A Multifaceted Approach to Spatial Analysis of Ecosystem Services

A case study in the New Forest National Park

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Abstract

This study investigated multiple facets of Ecosystem Services (ES), utilising the New Forest National Park as the study site. The scope of the thesis begins with assessment of Cultural Ecosystem Services (CES), before exploring a range of ES in the SSSI (Site of Special Scientific Interest) in the National Park and finishing with an investigation into the implications of climate change on ES provision.

CES are often highlighted as being important, though are often under-represented in ES assessments. The practical difficulties and varied methods available have been suggested to contribute to the lack of uptake of CES assessment. Stated methods of assessment (participatory GIS and stated surveys) and a behavioural method (GPS tracking) were used to ascertain differences in results. Stated response methodologies are suggested to be utilised for aesthetic assessment and behavioural monitoring methodologies for recreational assessment. Measured aesthetic and recreation values were utilised in a wider assessment of ES in the SSSI area within the National Park. Thus, the role of Protected Areas in conserving ES and biodiversity (which underpins many ES) were investigated. Multiple methods of mapping ecosystem services were adopted, including the use of the spatially explicit modelling tool InVEST, and GIS techniques. The SSSI was found to have a higher provisioning level of ES than the non-SSSI within the National Park. With this discovery for the National Park, a final analysis of the provisioning of ES with the impact of climate change was investigated within the Special Area of Conservation (SAC) within the National Park, to ascertain whether a decline could be seen. Low, medium and high climate change scenarios were used to model land change and by extension, ES provisioning change. A spatial multi-criteria analysis showed that the National Park will change extensively in species composition and thus show a decrease in ES provision in the future across all climate change scenarios investigated. Therefore, it is important that management strategies consider the uncertain impact of climate change in any decision making processes.

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Author’s declaration

I confirm the work presented in this thesis is my own.

Arjan S. Gosal

Chapter 1

Introduction

Importance of ES

The Millennium Ecosystem Assessment (MEA) (2005) defines ecosystem services as the benefits that humanity gains from a particular ecosystem. The MEA derives this from two already frequently referenced definitions; the first from Daily (1997) and the second from Costanza et al. (1997). Daily (1997) defines ecosystem services as 'the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life'. Costanza et al. (1997) state 'ecosystem goods (such as food) and services (such as waste assimilation) represent the benefits human populations derive, directly or indirectly, from ecosystem functions'. The MEA found that over the course of the last 50 years, mankind has changed ecosystems more than any other period in time, as demand for food, timber, fibre, water and fuel has increased. This has resulted in gains for humanity, though at the cost of degradation to ecosystem services, with risks of non-linear changes. Many large changes would need to be made to policies, practices and institutions if the damage is to halted or reversed (Millennium Ecosystem Assessment (MEA), 2005)

MEA

The MEA considers humans as integral parts of ecosystems, highlighting the dynamic interactions between humans and ecosystems, with the 'changing human condition' driving changes in ecosystems, and thus changes in human well-being. The MEA firmly cemented the concept of ecosystem services and identified four different types:

- **Provisioning** – These are services that are a direct output from ecosystems and include products such as wood for fuel, Non Timber Forest Products (NTFPs) and food.
- **Regulating** – These services act to regulate dynamic factors such as climate, flooding and quality of water. In a forest ecosystem this can include carbon sequestration.
- **Cultural** – These services are the recreational, aesthetic and spiritual benefits that people get from the ecosystem.
- **Supporting** – These are services that 'support' the output of other ecosystem services, such as photosynthesis and soil formation.

MEA and double counting

The MEA was designed to demonstrate the importance of ecosystems for human wellbeing, but was not designed as a valuation tool (Boyd and Banzhaf, 2007, Balmford et al., 2008). The conceptual framework the UK National Ecosystem Assessment (NEA) adopted is built around the processes that bridge the environment with human well-being; looking at the drivers of change on ecosystems and how goods flow out of that system. This is differentiated from the MEA, with the NEA trying to separate out the valuable goods (or benefits) (UK National Ecosystem Assessment (UK NEA), 2011). The MEA framework was critically evaluated by Fisher et al. (2008), who, while accepting that this typology is useful as a 'heuristic tool', noted that it creates a risk of double counting. Double counting is a term from economics, whereby a good or service are counted more than once (Fu et al., 2011). Fu et al. (2011) state that double counting is a frequent problem in ecosystem service research. The risk of double counting is apparent if we assess each benefit separately economically, as the intermediate services are included in the final service valuation (Fisher et al., 2008).

Other definitions

There are other definitions in the literature, including The Economics of Ecosystems and Biodiversity (TEEB) body whom define an ecosystem service as 'the direct and indirect contributions of ecosystems to human well-being' (Kumar, 2010). TEEB make it clear that ecosystem services are conceptualisations that people label as being indirect and direct 'useful things' that ecosystems provide for us (Kumar, 2010). TEEB used the MEA framework as a foundation for looking at the consequences of biodiversity loss (Balmford et al., 2008). TEEB's aim in valuing ecosystem services meant that they adapted the framework to make it more robust in terms of economic valuation. TEEB built on the MEAs framework by developing classification links between 'wild nature' (used by TEEB to refer to biodiversity in terms of species diversity and biomass) and ecosystems (Balmford et al., 2008). A major advantage of TEEBs method of classifying an ecosystem service is to remove 'double counting' that is inherent in the MEAs framework, as it mixes processes and benefits (Balmford et al., 2008). An apt example of double counting in the MEAs framework is explained by Balmford et al. (2008); pollination and pest regulations as well as food provision are separated, although both contribute to food production – hence double counting occurs. In contrast Fisher et al. (2008) differentiated between 'intermediate and final' services, which were used to distinguish between direct and indirect use of the service. In essence, the intermediate services are the phenomena and the final benefits are the end output which are used by people (Fisher et al., 2008).

Fisher et al. (2008) define ecosystem services as being intrinsically anthropogenic, and as 'the aspects of ecosystems utilised (actively or passively) to produce human well-being'. Boyd and Banzhaf (2007) argue that an ecosystem service definition should be clear enough to allow for environmental accounting. A focus on final services and standardised units are recommended to overcome the issue of the 'ecosystem service' definition being qualitative (Boyd and Banzhaf, 2007).

Church et al. (2014) presented an updated version of the UK National Ecosystem Assessment (UK NEA) (2011) conceptual framework for cultural ecosystems services based on a settings-based approach, with a focus investigating culture in a geographic context. This framework made a distinction between cultural values, environmental spaces, cultural practices and cultural benefits to reduce conflation of services and benefits (Church et al., 2014).

CICES and the ES cascade

Newer classifications have included the Common International Standard for Ecosystem Services (CICES), currently on v. 4.3 (http://cices.eu/). CICES was originally proposed in 2011 as a method for describing ecosystem services, resulting from a European Environment Agency hosted meeting on developing land and ecosystem accounts (Haines-Young and Potschin, 2011). CICES acknowledges the differences in approaches and groupings of ecosystem services (for example between MEA and TEEB) are necessary, though comparison between assessments and integration with other data can be problematic. It aims to be a classification that allows translation of statistical information between approaches, though has its own definitions, falling into the following themes and classes (Haines-Young and Potschin, 2011) (see Figure 1):

- Provisioning
    - Nutrition
    - Materials
    - Energy sources
- Regulating and Maintenance
    - Regulation and remediation of wastes
    - Flow regulation

  - Regulation of physical environment
  - Regulation of biotic environment
- Cultural and Social
  - Symbolic
  - Intellectual and Experiential

