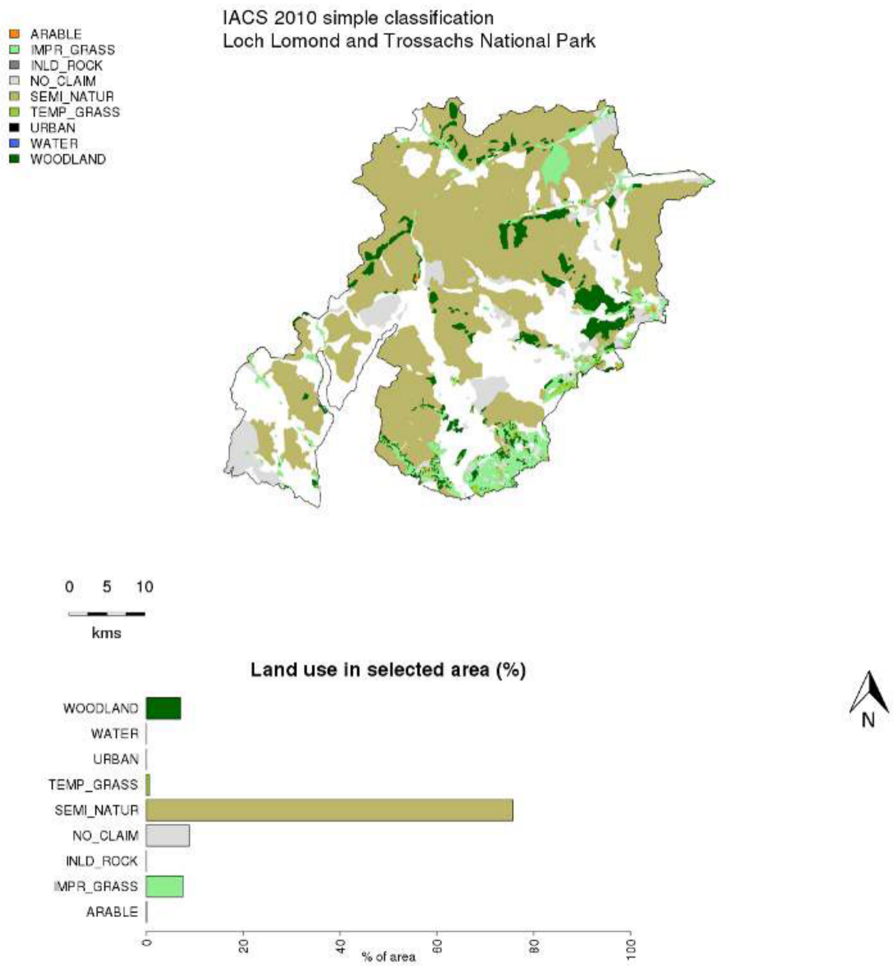

IACS simple classes 2010 and LCM 2007 with woodland inventory
Aberdeenshire River Dee catchment
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Calcareus Grassland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_improved_and_temporary
Heather
Heather grassland
Inland Rock
Montane Habitats
Rough grassland
Saltwater
Suburban
Urban
0 5 10
kms
N
Land use in selected area (%)
Urban
Suburban
Saltwater
Rough grassland
Montane Habitats
Inland Rock
Heather grassland
Heather
improved_and_temporary
Freshwater
Fen, Marsh and Swamp
Coniferous woodland
Calcareus Grassland
Broadleaved woodland
Bog
Arable unspecified
Acid grassland
0
20
40
60
80
100
% of area

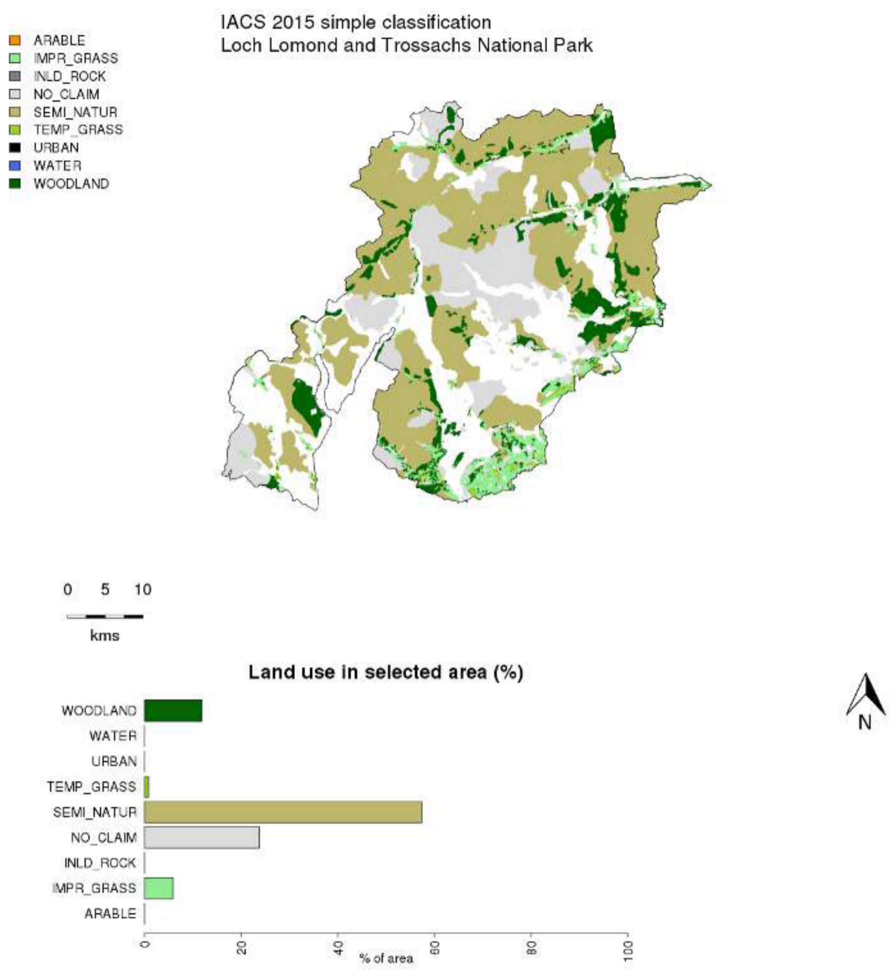

IACS simple classes 2015 and LCM 2007 with woodland inventory
Aberdeenshire River Dee catchment
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Calcareus Grassland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_improved_and_temporary
Heather
Heather grassland
Inland Rock
Montane Habitats
Rough grassland
Saltwater
Suburban
Urban
0 5 10
kms
N
Land use in selected area (%)
Urban
Suburban
Saltwater
Rough grassland
Montane Habitats
Inland Rock
Heather grassland
Heather
improved_and_temporary
Freshwater
Fen, Marsh and Swamp
Coniferous woodland
Calcareus Grassland
Broadleaved woodland
Bog
Arable unspecified
Acid grassland
0
20
40
60
80
100
% of area

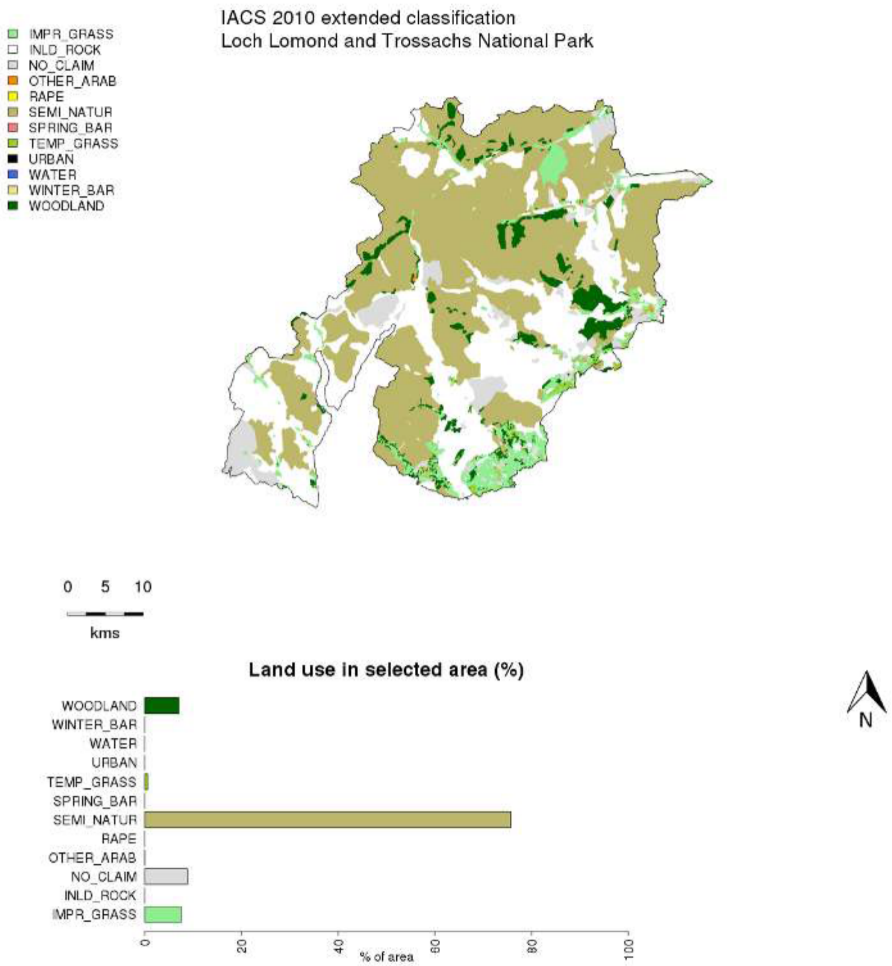

IACS extended classes 2010 and LCM 2007 with woodland inventory
Aberdeenshire River Dee catchment
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Calcareus Grassland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_unspecified
Heather
Heather grassland
Improved grassland
Inland Rock
Montane Habitats
Oats
Potatoes
Rape
Rough grassland
Saltwater
Spring barley
Suburban
Temporary grassland
Urban
Winter barley
Winter wheat
0 5 10
kms
Land use in selected area (%)
% of area
0
20
40
60
80
100
N

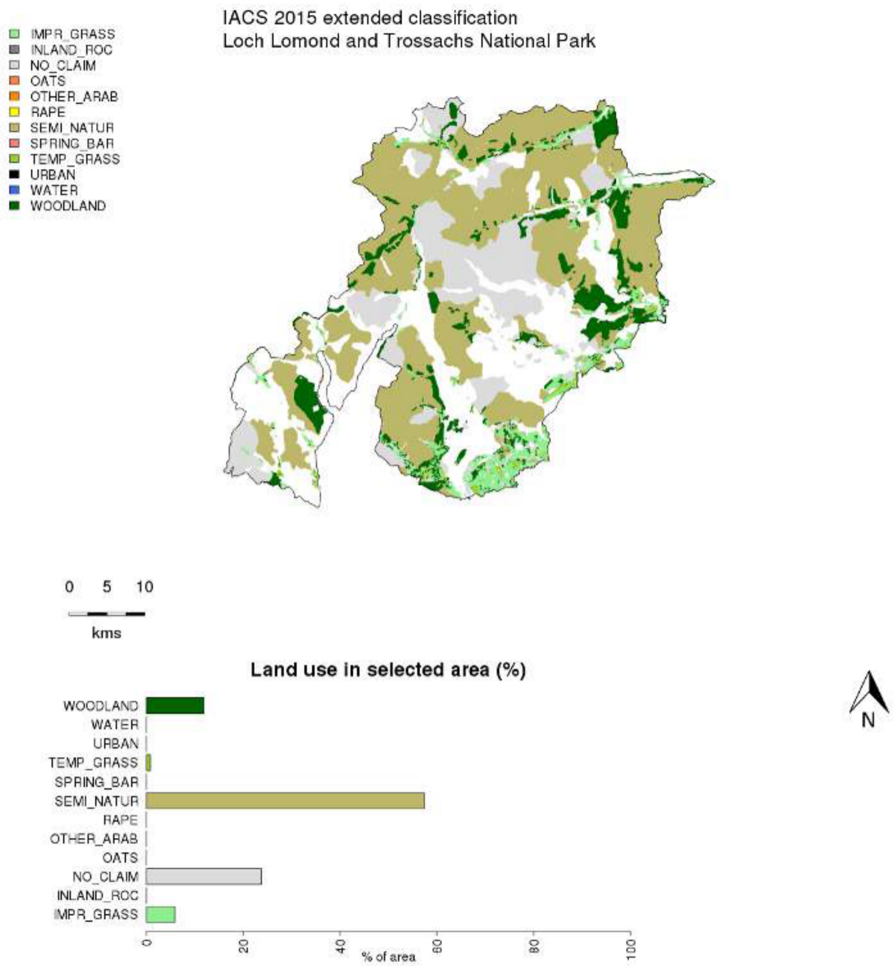

IACS extended classes 2015 and LCM 2007 with woodland inventory
Aberdeenshire River Dee catchment
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Calcareus Grassland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_unspecified
Heather
Heather grassland
Improved grassland
Inland Rock
Montane Habitats
Oats
Potatoes
Rape
Rough grassland
Saltwater
Spring barley
Suburban
Temporary grassland
Urban
Winter barley
Winter wheat
0 5 10
kms
Land use in selected area (%)
% of area
0
20
40
60
80
100
N