CICES was originally aimed at consistent use of data for ecosystem accounts, though now has a wider role, facilitating discussion on the assessment and characterisation of natural capital stocks and flows (Haines-Young and Potschin, 2011). Beyond classification systems, the links between ecosystems services and beneficiaries is becoming increasingly important. The 'ecosystem service cascade' framework is designed to establish that the study of ecosystem services is within a 'coupled socio-ecological system' (Haines-Young and Potschin, 2010). The links between ecosystem services and the benefits that people derive are usually neither linear nor simple, though rather there is a cascade linking both ends. The term 'function' is used like Costanza et al. (1997), Daily (1997) and De Groot et al. (2002) to describe some capacity or capability that people find useful in an ecosystem (Haines-Young and Potschin, 2010). The cascade model also highlights that ecosystem services cannot exist in isolation to the needs of people, and specific beneficiaries need to be identified, a key property in defining ecosystem service typologies (Haines-Young and Potschin, 2010, Boyd and Banzhaf, 2005). The cascade model is designed for valuing incremental changes and generating and allocating ecosystem services (Spangenberg et al., 2014). The cascade model creates a chain through several categories; biophysical structure or process, function, service, benefit and value (Fig 1.1). Although the cascade model links human-wellbeing and biophysical aspects of ecosystems it excludes societal processes (Spangenberg et al., 2014). Spangenberg et al. (2014) suggests that employing the use of 'use value attribution' (the recognition of the natural environments to be potentially 'usable') can identify 'ecosystem service potential'. The mobilisation of an ecosystem system service potential can thus generate ecosystem services and provide a means for cascade exploitation or even using the cascade in reverse as a 'stairway' to facilitate management (Spangenberg et al., 2014).

Figure 1.1: The ecosystem service cascade model (adapted from Haines-Young and Potschin (2010)).

Cultural ecosystem services

Whilst cultural ecosystem services are regularly highlighted as being important (Raymond et al., 2009, Millennium Ecosystem Assessment (MEA), 2005, Lawton et al., 2010), they are not yet clearly quantified or defined (Daniel et al., 2012, Hirons et al., 2016). Human dependence on cultural ecosystem services is now increasing more than on regulating services, with this projected to continue to at least 2020 (Guo et al., 2010). Once cultural ecosystem services are degraded, it is improbable they could be replaced by other means, technological or otherwise (Millennium Ecosystem Assessment (MEA), 2005). Coupled with research that demonstrates inclusion of cultural ecosystem services in assessments can incentivise the multi-functionality of landscapes, clearly shows the importance of these ecosystem services as a research priority.

The UK National Ecosystem Assessment Follow On identified four vital facets of CES; the benefits, cultural values and practices, and the environmental spaces (Church et al., 2014). The challenge of quantifying CES was acknowledged by Church et al. (2014); though it was made clear that quantitative indicators for CES can be developed, with participatory approaches, such as those used in the social sciences can be used successfully, with progress of CEs accounting relying on a diversity of approaches, with mapping methods allowing 'latent cultural values' to be discovered. Church et al. (2014) found that a large amount of variability in availability and access to environmental spaces, and emphasised the need for local data for robust local interpretation of CES indicators.

The relationship between ES and beneficiaries

Chan et al. (2016) suggests that the traditional method of focussing on instrumental (satisfying preferences) or intrinsic (possessing inherent worth) values of nature can disregard the 'well-being' aspects people derive from the environment. Chan et al. (2016) suggests that relational values, those of how people feel they relate to nature and others, need to be considered seriously, as these are derivative of relationships. This allows individual preferences to be reframed taking into consideration consistency with values such as virtue and justice. Considering relationships can allow management and decision makers to focus on increasing the positive effects of people and their lifestyles on the environment, whilst reducing the negative effects (Chan et al., 2016). It has been suggested by Cooper et al. (2016) that although human benefit from spiritual and aesthetic experience of nature, moral responsibilities regarding nature are only developed with an understanding of the value of nature, which are more important that the benefits people gain from nature. It is important for management and policy-makers to consider the existing relationships between people and nature. This has been demonstrated through people-centric activities such as participation in planning as part of a collective negotiation towards good stewardship (Chan et al., 2016).

It is important to consider how beneficiaries relate to ecosystem services. Schirpke et al. (2014) states that ecosystem services need to be considered in ecosystem services, with a focus on both supply and demand. Though studies often disregard beneficiaries, or purely deal with them in a qualitative manner (Burkhard et al., 2012). This flow is so important, that is has been suggested that beneficiaries are entitled to compensation for ecosystem service loss from land-owners (Davidson, 2014). In recent years, studies have started to quantitatively investigate the role of beneficiaries, including Schirpke et al. (2014) whom mapped the flows between Natura 2000 sites and ecosystem service beneficiaries. Others have modelled the flow, for example Bagstad et al. (2014) used the ARtificial Intelligence for Ecosystem Services (ARIES) model to map supply, demand and flow of ecosystem services in Puget Sound, Washington State, USA, finding that of those regions cable of supplying those ecosystem services measured, only 16-66% in reality did. Indeed it is important to identify demand, as threats to ecosystem services will often first manifest as loss of provision to beneficiaries (Maron et al., 2017).

BIODIVERSITY

Ecosystem goods and services, as well as the ecosystem properties that underpin them, all depend on biodiversity (Hooper et al., 2005). Hooper et al. (2005) state that ecosystem functioning is a broad term for ecosystem properties, ecosystem goods and ecosystem services. Here, I will use the term in the strict sense of only covering ecosystem properties as other researchers have done. Ecosystem properties cover sizes of compartments and rates of processes. Ecosystem goods are tangible outputs that have a market value and ecosystem services are those ecosystem properties that have an indirect or direct effect on human activities, such as hydrological cycles.

The effect of biodiversity on ecosystem services and goods (the benefits that people gain) has long been debated (Hooper et al., 2005). Biodiversity is a broad term that covers a vast array of biotic scales, from genetic diversity within a particular species to global biome distribution (Hooper et al., 2005). With anthropogenic influences impacting biodiversity, it is important to know whether this in turn has an effect on the provision of ecosystem services and functions (Hooper et al., 2005)

There is much interest in trying to identify how ecosystem services and biodiversity are linked. A recent study by Balvanera et al. (2006) highlighted that the issue of how biodiversity and ecosystem process rates are linked was contentious and conducted a meta-analysis; covering data from 1954 to 2004 showed that less well controlled biodiversity manipulations would lead to weaker biodiversity effects. They also found biodiversity change was greater at the community than ecosystem level and negative at population level. As trophic links increased between those elements measured and those manipulated, productivity-related effects declined (Balvanera et al., 2006). Of those assessed, Balvanera et al. (2006) found that biodiversity had a positive effect on most ecosystem services. Productivity (a supporting ecosystem service) was found to increase as biodiversity increases on the same trophic level, with plant diversity increasing erosion control (due to microbial biomass underground being enhanced). Ecosystem properties related to nutrient cycling were found to increase in relation to biodiversity. Furthermore, functional traits of species have a direct effect on biodiversity. Ecosystem properties are influenced and respond to species composition, richness, evenness and the interactions between these factors (Hooper et al., 2005).

The relationship between biodiversity and ecosystem services has been more recently investigated by Mace et al. (2012). The ‘ecosystem services perspective’ equates biodiversity and ecosystem services as the same often represented with simple ecosystem service flows, with one enhancing the other or vice versa, with the ‘conservation perspective’ defining biodiversity as an ecosystem service, though ignores its underpinning role to other ecosystem services (Mace et al., 2012). The complex nature of biotic and abiotic elements, interactions and all biodiversity components have a direct impact on the supply of ecosystem services (Mace et al., 2012). Mace et al. (2012) suggests that the role of biodiversity can occur across multiple ecosystem services and processes, hence referred to the relationship as multi-layered, with the recommendation that the relationship is acknowledged at various levels: biodiversity as a regulator of ecosystem processes (i.e. pollinators providing stability in ecosystems), final ecosystem service (i.e. organisms with novel pharmaceutical expression) and a good (flagship species for conservation).