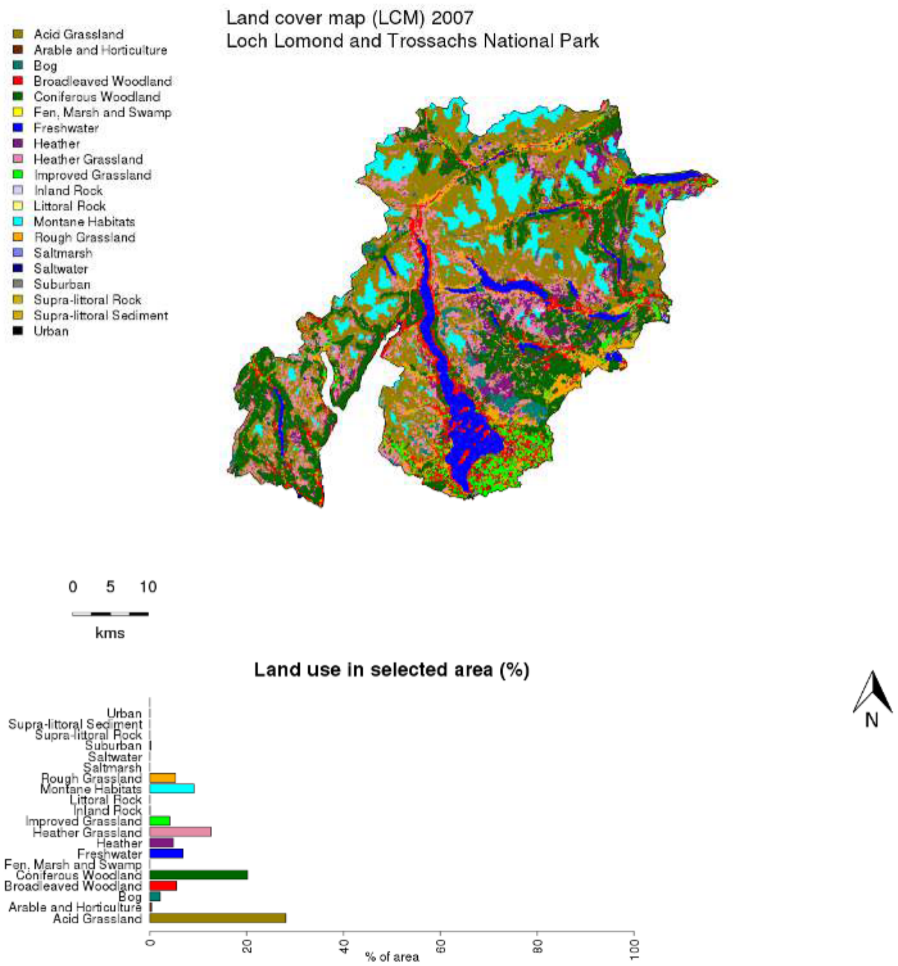

IACS 2008 simple classification
Loch Lomond and Trossachs National Park
ARABLE
IMPR_GRASS
LLO
NO_CLAIM
SEMI_NATUR
TEMP_GRASS
WOODLAND
0 5 10
kms
Land use in selected area (%)
WOODLAND
TEMP_GRASS
SEMI_NATUR
NO_CLAIM
LLO
IMPR_GRASS
ARABLE
0
20
40
60
80
100
% of area
N

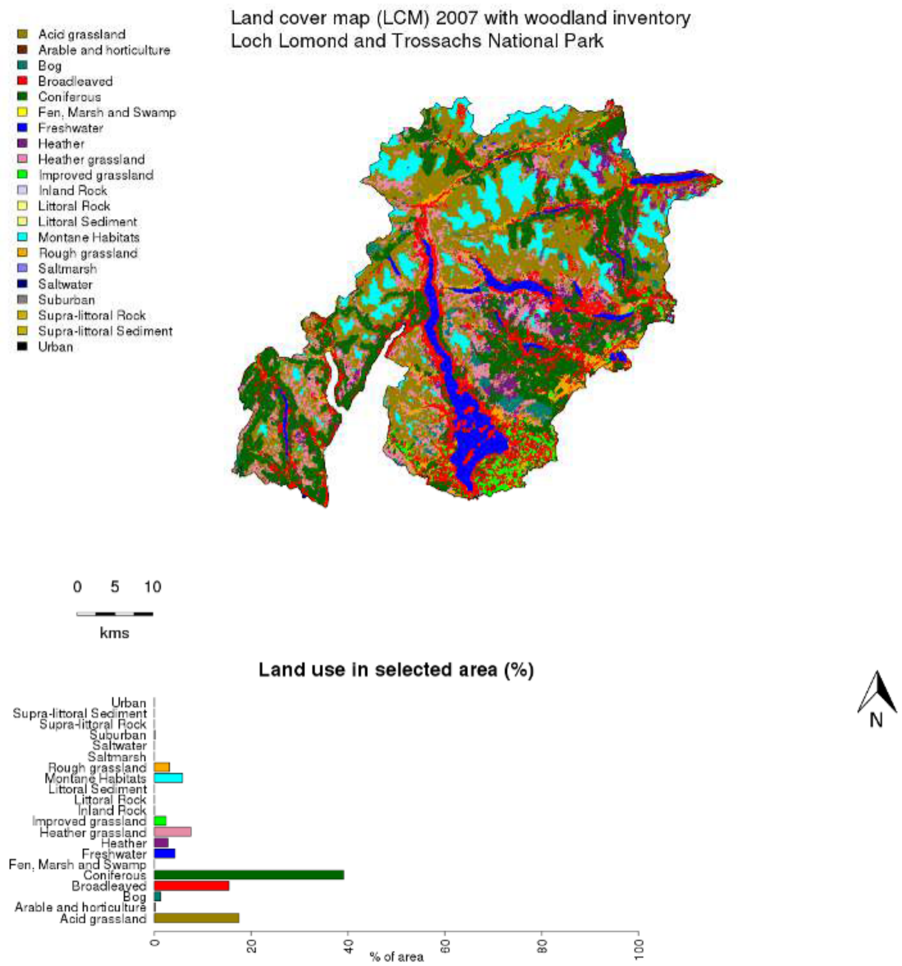

IACS 2010 simple classification
Loch Lomond and Trossachs National Park
ARABLE
IMPR_GRASS
INLD_ROCK
NO_CLAIM
SEMI_NATUR
TEMP_GRASS
URBAN
WATER
WOODLAND
0 5 10
kms
Land use in selected area (%)
WOODLAND
WATER
URBAN
TEMP_GRASS
SEMI_NATUR
NO_CLAIM
INLD_ROCK
IMPR_GRASS
ARABLE
0
20
40
60
80
100
% of area
N


IACS 2015 simple classification
Loch Lomond and Trossachs National Park
ARABLE
IMPR_GRASS
INLD_ROCK
NO_CLAIM
SEMI_NATUR
TEMP_GRASS
URBAN
WATER
WOODLAND
0 5 10
kms
N
Land use in selected area (%)
WOODLAND
WATER
URBAN
TEMP_GRASS
SEMI_NATUR
NO_CLAIM
INLD_ROCK
IMPR_GRASS
ARABLE
0
20
40
60
80
100
% of area


IACS 2010 extended classification
Loch Lomond and Trossachs National Park
IMPR_GRASS
INLD_ROCK
NO_CLAIM
OTHER_ARAB
RAPE
SEMI_NATUR
SPRING_BAR
TEMP_GRASS
URBAN
WATER
WINTER_BAR
WOODLAND
0 5 10
kms
Land use in selected area (%)
WOODLAND
WINTER_BAR
WATER
URBAN
TEMP_GRASS
SPRING_BAR
SEMI_NATUR
RAPE
OTHER_ARAB
NO_CLAIM
INLD_ROCK
IMPR_GRASS
0
20
40
60
80
100
% of area
N


IACS 2015 extended classification
Loch Lomond and Trossachs National Park
IMPR_GRASS
INLAND_ROC
NO_CLAIM
OATS
OTHER_ARAB
RAPE
SEMI_NATUR
SPRING_BAR
TEMP_GRASS
URBAN
WATER
WOODLAND
0 5 10
kms
Land use in selected area (%)
WOODLAND
WATER
URBAN
TEMP_GRASS
SPRING_BAR
SEMI_NATUR
RAPE
OTHER_ARAB
OATS
NO_CLAIM
INLAND_ROC
IMPR_GRASS
0
20
40
60
80
100
% of area
N


Land cover map (LCM) 2007
Loch Lomond and Trossachs National Park
Acid Grassland
Arable and Horticulture
Bog
Broadleaved Woodland
Coniferous Woodland
Fen, Marsh and Swamp
Freshwater
Heather
Heather Grassland
Improved Grassland
Inland Rock
Littoral Rock
Montane Habitats
Rough Grassland
Saltmarsh
Saltwater
Suburban
Supra-littoral Rock
Supra-littoral Sediment
Urban
0 5 10
kms
N
Land use in selected area (%)
% of area
0
20
40
60
80
100