Cardinale et al. (2012) reviewed the impact of biodiversity loss on ecosystem functioning and ecosystem services. Biodiversity loss as a reduction in genes and species has been found to reduce the uptake of resources such as water and light. In addition, biodiversity is found to increase the temporal stability of ecosystem functioning. Decreases in biodiversity has been found to accelerate the change in ecosystem processes. Individual species and complementarity amongst species aid in controlling ecosystems, with trophic level diversity mediating ecosystem functioning. Additionally functional traits of organisms have a direct impact on ecosystem function (Cardinale et al., 2012). Cardinale et al. (2012) did not include cultural services in their review, though highlighted the rarity that biological diversity had been investigated in regards to CES.

A study conducted by Chan et al. (2006) used a spatially explicit framework to assess the trade-offs and align biodiversity with a selection of ecosystem services including carbon storage and water provision. Chan et al. (2006) found that biodiversity conservation protected large proportions of collateral flows of these ecosystem services (Chan et al., 2006), demonstrating the importance of ecosystem service mapping. Naidoo et al. (2008) created a set of imperfect global proxies from available data to illustrate how mapping could be used. They found regions that maximised biodiversity did not increase ecosystem service provision over those picked at random (Naidoo et al., 2008). However, through the use of spatial mapping of the different services, 'win-win' areas for both ecosystem service provision and biodiversity could be identified. We may have already lost services provided at larger scales as habitats in the UK and Europe are currently fragmented, which may have negative impacts on species richness and on the resilience of natural communities to environmental change (Biodiversity and Ecosystem Service Sustainability (BESS), 2011, Lawton et al., 2010).

Abiotic drivers

Abiotic drivers such as land use have strong effects on ecosystem service provision, though these are also modulated by biological functional diversity. This can include relative abundance of plant functional traits, value and range (Díaz et al., 2007). Díaz et al. (2007) attempt to link functional diversity with provision of ecosystem services through a framework in which functional diversity (which responds to land use change) alters ecosystem service provision of those ecosystem services that are important to local stakeholders. Lavorel et al. (2011) found that using plant functional traits to analyse ecosystem services on a landscape scale was a powerful way to start to understand the underpinning ecological mechanisms of ecosystem service provision as well as any trade-offs or synergies between services. For example, it was found that vegetation height and leaf traits were affected strongly by the abiotic environment and land use. This in turn had a cascading effect on certain ecosystem properties that could then be used as functional markers of ecosystem services (Lavorel et al., 2011). Hence it is important to be aware of the link between function and biodiversity and work on an appropriate scale.

Landscape scale

A lot of recent literature has highlighted the need to understand dynamics of ecosystem services at the landscape scale. This requires spatial analysis and mapping of ecosystem services and their underlying functions. Studies in the past have looked at the supply and demand of ecosystem services, either assessing threats or looking at economic valuation. Early research often neglected the role of biodiversity in the provision of these ecosystem services (Kremen, 2005). Naidoo et al. (2008) suggest that unless ecosystem service production is mapped, it is difficult to establish which conservation actions would benefit both biodiversity and ecosystem service provision. In addition, biodiversity-function studies have looked at small scale communities that do not translate into real landscapes (Kremen, 2005). Kremen (2005) suggests that a 'bridge' is needed between these two approaches, including measuring of ecosystem services over the spatial-temporal scale upon which they operate. The need to understand how these relationships translate to a landscape scale is important for environmental management decisions, so that they are informed. At this larger scale, the biodiversity environment and trophic feedbacks are different from small-scale laboratory and field experiments. Small scale experiments often result in a decidedly non-linear scaling to landscape scale. Ecosystem provision is likely to be scale dependent, thus spatial scaling becomes a larger issue (Biodiversity and Ecosystem

Service Sustainability (BESS), 2011). When working at the landscape level there are a myriad of processes that underpin the provision of ecosystem services, but these remain poorly defined. Identifying how biodiversity within landscapes underpins the delivery of different ecosystem services at a range of scales and across gradients therefore represents a major research priority (Biodiversity and Ecosystem Service Sustainability (BESS), 2011).

Scaling issues

Scaling issue have long been recognised in the ecological community Wiens (1989), with spatial patterns often being scale dependent (Grafius et al., 2016). During recent year, research efforts into the effect of scale has continued, for example Anderson et al. (2009) used the concept of 'service providing units' to understand the landscape-ecosystem service connection by considering both spatial and temporal scale of urban ecosystem services. There can be practical constraints with research that aims to capture highly accurate data on spatial heterogeneity, with accuracy trade-offs (Schröter et al., 2015). Grafius et al. (2016) suggests that there are two vital issues when modelling ecosystem services. The first is to be able to specify the relationship between the ecosystem service and the environmental element being studied (e.g. habitat), and understanding the underpinning processes. The second is having the appropriate scale data for the environmental composition, otherwise ecosystem service provision can be misrepresented (Grafius et al., 2016). This was demonstrated by Di Sabatino et al. (2013) where the 30 m resolution CORINE land cover dataset underestimated the economic value of an Italian landscape.. Another study compared 1 km and 30 m resolution lands cover datasets for the US, finding that the finer resolution imagery led to a total ecosystem service value of nearly 200% greater (Konarska et al., 2002). A newer study by Foody (2015) highlighted the necessity of accurate and rigorous land cover maps, finding that the misclassification of land cover would lead to a US$ 518 billion year^{-1} over-calculation for total ecosystem service value. This scale issue has been demonstrated by Grafius et al. (2016), whom found that carbon storage was estimated at a higher level with finer resolutions, compared to pollinations which was assessed at higher levels at a courser resolution in urban environments. It is hence clear that the resolution of the data used, and validation is necessary for robust results. It is important to consider scaling issues, as the scale that an area is investigated can alter driving processes and the scale that ecosystem services are mapped can have large impacts on final values.

Methods of mapping and comparisons

After quantifying ecosystem services, various studies have used different approaches to compare spatial or temporal trends, for example Anderson et al. (2009) visualised relationships through mapping, White et al. (2012) used trade-off curves and Foley et al. (2005) used star diagrams, though for quantitative analysis pairwise correlation coefficients are popular (for example Chan et al. (2006) and Bai et al. (2011)) (Mouchet et al., 2014). Other methods for quantitation for investigating bundles of ecosystem services including chi-squared tests, Principal Component Analysis (PCA), Multiple Correspondences Analysis (MCA), analysis of clustering (i.e. using *K* means) and overlap analysis (Mouchet et al., 2014). Examples of the latter include Gos and Lavorel (2012) whom investigated stakeholder expectations and ecosystem service and biodiversity hotspots; finding that overlap depended on threshold used. For the quantitative analysis of identification of drivers of these bindles, multiple methods can also be used, including distance approaches (i.e. Analysis of Similarity (ANOSIM), analysis of variance, regression based models, machine-learning methods, time-series methods and canonical analysis (Mouchet et al., 2014).

Supply and demand mapping

Mouchet et al. (2014) urged the need for a consensual framework for assessment of both the supply (ecological) and demand (socio-economical) side of ecosystem assessments. Through a methodological approach which includes consideration of ecosystem service delivery and stakeholder needs can the supply-demand aspect of ecosystem services be fully appreciated (Mouchet et al., 2014). Associations between ecosystem services (i.e. due to being supplied by the same ecosystem process or external factors impacting multiple ecosystem services) can lead to positive covariance being often studied spatially or temporally (Mouchet et al., 2014).

Ecosystem service bundles

Ecosystem services bundles are important when considering managing natural resources, often explained due to landscape multi-functionality (delivery of multiple benefits from ecosystems), though the association between landscape heterogeneity and multiple ecosystem services is not conclusive (Crouzat et al., 2015, Mastrangelo et al., 2014). Pattern-based multi-functionality can be investigated using spatial techniques to detect coincidence of ecosystem services. To identify process-based multi-functionality spatiofunction techniques need to be used (Crouzat et al., 2015). Crouzat et al. (2015) used a spatial approach to investigate supply of multiple ecosystem services, finding that a multi-step analysis (at nested-scales) allowed identification of associations between ecosystem services, with local landscape heterogeneity not implying multiple ecosystem service delivery.