Land cover map (LCM) 2007 with woodland inventory
Loch Lomond and Trossachs National Park
Acid grassland
Arable and horticulture
Bog
Broadleaved
Coniferous
Fen, Marsh and Swamp
Freshwater
Heather
Heather grassland
Improved grassland
Inland Rock
Littoral Rock
Littoral Sediment
Montane Habitats
Rough grassland
Saltmarsh
Saltwater
Suburban
Supra-littoral Rock
Supra-littoral Sediment
Urban
0 5 10
kms
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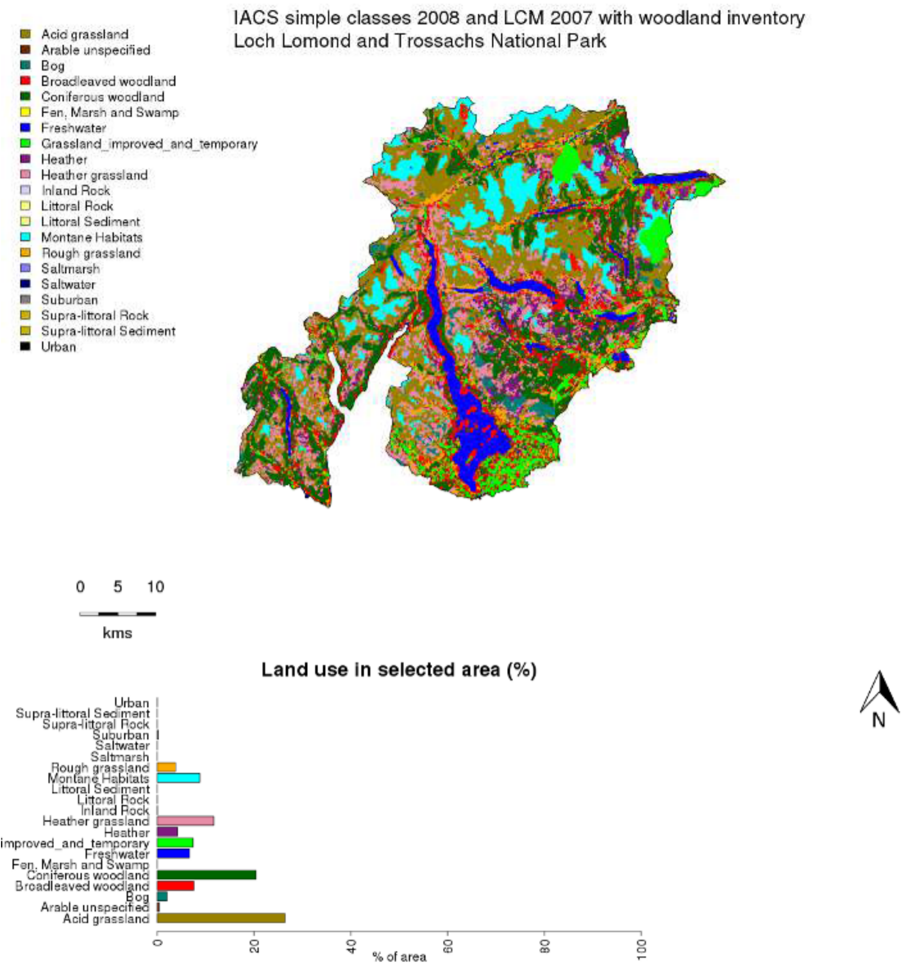
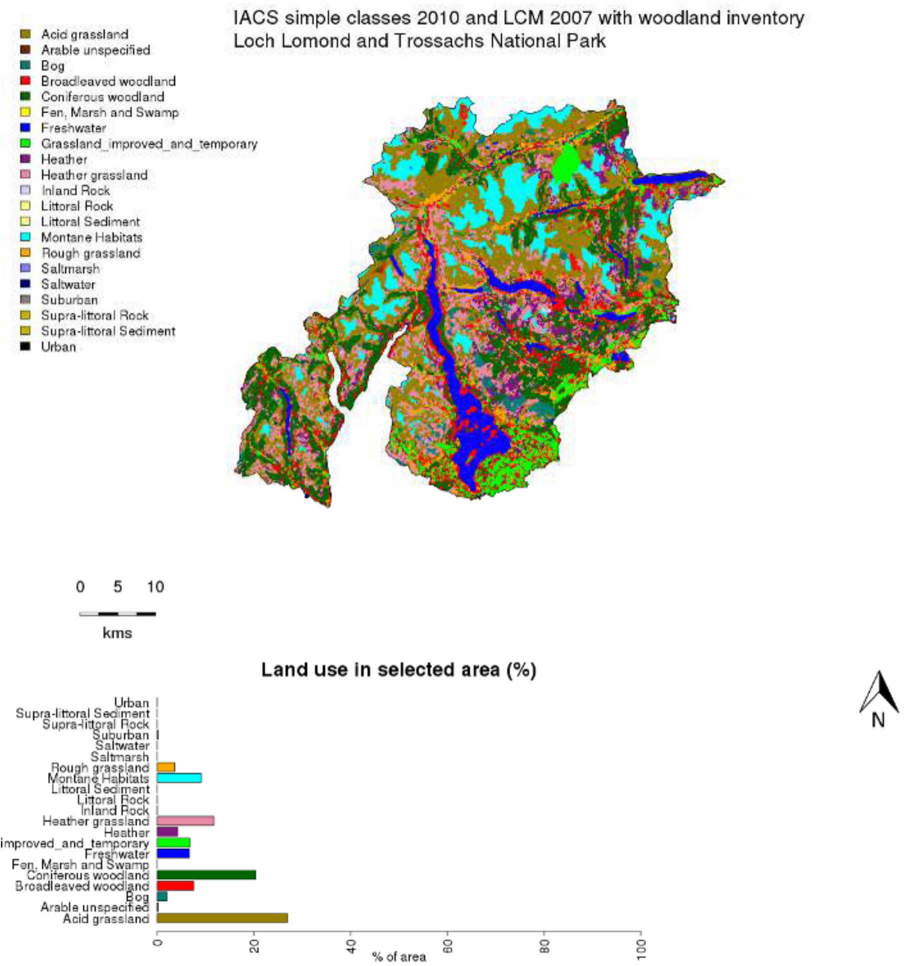
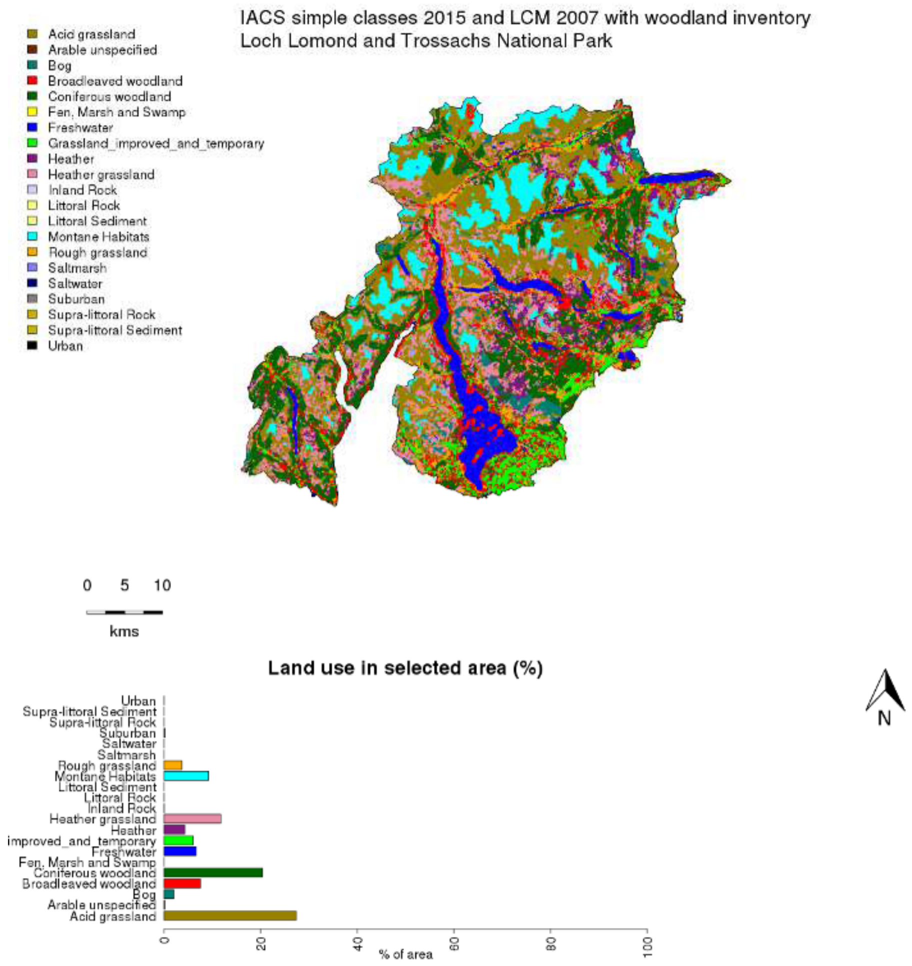
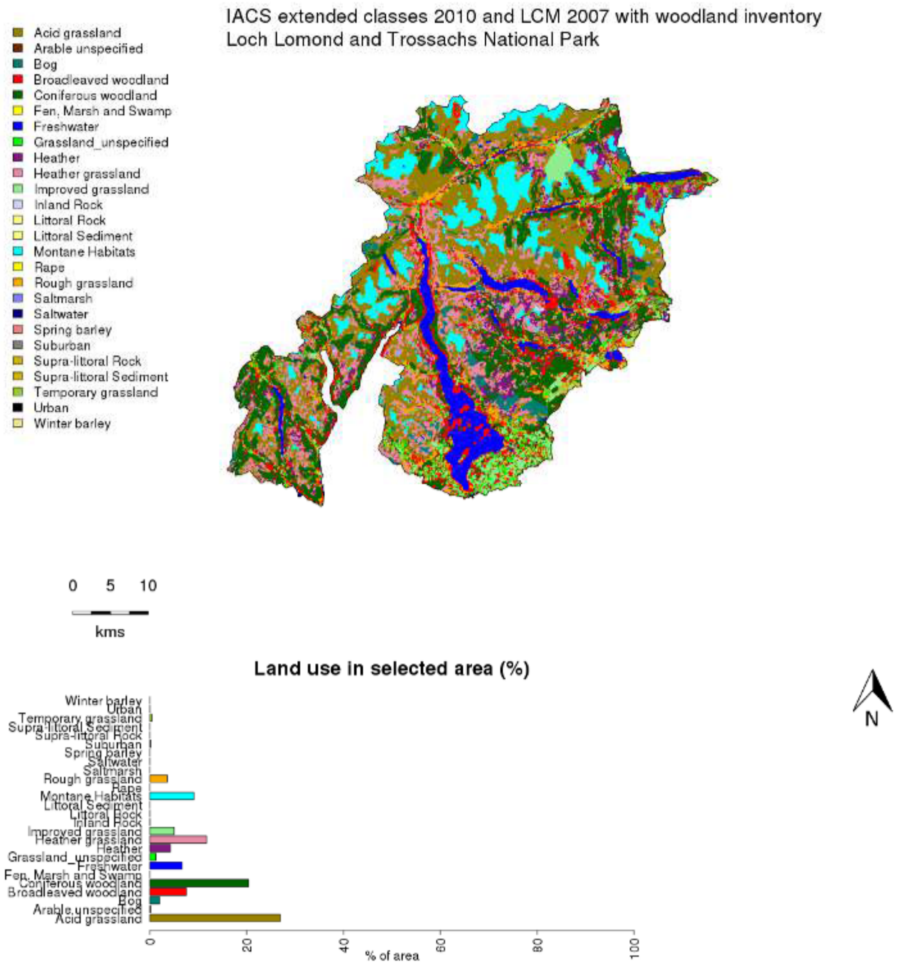
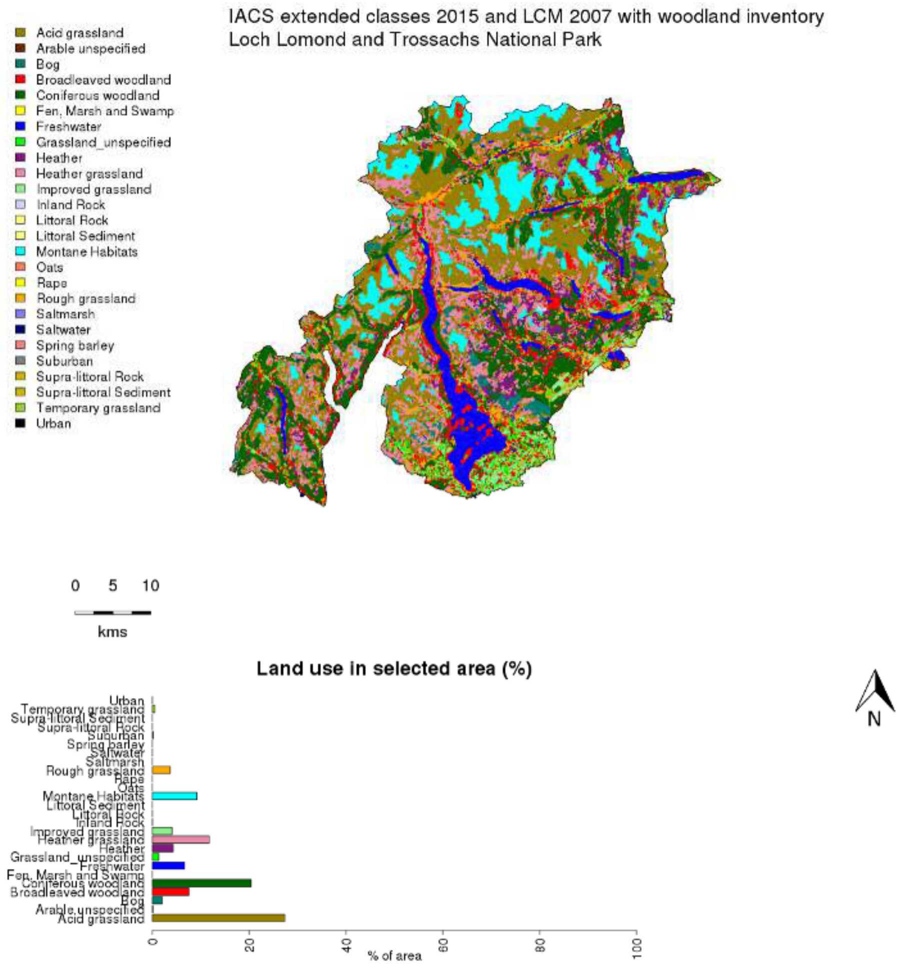
Land use in selected area (%)
% of area
0
20
40
60
80
100


IACS simple classes 2008 and LCM 2007 with woodland inventory
Loch Lomond and Trossachs National Park
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_improved_and_temporary
Heather
Heather grassland
Inland Rock
Littoral Rock
Littoral Sediment
Montane Habitats
Rough grassland
Saltmarsh
Saltwater
Suburban
Supra-littoral Rock
Supra-littoral Sediment
Urban
0 5 10
kms
Land use in selected area (%)
Urban
Supra-littoral Sediment
Supra-littoral Rock
Suburban
Saltwater
Saltmarsh
Rough grassland
Montane Habitats
Littoral Sediment
Littoral Rock
Inland Rock
Heather grassland
Heather
improved_and_temporary
Freshwater
Fen, Marsh and Swamp
Coniferous woodland
Broadleaved woodland
Bog
Arable unspecified
Acid grassland
0 20 40 60 80 100
% of area
N


IACS simple classes 2010 and LCM 2007 with woodland inventory
Loch Lomond and Trossachs National Park
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_improved_and_temporary
Heather
Heather grassland
Inland Rock
Littoral Rock
Littoral Sediment
Montane Habitats
Rough grassland
Saltmarsh
Saltwater
Suburban
Supra-littoral Rock
Supra-littoral Sediment
Urban
0 5 10
kms
Land use in selected area (%)
Urban
Supra-littoral Sediment
Supra-littoral Rock
Suburban
Saltwater
Saltmarsh
Rough grassland
Montane Habitats
Littoral Sediment
Littoral Rock
Inland Rock
Heather grassland
Heather
improved_and_temporary
Freshwater
Fen, Marsh and Swamp
Coniferous woodland
Broadleaved woodland
Bog
Arable unspecified
Acid grassland
0 20 40 60 80 100
% of area
N