Marginal change over total stocks

When valuing ecosystem services, it is important that marginal change is considered rather than total stocks, as it is at the margin that policy and economic decisions operate (Fisher et al., 2008). Robust valuation of ecosystem services therefore requires two contrasting hypothetical states to be accounted for; such as a business-as-usual state (where the action has not been implemented) and a biodiversity-friendly state (where the action has been implemented). While these 'marginal' changes may seem relatively small in comparison to full stocks, they can be large in absolute terms, for example a piece of land covering hundreds of hectares going through land use change (Balmford et al., 2008). It is important to measure the consequences of marginal changes to examine the potential impact of management or policy interventions (Balmford et al., 2008), to ensure that the outcomes of research are policy relevant.

Ecosystem services and policy

There has been a recent focus on ecosystem services in both policy and research. For example, the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) was launched in April 2012 and there have been recent reports highlighting the importance of ecosystem services in England, such as the National Ecosystem Assessment (UK National Ecosystem Assessment (UK NEA), 2011) and the Lawton review (Lawton et al., 2010). The Lawton Review (Lawton et al., 2010), a report to Defra, looked at how a coherent and resilient ecological network was important – highlighting how the natural environment provided us with many ecosystem services, making it clear that biodiversity underpins most (if not all) of them. Similarly, 'Biodiversity and Ecosystem Service Sustainability', or BESS, is a research programme that was commissioned in 2011 for five years by the Natural Environment Research Council (NERC), with a view to understand the functional role of biodiversity in ecosystem processes (Biodiversity and Ecosystem Service Sustainability (BESS), 2011). This has meant there has been much attention to the fact that we need to understand the drivers and dynamics of ecosystem services if we are to manage and understand the provisioning of them.

The links between ecosystem services and policy are increasing. Waage et al. (2014) notes several global trends, including the inclusion of GDP measures to include natural capital, for example in addition to the European Union, 18 countries are involved with the World Banks 'Wealth Accounting and the Valuation of Ecosystem Services' (WAVES) programme. Payment for ecosystem services (PES) schemes are being more popular, for example Vietnam earning US 58 million in 2012 from payments from water and hydroelectric plants, using the funds for ecological restoration (Waage et al., 2014). In total 68 countries were engaging and working on ecosystem services, with several national assessments, including the UK National Ecosystem Assessment (UK NEA) (2011) (Wong et al., 2015). As government interest in ecosystem services increases, demand for the standard assessment of the valuation and mapping of ecosystem services is increasing, with a need for data on the link between biophysical measurements and final ecosystem services (Wong et al., 2015). Benefits transfer has been the predominant method to address this issue, with land cover proxies often being used (Seppelt et al., 2011), though these are lacking as they often use secondary data hence are not a robust method (Wong et al., 2015). Wong et al. (2015) suggests that ecological production functions can be used for assessing ecosystem services so that marginal changes in ecosystem characteristics and then final ecosystem services can be investigated to determine trade-offs; allowing policy-makers to make informed management decisions. A 10-step approach for measuring ecosystem services for public policy has been developed by Wong et al. (2015), split into two phases, the first identifies metrics and indicators and the second regards biophysical measurement.

Protected areas

The current status of England's ecological network is dependent on a wide selection of statutory and non-statutory wildlife sites across a range of habitats. The Sites of Special Scientific Interest (SSSI) series of sites alone do not make a resilient ecological network. Lawton et al. (2010) found that across England the size of wildlife sites was too small. Some habitats were found to have incurred losses that require significant resources to prevent further degradation of wildlife and connections to other sites. Under-management and a lack of protection was noted across most of the semi-natural habitats (the exception being Natura 2000 and SSSIs sites). It was also highlighted that access to wildlife was limited by a lack of easy access (Lawton et al., 2010).

Ecosystem services in the UK

The UK National Ecosystem Assessment (UK NEA) (2011) found that timber production was an important provisioning service from woodlands in the UK, with 8.5 million green tonnes of softwood being produced in 2009, with a 60% annual growth increment. Estimates suggest that by the 2020s there will be an increase of up to 12 million tonnes annually. Hardwood production from broadleaved trees was at 0.4 million tonnes with a 20% annual growth increment. Timber is not the sole product from UK woodlands, non-timber products can also be important, with game shooting bringing the UK economy an approximate £640 million annually (Quine et al., 2011). The UK National Ecosystem Assessment (UK NEA)
(2011) also found that woodlands provided a large amount of social and cultural services, with an estimated 250-300 million day visits annually; recreational visits valued at £484 million in 2002 (at 2010 prices). Other services include carbon sequestration, with 800 megatons of carbon stocks in UK forests, with a further 80 megatons in timber and wood products (Quine et al., 2011).

Knowledge gap

The importance of quantifying ecosystem services has been identified as being pivotal to the allocation of environmental resources (Millennium Ecosystem Assessment (MEA), 2005, Troy and Wilson, 2006, National Research Council, 2004). Whilst biophysical and economic vales are often used in management and conservation planning decision making, cultural ecosystem system values are often seldom considered. It is with a view to fill this knowledge gap that this study is conducted. This literature review highlights that a number of knowledge gaps exist, particularly the spatial distribution of cultural ecosystem services at the landscape scale.

Thesis format

More details on relevant research areas, such as ecosystem services, protected areas and climate change will be covered in subsequent chapters. The format of this thesis will follow four discrete research chapters, each aiming to answer distinct questions related to ecosystem services, using the New Forest National park as a case study. The first will examine the importance of ecosystem services and use an online survey to discover the differences in using a fixed-sum scoring method and online participatory GIS (pGIS) to elicit values for recreational and aesthetic ecosystem services. The second chapter takes this further, surveying people face-to-face within the New Forest National Park, comparing stated responses from face-to-face surveys to GPS tracking of individuals to investigate the aesthetic and recreational values from the two methods. The third chapter broadens the scope of the ecosystem services examined, and delves into the ecosystem service provision within the SSSI protected area within the New Forest National Park. The fourth and last data chapter looks forward into the uncertainty of climate change, and its impact of the provision of ecosystem services within protected areas of the New Forest. This thesis is the culmination of an effort to delve deeper into the quantification of cultural ecosystem services, ecosystem service provision, and place these firmly into the context of the current issues of protected areas management, within a future of changing climate.

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Chapter 2

Eliciting cultural ecosystem service values for the New Forest National Park

2.1. Abstract

Often ignored in ecosystem service assessments, Cultural Ecosystem Services (CES) are highlighted as important by several critical global reports including the Millennium Ecosystem Assessment and the UK National Ecosystem Assessment and over 60% having already been deteriorated or over-used. Cultural services, when investigated, are often inferred from sites of recreation or tourism. This study aimed to increase the understanding of recreation and aesthetic values within the New Forest National Park, southern England. Novel approaches to using a web-based participatory GIS (pGIS) web page and using fixed-sum scoring questions were used as part of an online survey to elicit these values. Results found that participants ranked cultural ecosystem services as more important than other ecosystem services. Broadleaved woodland, followed by dwarf shrub heath was found to be most aesthetically pleasing, with broadleaved woodland also being most important for recreation. Coastal areas and water habitats were also found to have high cultural values. PGIS and fixed-sum methods of ascertaining value for cultural ecosystem services showed contrast, suggesting the need for local quantification of cultural ecosystem services using appropriate methods.