IACS simple classes 2015 and LCM 2007 with woodland inventory
Loch Lomond and Trossachs National Park
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_improved_and_temporary
Heather
Heather grassland
Inland Rock
Littoral Rock
Littoral Sediment
Montane Habitats
Rough grassland
Saltmarsh
Saltwater
Suburban
Supra-littoral Rock
Supra-littoral Sediment
Urban
0 5 10
kms
Land use in selected area (%)
Urban
Supra-littoral Sediment
Supra-littoral Rock
Suburban
Saltwater
Saltmarsh
Rough grassland
Montane Habitats
Littoral Sediment
Littoral Rock
Inland Rock
Heather grassland
Heather
improved_and_temporary
Freshwater
Fen, Marsh and Swamp
Coniferous woodland
Broadleaved woodland
Bog
Arable unspecified
Acid grassland
0 20 40 60 80 100
% of area
N


IACS extended classes 2010 and LCM 2007 with woodland inventory
Loch Lomond and Trossachs National Park
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
Grassland_unspecified
Heather
Heather grassland
Improved grassland
Inland Rock
Littoral Rock
Littoral Sediment
Montane Habitats
Rape
Rough grassland
Saltmarsh
Saltwater
Spring barley
Suburban
Supra-littoral Rock
Supra-littoral Sediment
Temporary grassland
Urban
Winter barley
0 5 10
kms
N
Land use in selected area (%)
% of area
0
20
40
60
80
100


IACS extended classes 2015 and LCM 2007 with woodland inventory
Loch Lomond and Trossachs National Park
Acid grassland
Arable unspecified
Bog
Broadleaved woodland
Coniferous woodland
Fen, Marsh and Swamp
Freshwater
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Heather grassland
Improved grassland
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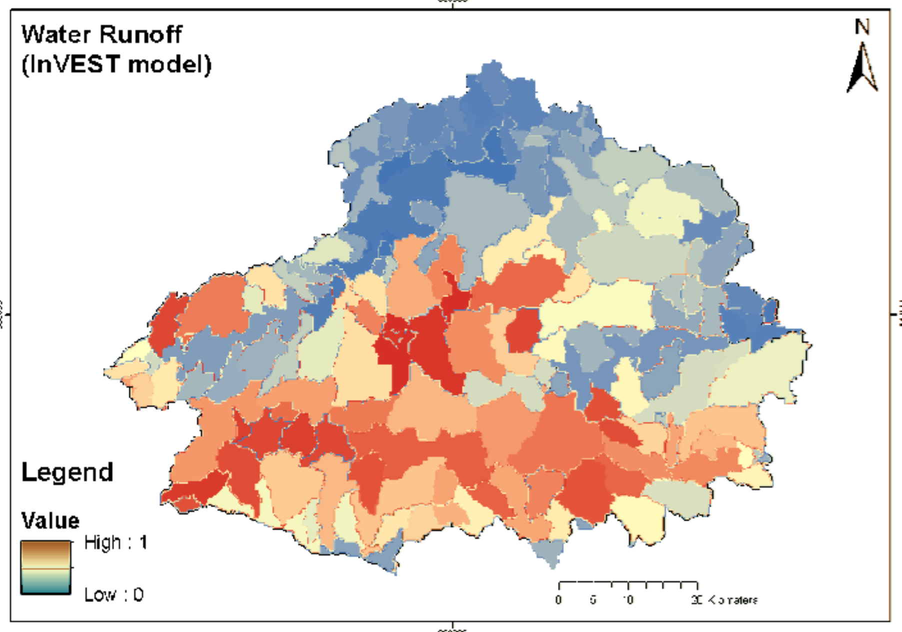
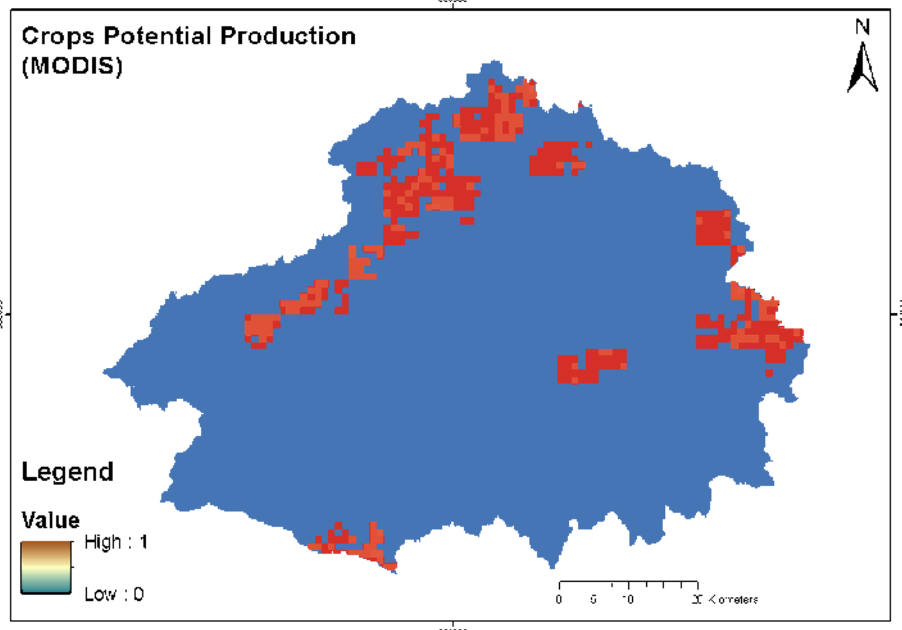
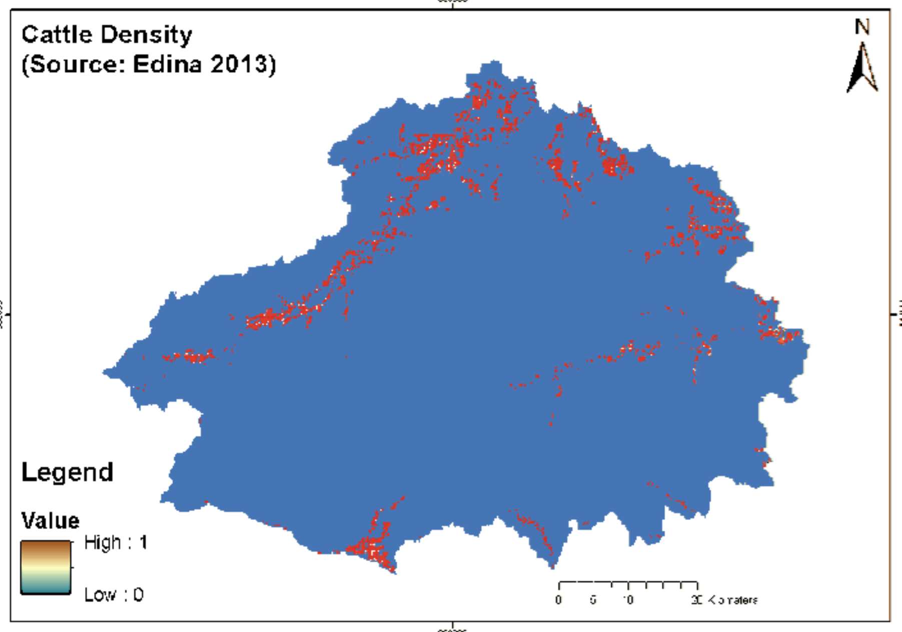
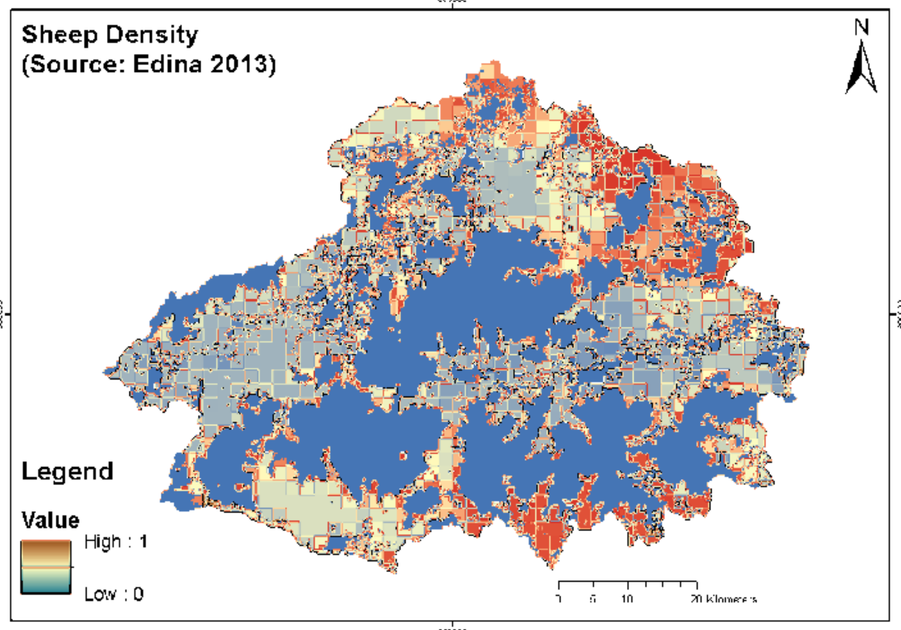
Littoral Rock
Littoral Sediment
Montane Habitats
Oats
Rape
Rough grassland
Saltmarsh
Saltwater
Spring barley
Suburban
Supra-littoral Rock
Supra-littoral Sediment
Temporary grassland
Urban
0 5 10
kms
N
Land use in selected area (%)
% of area
0
20
40
60
80
100

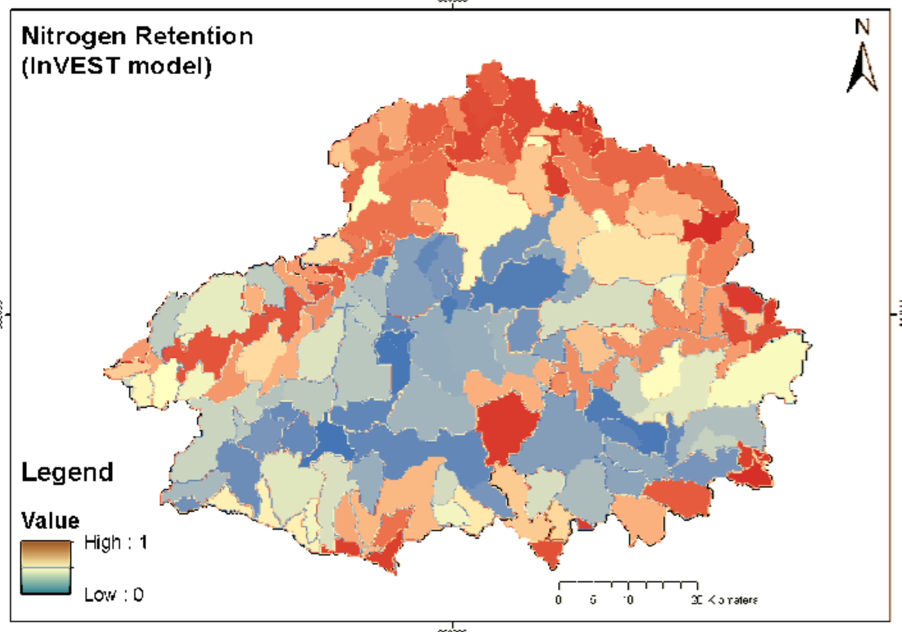
Appendix 2: Maps of ecosystems services

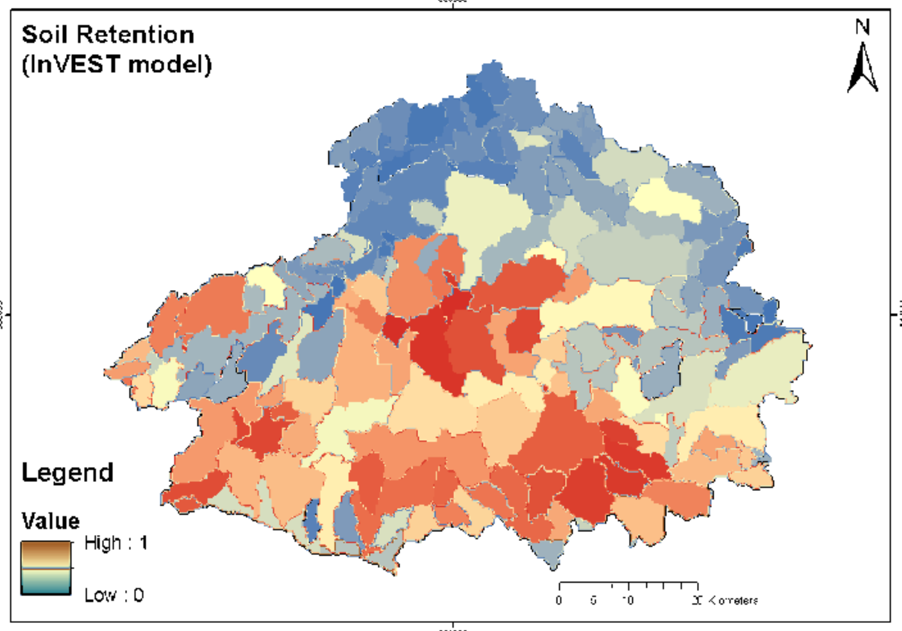

Water Runoff
(InVEST model)
N
Legend
Value
High : 1
Low : 0

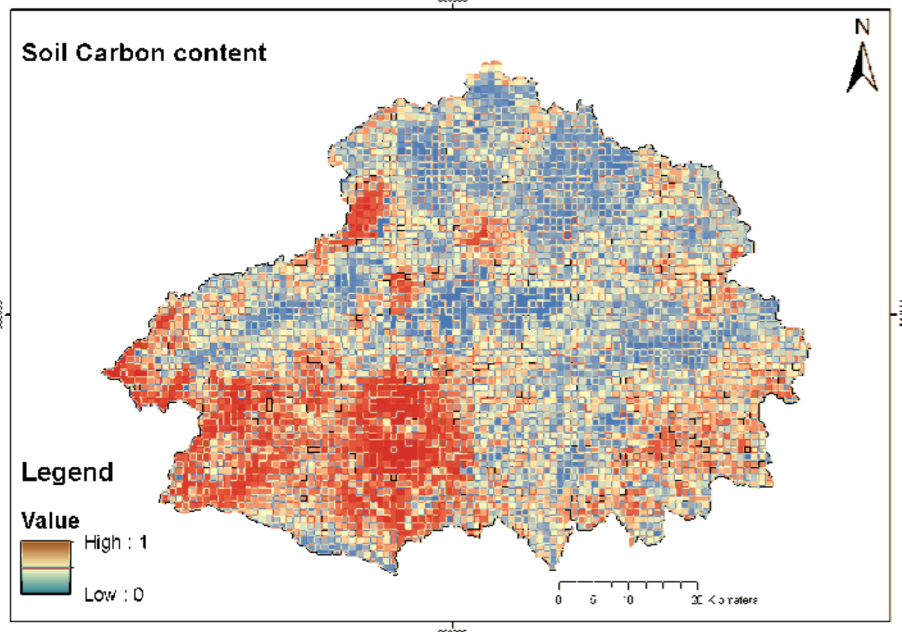

Crops Potential Production
(MODIS)
N
Legend
Value
High : 1
Low : 0

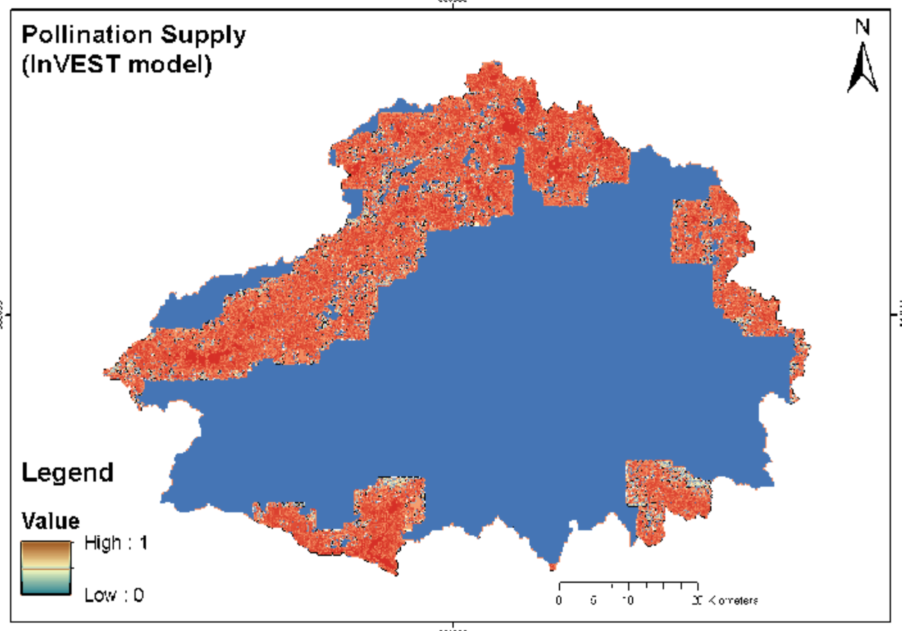

Cattle Density
(Source: Edina 2013)
N
Legend
Value
High : 1
Low : 0

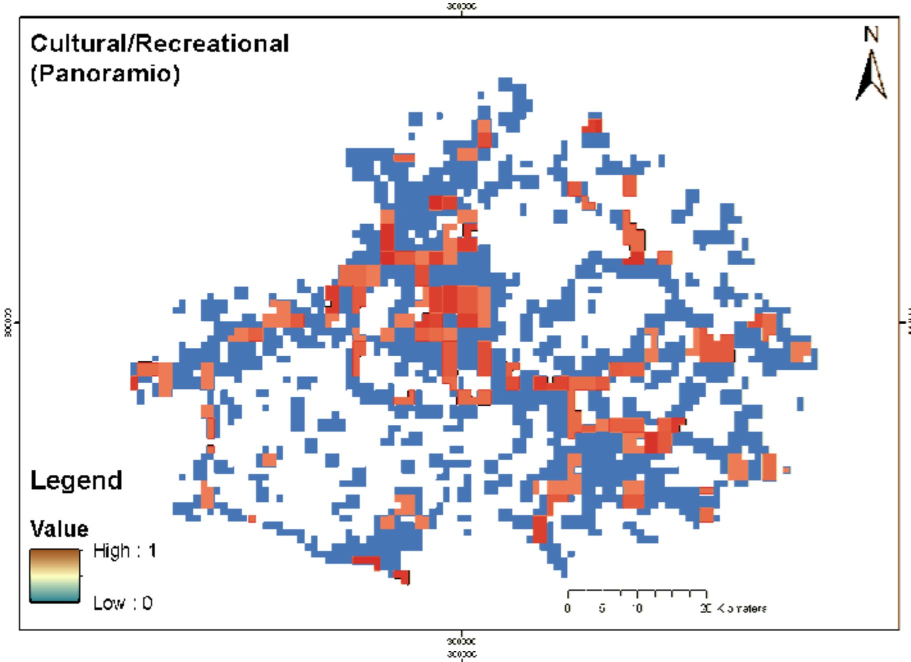

Sheep Density
(Source: Edina 2013)
N
Legend
Value
High : 1
Low : 0


Nitrogen Retention
(InVEST model)
N
Legend
Value
High : 1
Low : 0

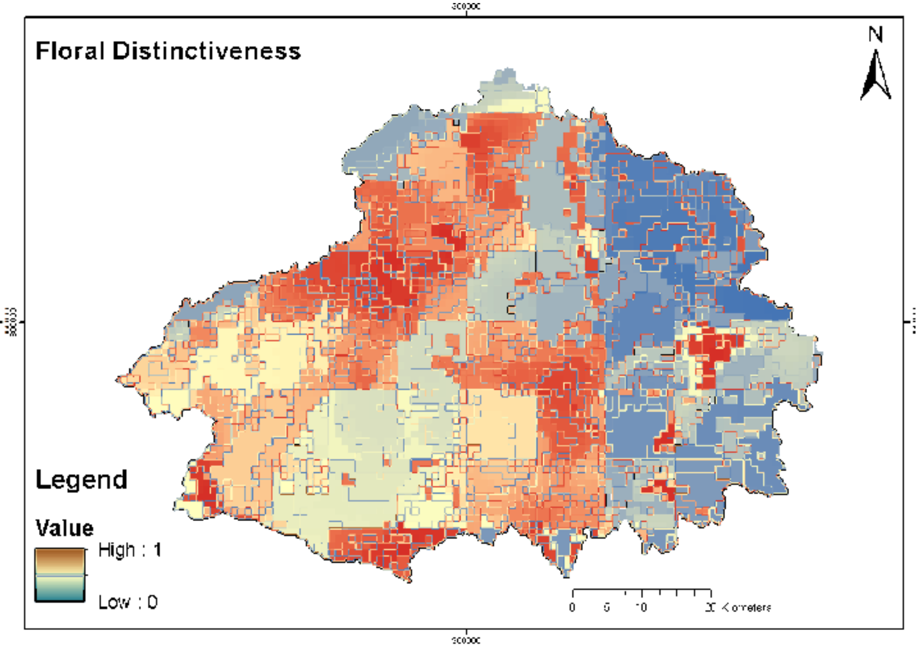

Soil Retention
(InVEST model)
N
Legend
Value
High : 1
Low : 0


Soil Carbon content
N
Legend
Value
High : 1
Low : 0

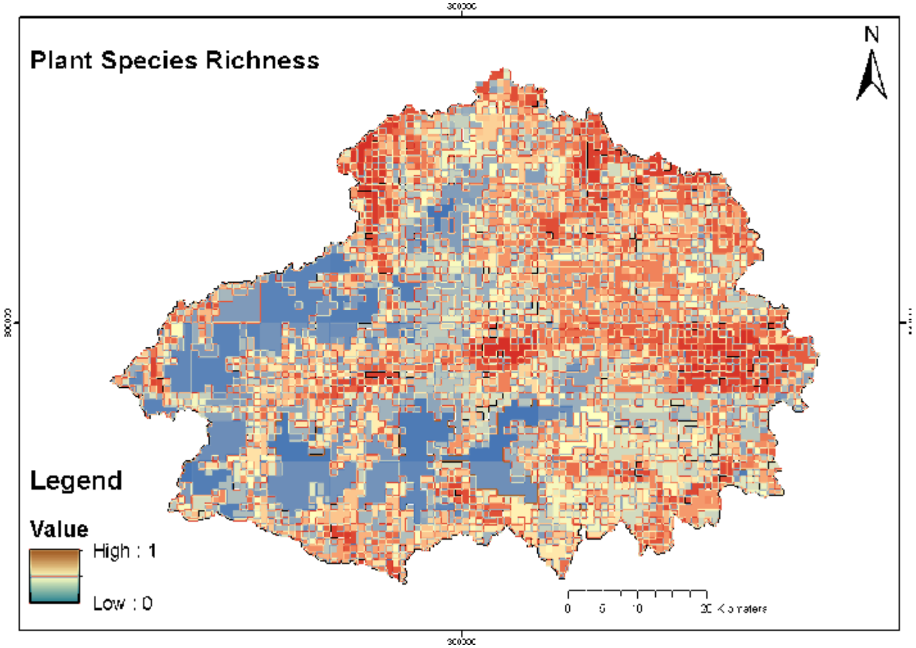