2.2. Introduction

The need to address the development of ecosystem management science has been highlighted by Mace et al. (2012). The Millenium Ecosystem Assessment (2005) found that over the course of the last 50 years, mankind had changed ecosystems more than any other period in time, as demand for food, timber, fibre, water and fuel has increased. This has resulted in gains for humanity, at the cost of degradation to ecosystem services, with risks of non-linear changes. Many large changes would need to be made to policies, practices and institutions if the damage is to halted or reversed (Millennium Ecosystem Assessment 2005b). Quantification of ecosystem services is pivotal to the allocation of environmental resources, with over 60% having already been over-used or deteriorating (Millenium Ecosystem Assessment, 2005). The value of ecosystem services, both monetary and non-economic, is high. As far back as 1997 the global national product total of ecosystem services worldwide was estimated at $18 trillion per year, providing an overwhelming contribution to human welfare (Costanza et al., 1997). More recently it has been acknowledged that ecosystems and natural capital are of great value; indeed, nature-based solutions could be more cost effective, whether it is for carbon storage, water purification, flood protection or recreational opportunities (Daily and Matson, 2008, Ervin et al., 2012, Maes et al., 2012). The importance of quantifying ecosystem services has been identified as being pivotal to the allocation of environmental resources (Troy and Wilson, 2006; Heal et al., 2005 and Millennium Ecosystem Assessment, 2003).

The challenge of deciding optimal allocation and management of landscapes has been deemed a priority, with demand from a broad-spectrum of stakeholders (De Groot et al., 2010). Natural resource management is key to conservation biology and both landscape and restoration ecology (Lindenmayer et al., 2008). Ecosystem service provision has been very popular in relation to policy recently. The Lawton Review (Lawton et al., 2010), a report to Defra, looked at how a coherent and resilient ecological network was important – highlighting how the natural environment provided us with many ecosystem services, making it clear that biodiversity underpins most (if not all) of them. Research programmes have reflected the popularity of ecosystems services, for example 'Biodiversity and Ecosystem Service Sustainability' (BESS) is a research programme that was commissioned in 2011 for five years by the Natural Environment Research Council, with a view to understand the functional role of biodiversity in ecosystem processes (Biodiversity and Ecosystem Service Sustainability (BESS), 2011). The Conference of Parties (COP 10) to the

Convention on Biological Diversity (CBD) in 2010 led to the Aichi Targets, a collection of 20 targets as part of the Strategic Plan. Target 11 (global expansion of protected areas) and Target 14 (priority for protection and restoration) see ecosystem services as important, and require their due consideration (EU Commission, 2011, Maes et al., 2012). More and more we can see the implementation of the ecosystem approach at policy level, for example ecosystem services were included in the assessment stage for the European Commission's 'The blueprint to Safeguard Europe's Water resources' (European Commission, 2012).

Whilst biophysical and economic values are often used in management and conservation planning decision making, community ecosystem system values are often seldom considered (Raymond, et al. 2009). Whilst cultural ecosystem services are regularly highlighted as being important, they are not yet clearly quantified or defined, often due to methodological challenges (Plieninger et al., 2013, Daniel et al., 2012), with the exception of recreation, being considered in environmental assessment. The UK National Ecosystem Assessment Follow On identified the benefits, cultural values and practices, and the environmental spaces as being the four most important aspects of CES (Church et al., 2014). Church et al. (2014) suggests that there is usually a strong recursive and non-linear dimension to the relationship between these aspects. Cultural services are usually inferred from recreation and tourism or heritage sites, with cultural mapping being underrepresented, and recreation being the most often mapped (Brown and Fagerholm, 2015, Crossman et al., 2013). This is partly due to cultural ecosystem services and nonmaterial values having no common framework of elicitation (Chan et al., 2012). Guo et al. (2010) predicted that humanity's reliance on cultural services will increase more than for regulating and provisioning services between 2005 – 2020. In a review conducted by (Milcu et al., 2013), it was found that most studies used many differing approaches, owing to the eclectic nature of cultural ecosystem services or being assessed as a peripheral addition whilst assessing other services. Chan et al. (2016) suggests that the 'well-being' aspects of nature can be easily disregarded when focusing on instrumental values, proposing instead 'relational values' that allow for values that are derivative values such as virtue and justice to be considered.

Mapping of cultural services is a popular method as an alternative to valuation (Kumar, 2010). Mapping of ecosystems has been used for prioritisation of areas in management and planning. A number of different studies have developed methods for assessing cultural ecosystems services, for example, Chan et al. (2006) used a spatially explicit planning framework in the Central Coast ecoregion of California, United States, to highlight conservation goals and six ecosystem services: outdoor recreation, carbon storage, flood control, forage production, crop pollination, and water provision, with the only negative correlations being forage production and crop pollination. Similarly, O'Farrell et al. (2011) studied the use of sustainable land-use practices (which also had human benefits) to promote conservation in the Succulent Karoo biome in western South Africa. Coupled socio-ecological systems have been explored by Bryan et al. (2010) who interviewed community representatives in the south Australian Murray-Darling Basin region to elicit ecosystem service values for identifying priority management areas. Alessa et al. (2008) similarly collected community values for ecosystem services as point data finding that there were moderate positive relationships between perceived biological diversity and net productivity in half the communities studied.

However, different ecosystem services can be provided by different places – they are not necessarily co-located. The challenge of managing multiple ecosystem services often leads to trade-offs, as well as synergies (Raudsepp-Hearne et al., 2010, Bennett et al., 2009). Raudsepp-Hearne et al. (2010) investigated ecosystem service bundles (those services that often appear together) in Quebec, Canada, finding that trade-offs existed between provisioning and nearly all cultural and regulating services. A study by Murray et al. (2015) explored the spatial overlaps of biodiversity, carbon stocks and deforestation threats in REDD+ (Deforestation and forest Degradation) areas in Indonesia and found no spatial congruence between carbon stocks and biodiversity richness. Owing to cultural services often being unquantified, losses in these services are not accurately reflected in trade-off models (Rodríguez et al., 2006). The need to map both ecological (supply) and socio-economical (demand) side of ecosystem services is important, as it allows both ecosystem service delivery and stakeholder needs to be fully understood (Mouchet et al., 2014)

It is important to elicit stakeholder views, which are often excluded from the management process. Stakeholders here are considered to be any affected individuals or organisations – anyone with a 'stake' in the process. The complexity of environmental projects can benefit from the diversity and knowledge that comes from multiple inputs. The inclusion of stakeholder participation can increase the durability and quality of decisions and through including those that may be marginalised on the periphery of the decision making process, promoting active citizenship and nurture long term support (Fischer, 2000, Beierle, 2002, Reed, 2008). The benefits of stakeholder participation in environmental decision making has been recommended widely, and is often incorporated into policy at multiple levels for example, the United Nations Convention to Combat Desertification (UNCCD) places great emphasis of community participation in land degradation assessment (Reed, 2008). Participation is also seen as a democratic right, being contained within the United Nations Economic Commission for Europe's 1998 Arhus Convention (Reed, 2008). One approach that can potentially be used to strengthen stakeholder involvement in ecosystem service mapping is participatory GIS (pGIS).

PGIS methods are spatially explicit techniques that have been used to identify and map ecosystem services, especially cultural and provisioning benefits (Brown and Fagerholm, 2015). These can simple and paper based to more complex online maps (Brown et al., 2012). PGIS methods have been used across a wide array of scenarios, from community planning to ecological knowledge. It has been used to elicit multiple ecosystem services values (including spiritual, aesthetic and recreational) as part of the planning process for Chugach National Forest, Alaska, United States, whom used paper maps (Brown and Reed, 2000). Beverly et al. (2008) used an online mapping application to collect forest landscape values, finding that recreational, existence, wilderness and biological diversity values clustered together at the landscape scale. Community participation through the use of postal surveys and grayscale maps to ascertain priority areas for conservation has been used for the Murray River reserves, Victoria, Australia (Pfueller et al., 2009). Other studies have included mapping social values in Helsinki, Finland for strategic green area planning (Tyrväinen et al., 2007) and Levine and Feinholz (2015) used PGIS to inform coral reef ecosystem management in Hawaii, with the process being valuable for stakeholder identification and engagement, and the results being informative to federal managers to understand human implications for management decisions.

Internet based participatory GIS has been used for Grand County, Colorado to identify ecosystem services by Brown et al. (2012), covering 22 services including cultural services (including recreation and aesthetics), provisioning services (e.g. natural materials), regulating services (air quality) and supporting services (e.g. soil formation).They found that participants found cultural and provisioning services the easiest to identify on the map, with supporting and regulating the most difficult. This sentiment has been mirrored by researchers in Africa, with cultural ecosystem services being easily articulated, as relating to daily life of the individual (Brown et al., 2012)

Participatory mapping approaches are not without criticism. There can be methodological trade-offs with using an online based self-administering participatory GIS system compared to face-to-face interview-based mapping. With the latter researchers are able to actively monitor participation, whilst the former allows use of advanced cartographic features (i.e. zoom) and accurate targeting of users, though requires an internet connection and appropriate hardware and software. The complexity of online systems can lead to a decrease in participation (Brown et al., 2012). Participants in pGIS are often referred to as the public, with much ambiguity as to whom this group is. It can potentially encompass decision makers, affected individuals or even random public – there can be a difficulty in delineating the term (Schlossberg and Shuford, 2003).

However, a participatory GIS approach has not been compared to questionnaire responses within the same group of participants. This is important because online pGIS approaches are more resource intensive in terms of both technical ability and economic cost than more traditional methods of gathering opinions, such as surveys. It is important to compare both methods, as if the results are the same, then the easiest method could be adopted with less complexity in resource needs. If differences in results occur, it is important to understand why these take place.

Consequently, this chapter investigated the spatial distribution of ecosystem services at the landscape scale, and how this relates to the distribution of land cover types, using both surveys and online GIS. The objectives of this research were;

- To investigate the relative value of different ecosystem services in the New Forest, in relation to the demographics of the participants.
- To investigate if differences between habitats in the aesthetic, recreation and conservation value of the New Forest exist, using fixed sum questions
- Compare pGIS and fixed-sum scoring methods for differences, as these methods have not been compared in literature.

2.3. Materials and methods

Ethical statement

The participants in this study were self-selecting and chose to voluntarily guide themselves to the website (www.newforestsurvey.co.uk). Individuals were greeted with an introductory page that explained the background of the study, the researcher and their contact details (Appendix I.1). Individuals checked a digital consent box confirming that they understood that the voluntary nature of participation and were made aware that the data would be anonymous, with no responses being identifiable in any reports or publications before being able to proceed with the survey. This study was approved by the Bournemouth University Research Ethics Committee.

Study area

The New Forest covers over 56,000 ha of land in the south of England (New Forest Park Authority (NFPA), 2010), including the counties of Wiltshire and Hampshire with the English Channel directly to the south (Figure 2.3.1). This heterogeneous landscape is mostly unenclosed forest, heath and grazed land that to this day has commoner's grazing rights (Grant and Edwards, 2008). Having attained National Park status in 2005, the New Forest also houses 20 SSSIs, two Ramsar Convention sites and six Natura 2000 sites, encompassing a mix of habitats including woodlands, mires and grassland; with 9 rare and 25 nationally scare vascular plant species (Newton, 2010, New Forest Park Authority (NFPA), 2008). Another highly prized asset are the Ancient and Ornamental woodlands that are the mark of the unique interactions between human activities and the beech and oak ecological dominance in the area (Grant and Edwards, 2008).

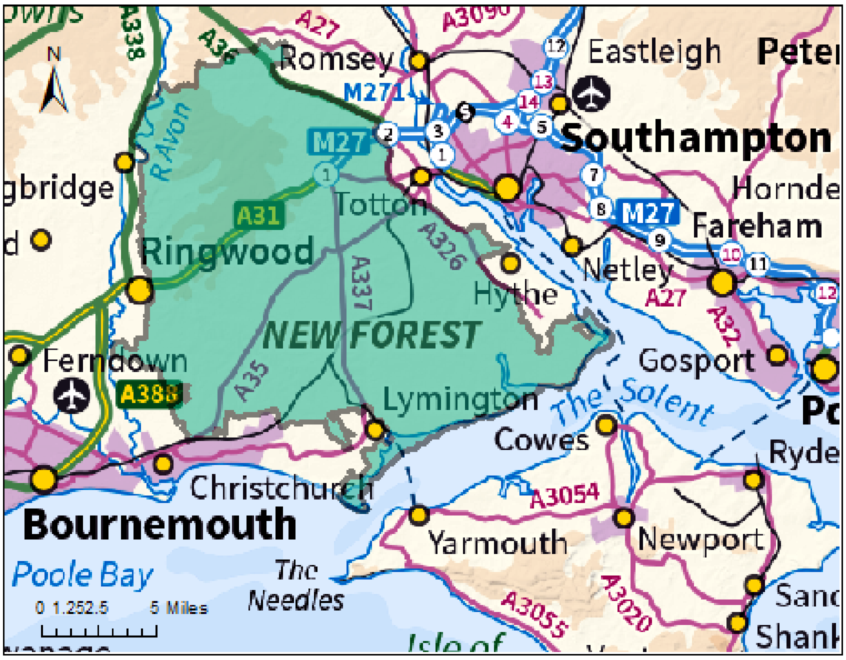

Figure 2.3.1: Map of the New Forest National Park on the south coast of England, with the park area overlaid in green over a Ordnance Survey map in relation to the local area (© Crown Copyright and Database right 2016).

Recreation is important to visitors of the New Forest National Park. With over 13 million visitors every year, it is estimated that visitor spending within the New Forest National Park brings in £107.6 million per year, sustaining 2,451 jobs (New Forest Park Authority (NFPA), 2007), with £8.34 spent per visit. It was found that on average each visitor who lived locally made 257 day visits per year, with other non-locally residing visitors making 45 day trips a year from home each year (New Forest Park Authority (NFPA), 2007).

Questionnaire design

Questionnaires were designed to collection information on (i) demographic and personal information of participants, (ii) the importance to the participant of the New Forest for the following ecosystem services: aesthetic value, carbon storage, crop production, cultural value, flood risk mitigation, livestock production, recreation and timber, (iii) the importance of different habitats in the New Forest for aesthetic value, recreation value and conservation value and (iv) areas in the New Forest that participants valued for cultural (heritage), aesthetic and recreation value (Appendix I.1). The questionnaire design was carefully considered, firstly to capture a wide variety of demographic information, and secondly collecting stakeholder views on ecosystems services, whilst being user-friendly and time efficient.

For (i), the following demographic and personal information was collected: (1) Gender, (2) Age group, (3) Highest level of education completed, (4) Area of residence, (5) Proximity of residence to the New Forest National Park, (6) Interest groups or affiliations and (7) Land or property ownership and management within the New Forest National Park and (8) Agreement to the statement 'I have a good understanding of conservation and ecosystem management issues within the New Forest'.

Fixed-sum scoring questions are a type of ranking format question that asks a participant to score or weight a set of choices that sum to 100, resulting in data that shows the relative ranking within a group and is useful when weighting is needed (Thomas, 2004). For (ii) participants were asked to indicate how important they felt the New Forest was for providing each of the following ecosystem services by allocating 100 points amongst them; aesthetic value, carbon storage, crop production, cultural value, flood risk mitigation, livestock production, recreation and timber. Descriptions of each ecosystem service were provided as a 'pop up' window (Appendix I.2). For (iii) participants were asked to indicate how important they felt different New Forest habitats were for conservation, aesthetic and recreation value by allocating 100 points amongst them; acid grassland, arable cereals, arable horticulture, broadleaved/mixed woodland, coniferous woodland, dwarf shrub heath, fen/marsh/swamp, improved grassland, neutral grassland, suburban/rural developed and urban. These broad habitat types were chosen due to being the most dominant terrestrial habitats in the New Forest, with names being slightly altered to ease public understanding. Certain HBIC categories were split; 'Arable and horticulture' was split into arable cereals and arable horticulture and 'built-up areas and gardens' into suburban and urban (Table 2.3.1).