Pollination Supply
(InVEST model)
N
Legend
Value
High : 1
Low : 0



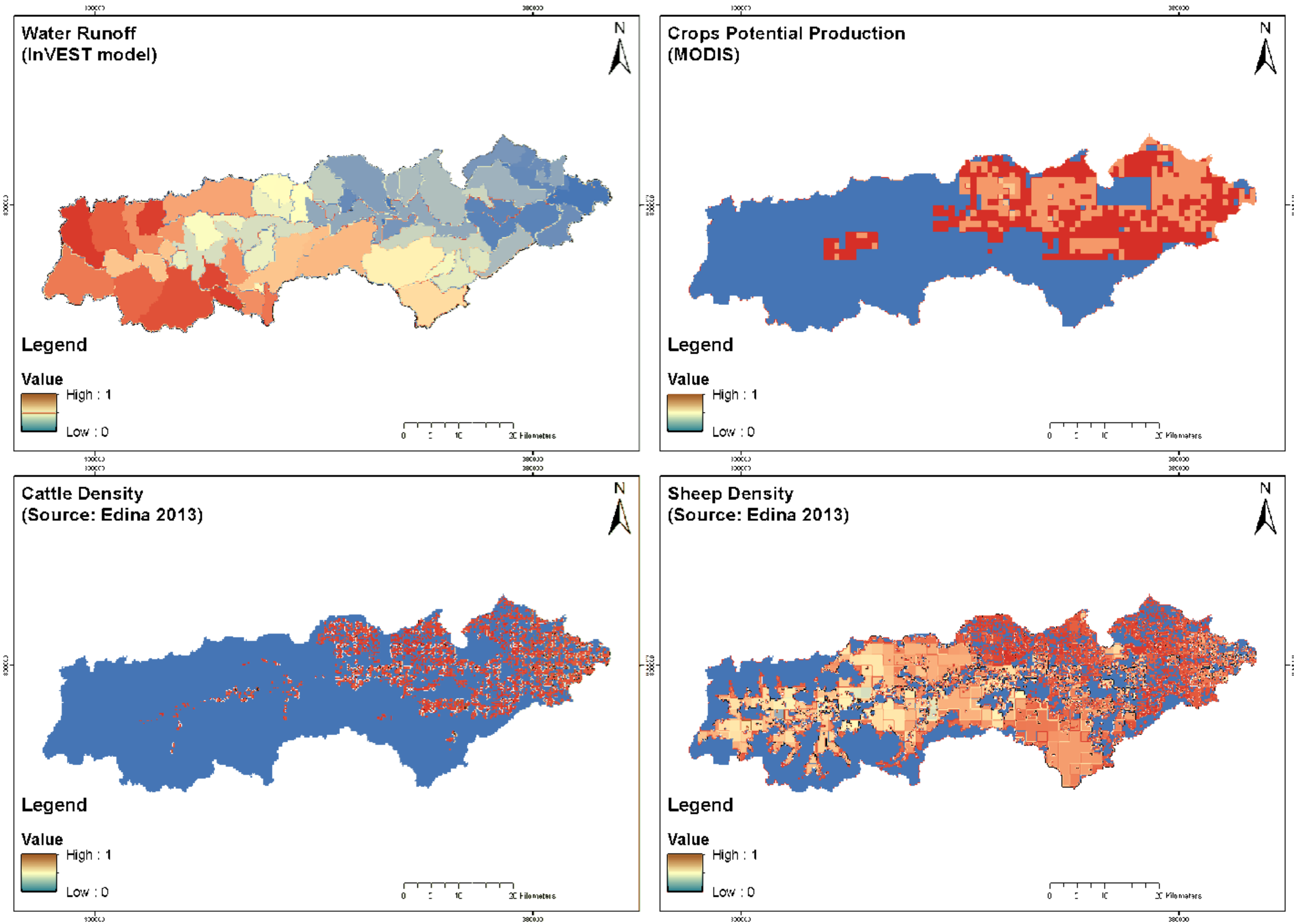
Cultural/Recreational
(Panoramio)
N
Legend
Value
High : 1
Low : 0

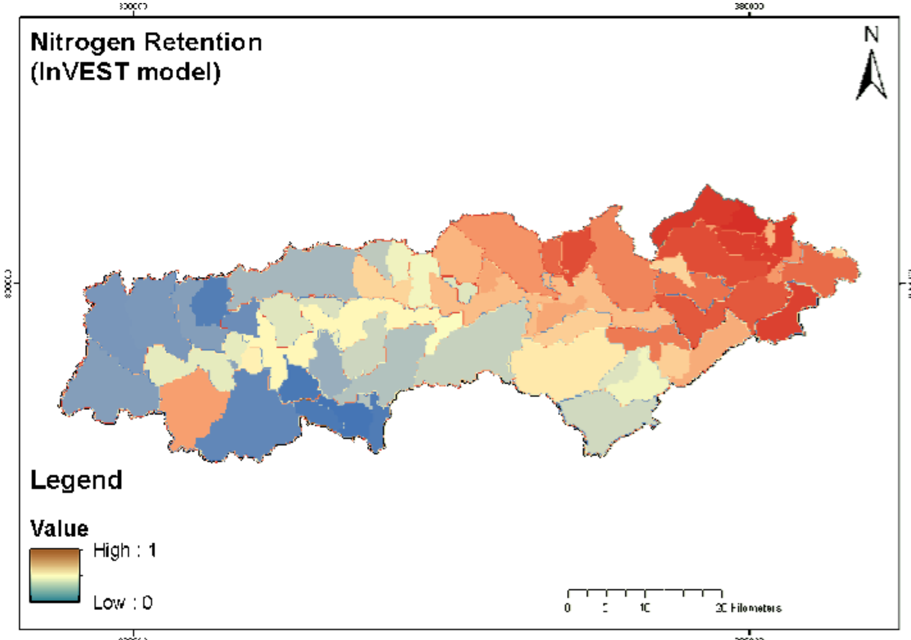


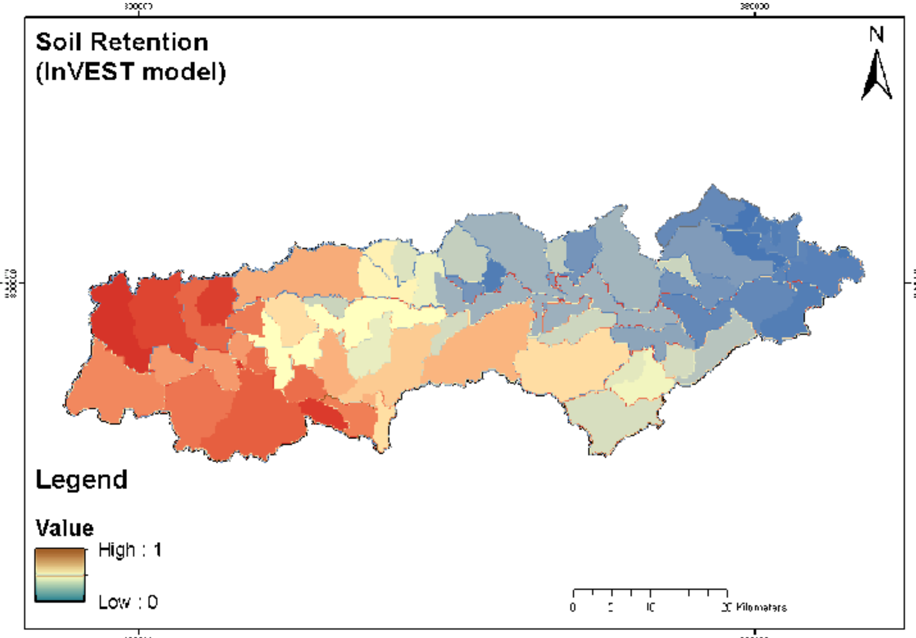
Floral Distinctiveness
N
Legend
Value
High : 1
Low : 0

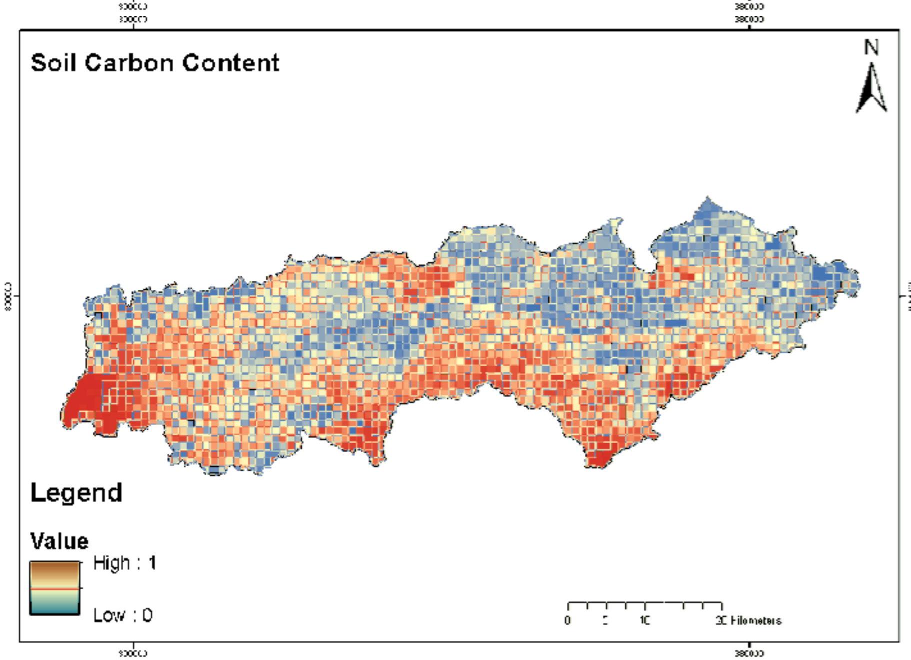


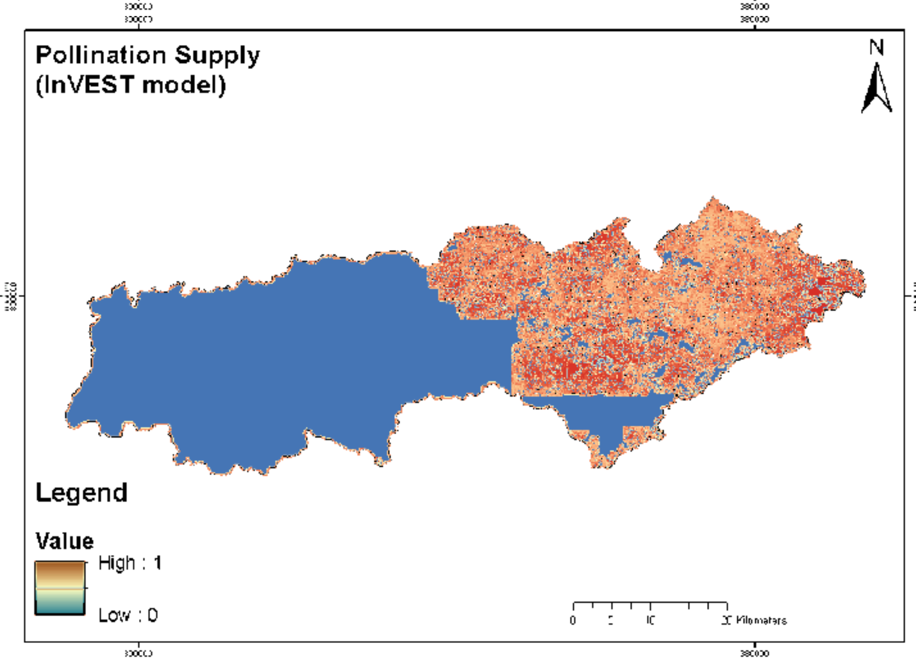
Plant Species Richness
N
Legend
Value
High : 1
Low : 0

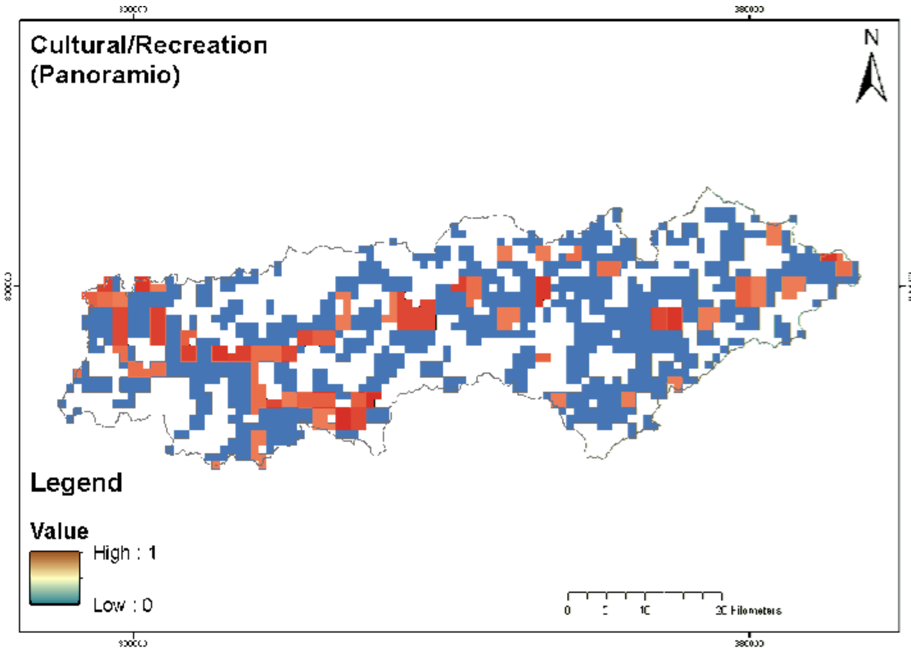

Water Runoff
(InVEST model)
N
Legend
Value
High : 1
Low : 0
Crops Potential Production
(MODIS)
N
Legend
Value
High : 1
Low : 0
Cattle Density
(Source: Edina 2013)
N
Legend
Value
High : 1
Low : 0
Sheep Density
(Source: Edina 2013)
N
Legend
Value
High : 1
Low : 0