The Hampshire Biodiversity Information Centre (HBIC) classification for arable horticulture was split into arable cereals and arable horticulture and 'built-up areas and gardens' into suburban and urban. Neutral grassland was included due to the large recreational lawn areas in the New Forest used for recreation. The question was designed so that users could not progress onto the next question until the sums totalled 100. Images of each habitat were provided as a 'pop-up' box to facilitate understanding of the habitat type (Appendix I.3).

A broad habitat vector shape-file was acquired under a data supply agreement from Hampshire Biodiversity Information Centre (HBIC, Hampshire County Council, Winchester) dated 1 February 2015 (job number: 5335). This was converted to a raster to allow later analysis against raster ecosystem service values, using the ‘Polygon to raster’ conversion tool, setting the cell size to 25 m x 25 m using the maximum combined area to represent the pixel’s habitat type the most appropriately (pixel counts can be seen in Table 2.3.1). The CEH landcover map was not used due to the HBIC map found to be more accurate for habitat type using a 100 random point check within the boundaries of the New Forest National Park.

Table 2.3.1: Hampshire Biodiversity Information HBIC broad habitat classifications (with raster dataset pixel counts) used in the participatory GIS mapping task, compared to habitats types used in the fixed sum survey questions. The most dominant land cover types were used for the latter, with the exclusion of littoral sediment and unidentified habitat. *These HBIC habitat types were further split.

HBIC Broad Habitat Classification used for participatory GIS analysis	Raster pixel count (25 x 25 m	Comparative habitat used for fixed sum-scoring questions
Broadleaved, mixed, and yew woodland	228696	Broadleaved/ mixed woodland
Dwarf shrub heath	159485	Dwarf shrub heath
Improved grassland	158785	Improved grassland
Coniferous woodland	79183	Coniferous woodland
Arable and horticulture*	72622	Arable cereals & Arable horticulture
Acid grassland	50665	Acid grassland
Fen, marsh and swamp	48486	Fen/marsh/swamp
Built-up areas and gardens*	41330	Suburban/rural developed & Urban
Littoral Sediment	20141	-
Unidentified habitat	17167	-
Neutral grassland	16474	Neutral grassland
Boundary and linear features	6195	-
Bracken	4842	-
Unidentified water	2292	-
Rivers and streams	1743	-
Standing open water and canals	1264	-
Supralittoral Sediment	879	-
Inland rock	521	-
Inshore sublittoral sediment	521	-
Littoral Rock	42	-
Supralittoral Rock	9	-
Calcareous grassland	2	-

For (iv) participants were asked place ‘pins’ onto a map of the study area to highlight locations of cultural, aesthetic and recreation value. Each individual could place up to five pins for every service, and had to score the pin from 1-5, most important to least important (Figure 2.3.2). The nature of the online participatory GIS method allowed the study site to be encapsulated in its entirety, using the New Forest National Park boundary. The map allowed the individual to zoom in and out freely, though only allowed pins to be placed within the boundary. The participant was not required to place all five pins, but could only place one for each score, i.e. one 5, one 4, etc. For example, a participant could place pins of score 5, 4, 3, 2, 1 for recreational value but only three pins of 5, 2 and 1 for cultural services. The ‘cultural value’ term was used for ease of participant understanding and was described as being areas with historical or cultural important including historical monuments and memorials, locations featured in artistic works or literature, and sites of historical events.

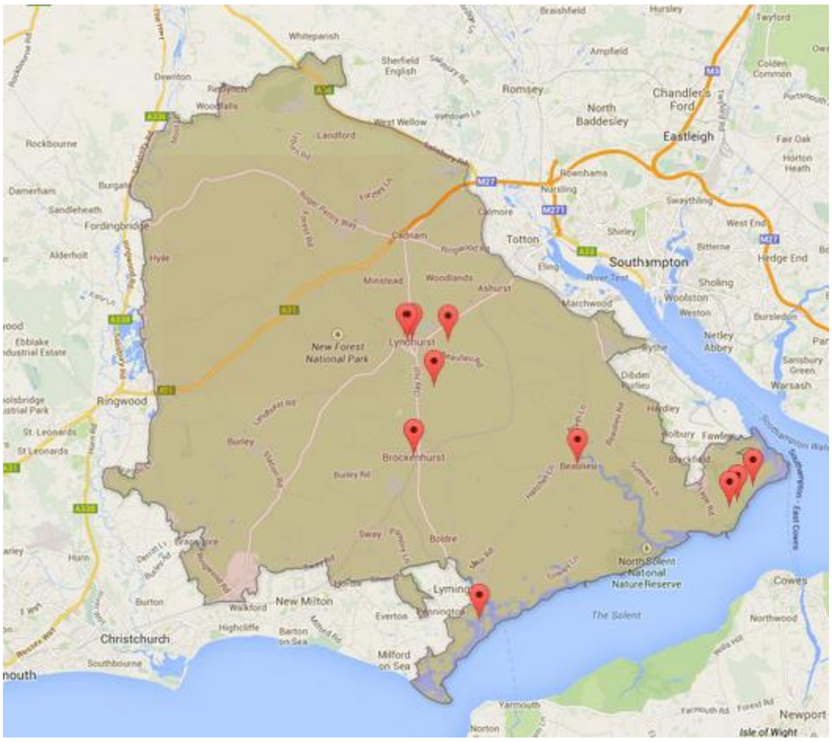

Figure 2.3.2: Example map with locational pins denoting areas of value for an example survey participant, each pin also stored whether it is cultural, aesthetic or recreation value, and a score from 1 to 5, from most important to least important (not shown).

Questionnaire Construction

The questionnaire was designed using Adobe Dreamweaver CS6 (Adobe Systems Europe Ltd, Maidenhead, UK) and HTML and PHP coding using the standard text editor Notepad v6.1 (Microsoft Corporation, Reading, UK). Dreamweaver was used due to its robust webpage editing capabilities. Background maps ran through Google Maps API version 3 (Google UK Ltd, London, UK) as part of the participatory GIS question. Web hosting and domain purchase of www.newforestsurvey.co.uk were managed though 1&1 Internet Limited (Slough, UK) with the website files being uploaded to the web-space with Filezilla v.3.10.2 (Filezilla Project, 2016). Participant data was stored within a MySQL database on the web-space (Appendix I.1, Figure I.12).

Survey method

The survey site was live between January 2014 and July 2014. The study was self-selecting, with participants responding to circulated emails (either directly from the author or forwarded on) sent to every primary and secondary school in the New Forest boundary requesting details be passed onto teachers and parents, and through their newsletters and local interest and recreational groups in the area. The survey was also promoted through the Bournemouth University Research blog and a letter in the Lymington Times in February 2014 (Appendix I.4).

Data and statistical analysis

Fixed-sum survey data for scoring of the eight ecosystem services (aesthetic value, carbon storage, crop production, cultural value, flood risk mitigation, livestock production, recreation and timber) and aesthetic, recreation and conservation value (acid grassland, arable cereals, arable horticulture, broadleaved/mixed woodland, coniferous woodland, dwarf shrub heath, fen/marsh/swamp, improved grassland, neutral grassland, suburban/rural developed and urban) were tested for normality using the Kolmogov-Smirnov test. Due to data being non-parametric, significance of differences were tested using the Kruskal-Wallis H test.

To determine differences between individual ecosystem services and habitats, post-hoc pairwise comparisons were used. Results were reported as reversed axis boxplots for aesthetic, recreation and conservation value to facilitate interpretation of results, with post-hoc test results shown through bars grouped by the same letter not being significantly different from each other.

Correlation between aesthetic, recreation and conservation value was tested across all habitat types and individually for all habitat types using Spearman's rank-order correlation coefficient (rho), with clustered scatterplots being used to visually represent the data. Differences between sub-groups for scoring ecosystem services was explored through the use of appropriate tests depending on the amount of categories; Mann-Whitney U (for gender) and Kruskal-Wallis H test (area of residence, proximity to the New Forest National Park, area of residence and education level). Mann-Whitney U was used to test the multiple categories groups of interest group and organisational affiliation and land or property ownership or management in the New Forest National Park, due to the categories not being mutually exclusive. Only significant results were reported.