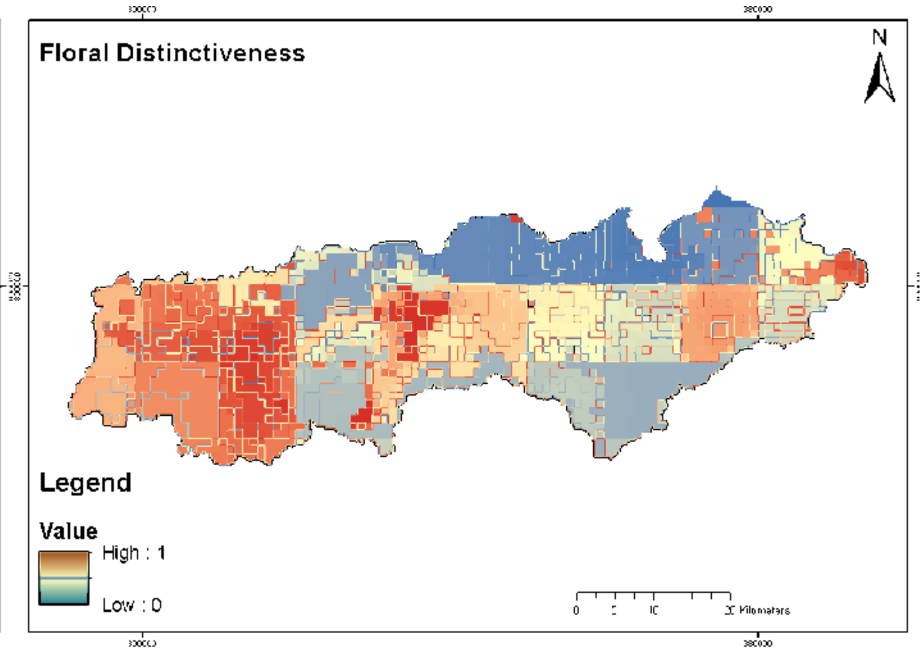

Nitrogen Retention
(InVEST model)
N
Legend
Value
High : 1
Low : 0
Kilometers

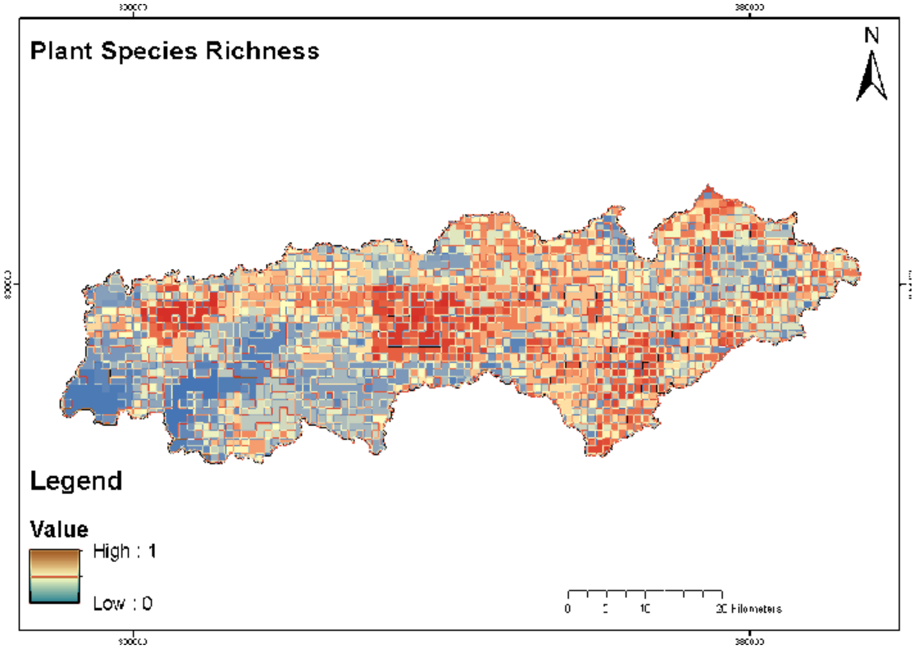

Soil Retention
(InVEST model)
N
Legend
Value
High : 1
Low : 0
Kilometers

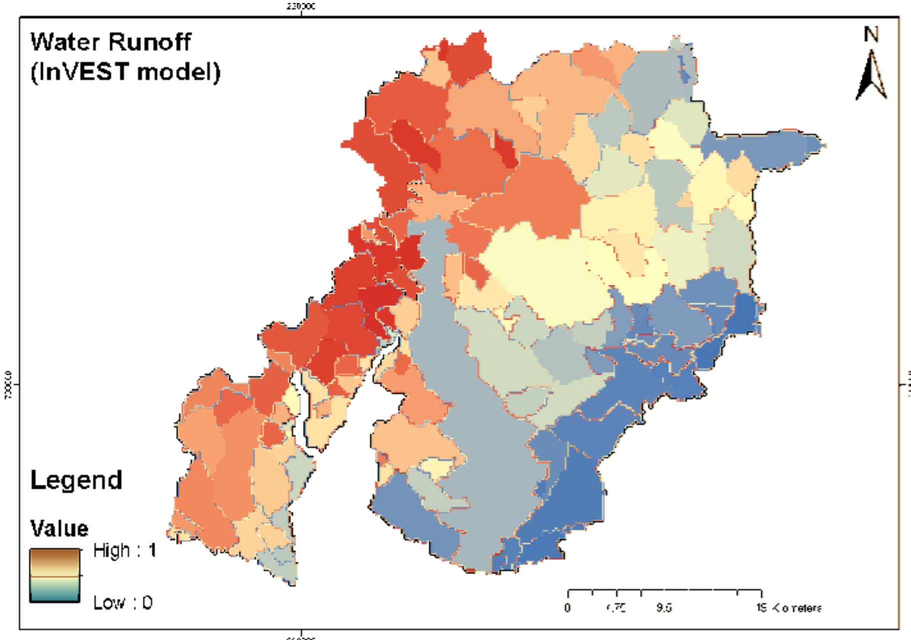

Soil Carbon Content
N
Legend
Value
High : 1
Low : 0
Kilometers

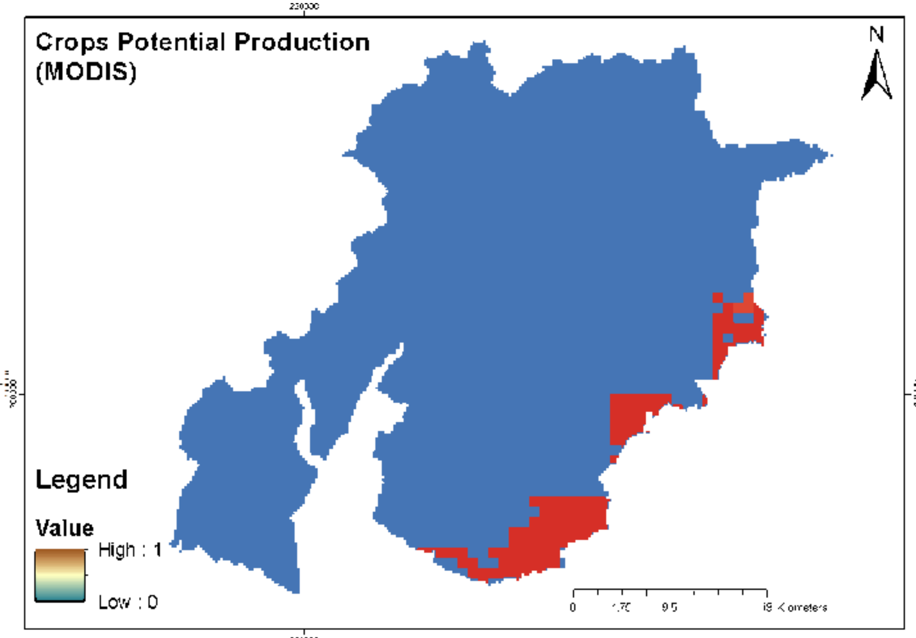

Pollination Supply
(InVEST model)
N
Legend
Value
High : 1
Low : 0
Kilometers

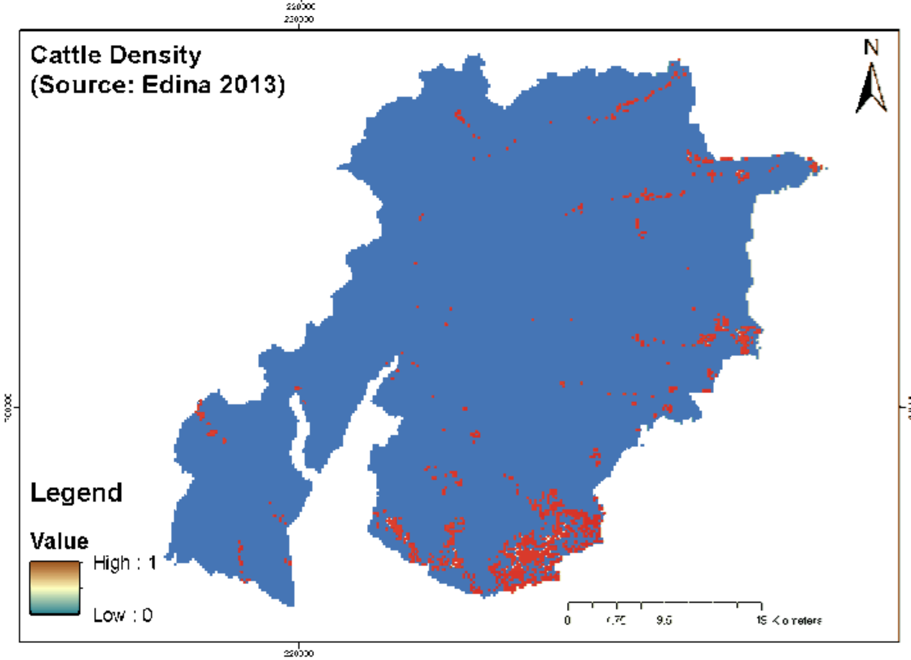

Cultural/Recreation
(Panoramio)
N
Legend
Value
High : 1
Low : 0
Kilometers

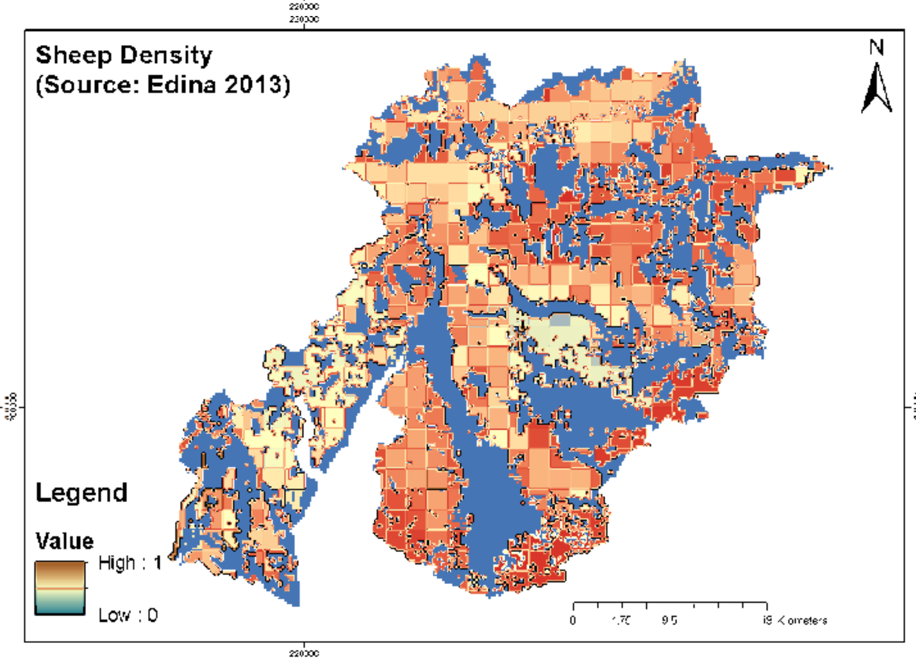

Floral Distinctiveness
N
Legend
Value
High : 1
Low : 0
Kilometers

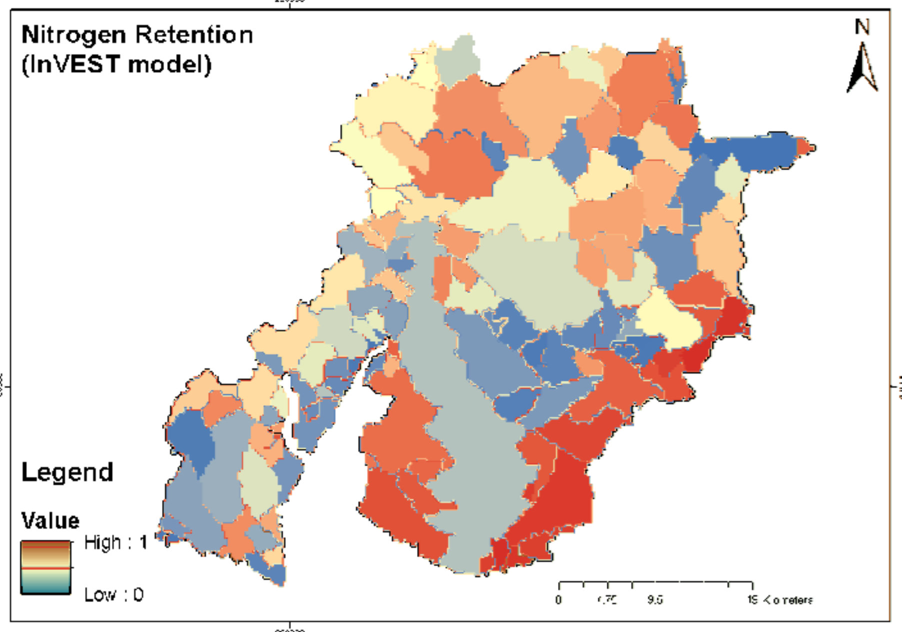

Plant Species Richness
N
Legend
Value
High : 1
Low : 0
Kilometers

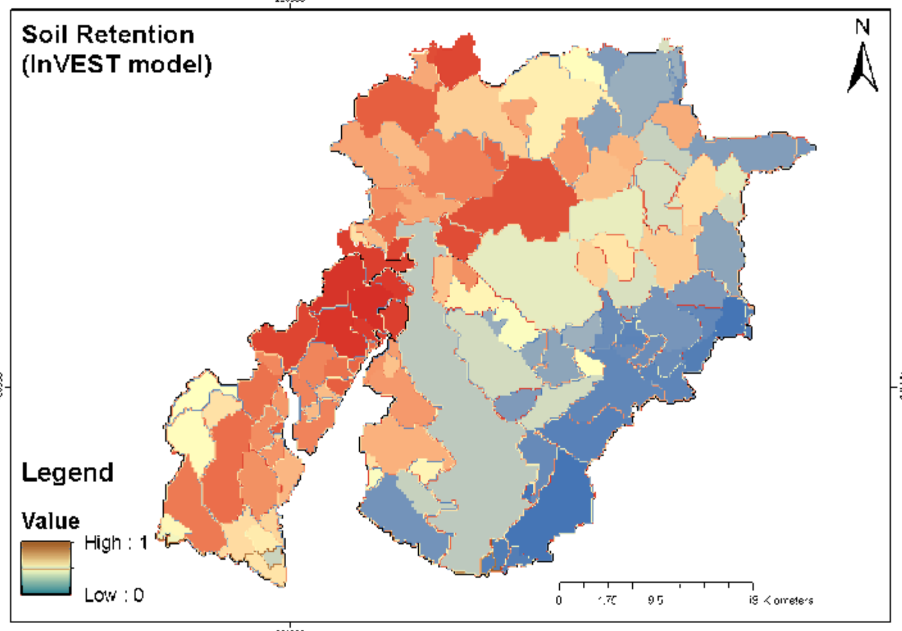

Water Runoff
(InVEST model)
N
Legend
Value
High : 1
Low : 0

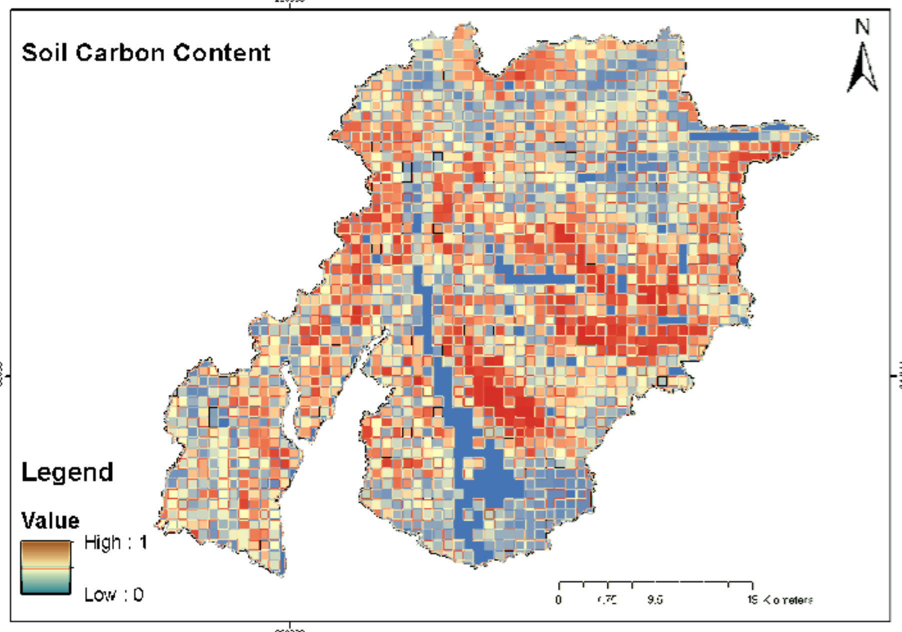

Crops Potential Production
(MODIS)
N
Legend
Value
High : 1
Low : 0

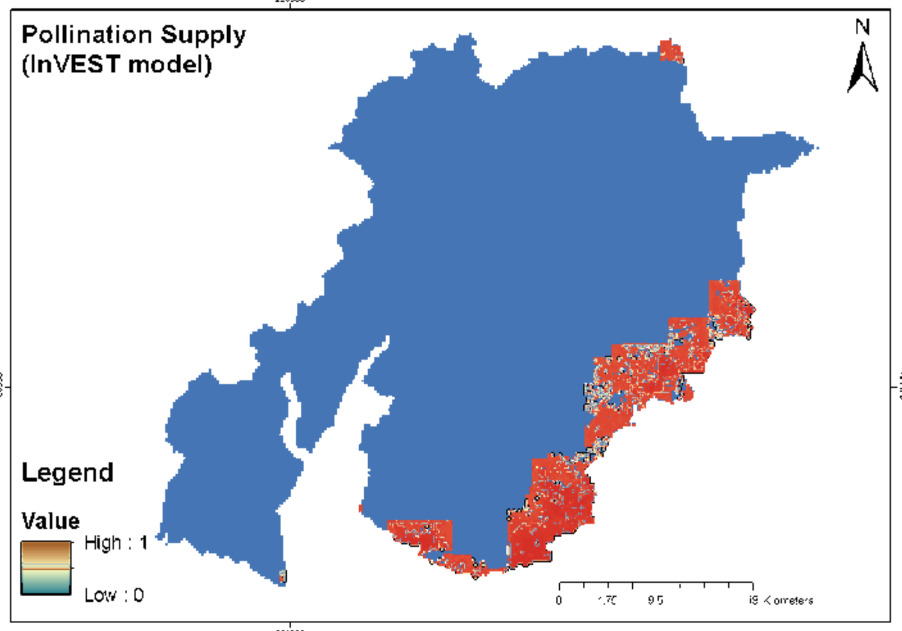

Cattle Density
(Source: Edina 2013)
N
Legend
Value
High : 1
Low : 0

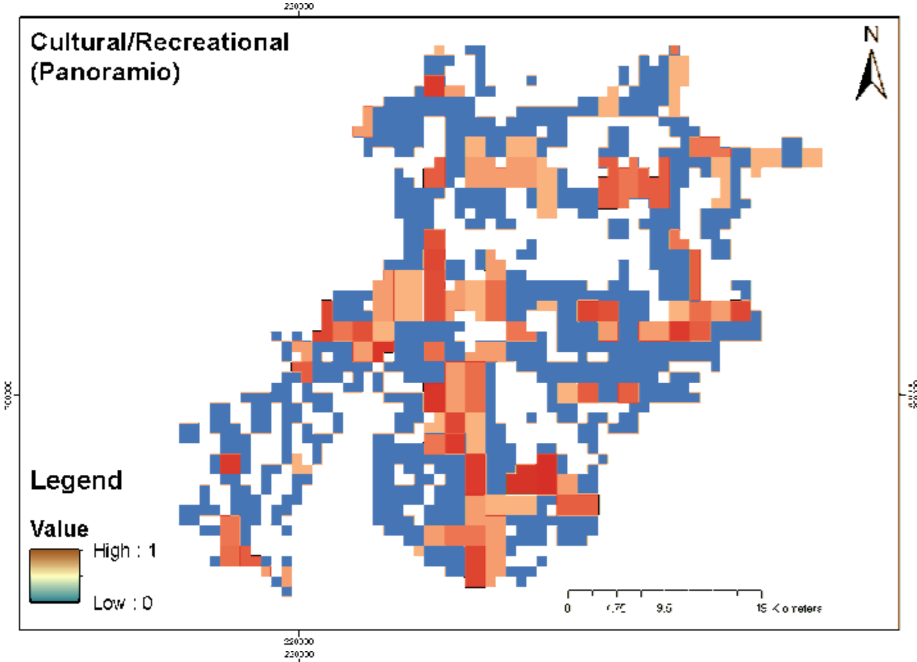

Sheep Density
(Source: Edina 2013)
N
Legend
Value
High : 1
Low : 0

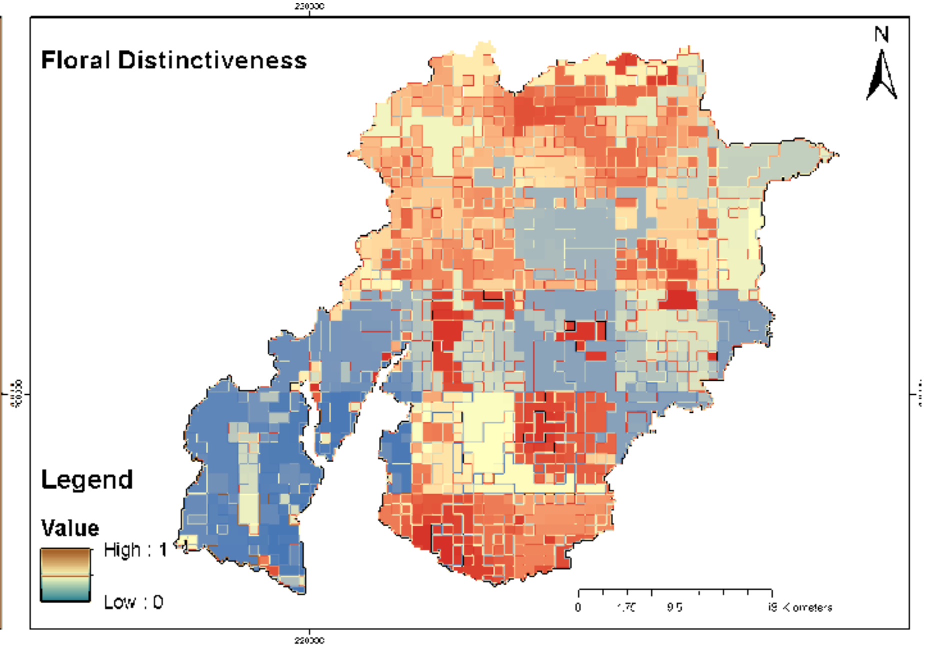

Nitrogen Retention
(InVEST model)
N
Legend
Value
High : 1
Low : 0

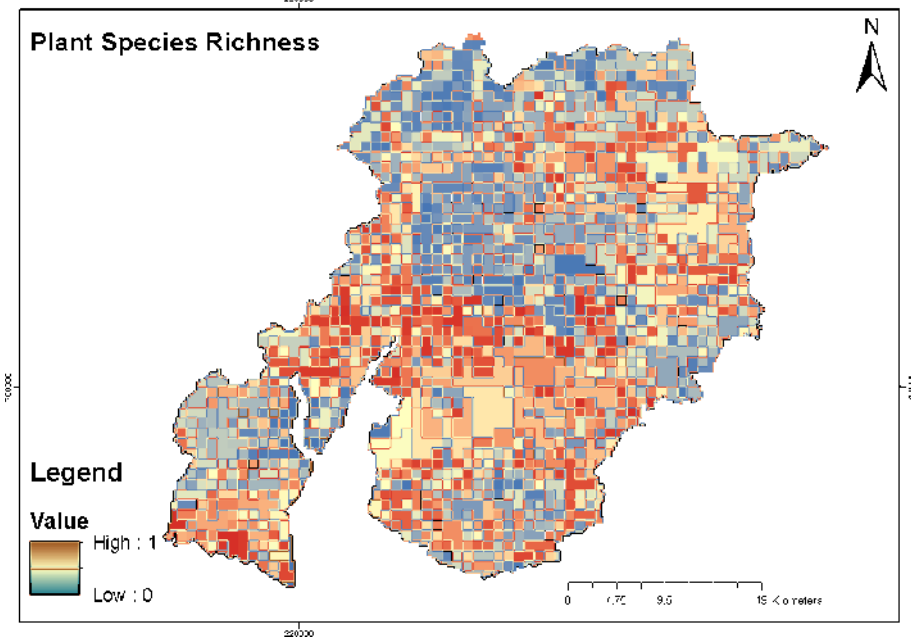

Soil Retention
(InVEST model)
N
Legend
Value
High : 1
Low : 0


Soil Carbon Content
N
Legend
Value
High : 1
Low : 0


Pollination Supply
(InVEST model)
N
Legend
Value
High : 1
Low : 0


Cultural/Recreational
(Panoramio)
N
Legend
Value
High : 1
Low : 0


Floral Distinctiveness
N
Legend
Value
High : 1
Low : 0


Plant Species Richness
N
Legend
Value
High : 1
Low : 0