IBM SPSS Statistics v.22 (IBM United Kingdom Limited, Portsmouth, UK) was used for statistical analysis and SigmaPlot 13 (Systat Software Inc, London, UK), Excel 2010 (Microsoft Corporation, 2010) and IBM SPSS for data visualisation.

GIS analysis

Data was exported from the MySQL database into CSV file format, with any data with missing scores, or missing service (i.e. cultural, recreation or aesthetic) removed. This left a total of 786 markers across all services; 268 for aesthetic, 233 for cultural and 285 for recreation, with total points of 862, 775 and 896 respectively.

The scores were inverted to give correct weighting, as survey participants used the scale of 1-5, with 1 being most important. These point values were then divided by total point values. The latitude, longitude and score for each point, for each of the services were imported into ArcMap 10.1 (ESRI UK Limited, Aylesbury, UK) using the 'Add XY data' tool. Point data were projected to using British National Grid (OSGB 36) using the Edina Digimap recommended transformation for accuracy, the Ordnance Survey OSTN02 transformation (OSGB_1936_To_WGS_1984_7)' (University of Edinburgh, 2015) from the Google Maps API default projection of WGS84.

Individual point score was divided by total points for each ecosystem service. A Kernel density analysis (a quadratic kernel function) was conducted on this score divided by total points data points using the 'Kernel Density' spatial analyst tool using the kernel density methodology (Silverman, 1986). This methodology has been used by studies in social mapping for example by Sherrouse et al. (2011) and Alessa et al. (2008). Each individual score was divided by total points, acting as a weighting factor. The densities and planar methods were used with a 25 m by 25 m cell size to match the land cover layer. The function created circular areas around each point and outputted a smoothed surface map (which was set to the extent of the boundary of the New Forest National Park. Kernel density layers were run with aesthetic, recreation and cultural markers independently, replicating each with a search radius of 500 m, 1000 m and 2000 m for sensitivity analysis. The 1000 m recreation and cultural maps, and 2000 m aesthetic maps were selected, as Newton et al. (2012) found through a stakeholder consultation that 1000 m for recreation and cultural valuation and 2000 m for aesthetic values was most appropriate.

There raster maps were then normalised using raster calculator, dividing each raster by the largest value, to get rasters with normalised values between 0 and 1 (with no 'No Data' cells). The 'zonal statistics as table' tool was then used to calculate statistics, using the HBIC raster as the zone layer (using habitat type as the zone field) and the cultural service maps as the value layer. Aesthetic service had a search radius of 2000m (with a total area of: $\pi \times 2000^2 = 12566370.614$) compared to 1000m for cultural and aesthetic (with a total area of: $\pi \times 1000^2 = 3141592.653$), making the aesthetic service have an areas four times as large. Hence aesthetic value was divided by 4 to make it comparable to the other ecosystem services. The Friedman test was used for comparing the mean aesthetic, recreation and cultural participatory GIS results. Correlation between fixed-sum scoring and pGIS for aesthetic and recreation value was tested using Spearman's rank-order correlation coefficient (rho) and visually represented as a scatterplot.

2.4. Results

A total of 132 surveys were completed fully online, with the online participatory GIS task yielding 786 pins placed within the boundaries of the New Forest National Park. The choice of the number of pins (up to a maximum of five for each ecosystem service) were entirely the participants; most pins were placed for areas for recreation (285) and aesthetic value (268), with the least for cultural value (233).

Demography of participants

The surveys were completed by slightly more females at 51.5%, with a considerable distribution of ages with the largest age bracket of 18-30 year olds (31.1%), following by 51-60, 31-40 and 41-50 year olds, with 20.5%, 19.7% and 17.4% respectively. The survey had the least participants from 61-70 and 71+ year olds (with 6.8% and 4.5% respectively).

Most participants lived in towns (44.7%), followed by villages (33.3%) and cities (13.6%), with only 8.3% living in the countryside. Of these 37.9 % lived 10 – 30km away from the National Park, 28.8% within the National Park boundaries, 25.8% within 10km of the National Park and 7.6% more than 30km away from the National Park.

Participants were asked about their level of education, with 42.4% having completed a postgraduate course (MSc/ PhD or equivalent), following by 34.1 having completed higher education (BSc or equivalent), 13.6 further education (A-level or equivalent) and 9.8% secondary education (GCSE or equivalent).

Understanding of issues in the New Forest National Park

The majority of participants expressed an understanding of the conservation and environmental issues in the New Forest National park, with a 56.1% either agreeing or agreeing strongly compared with 21.2% either disagreeing or disagreeing strongly (Figure 2.4.1).

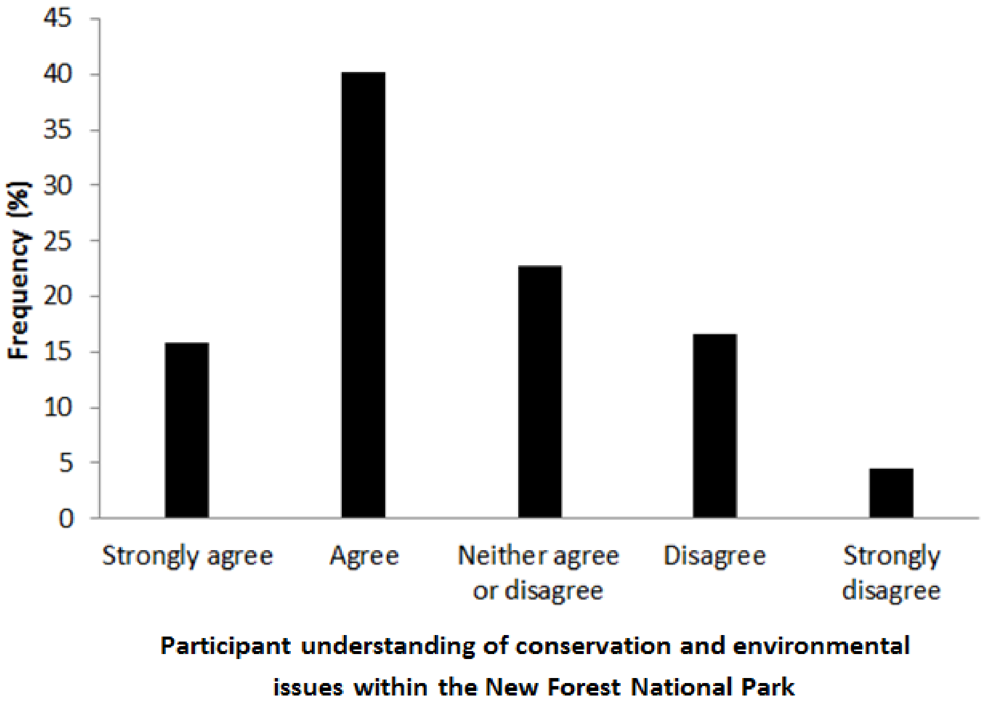

Figure 2.4.1: Bar graph illustrating frequency scores for participant self-assessed understanding of conservation and environmental issues within the New Forest National Park.

Weighting of ecosystem services

All eight ecosystem services had distributions significantly deviating from normal using Kolmogorov-Smirnov test for normality (Appendix I.5). The distribution of scores was found to be significantly different between ecosystem services (Kruskal-Wallis H test, $\chi 2(7) = 235.582$, $p < 0.001$, 2-sided) (Figure 2.4.2). Post-hoc pair-wise comparisons show that Crop production had a significantly different median score to all other ecosystem services, scoring the lowest median of 5. Aesthetic value and recreation value were not significantly different, but scored the highest medians, 20 and 18 respectively. All other services had medians of 10, with carbon storage, cultural value, flood risk mitigation and livestock production not having significantly differing means. Timber, flood risk mitigation and livestock production were also not significantly different (Figure 2.4.2).

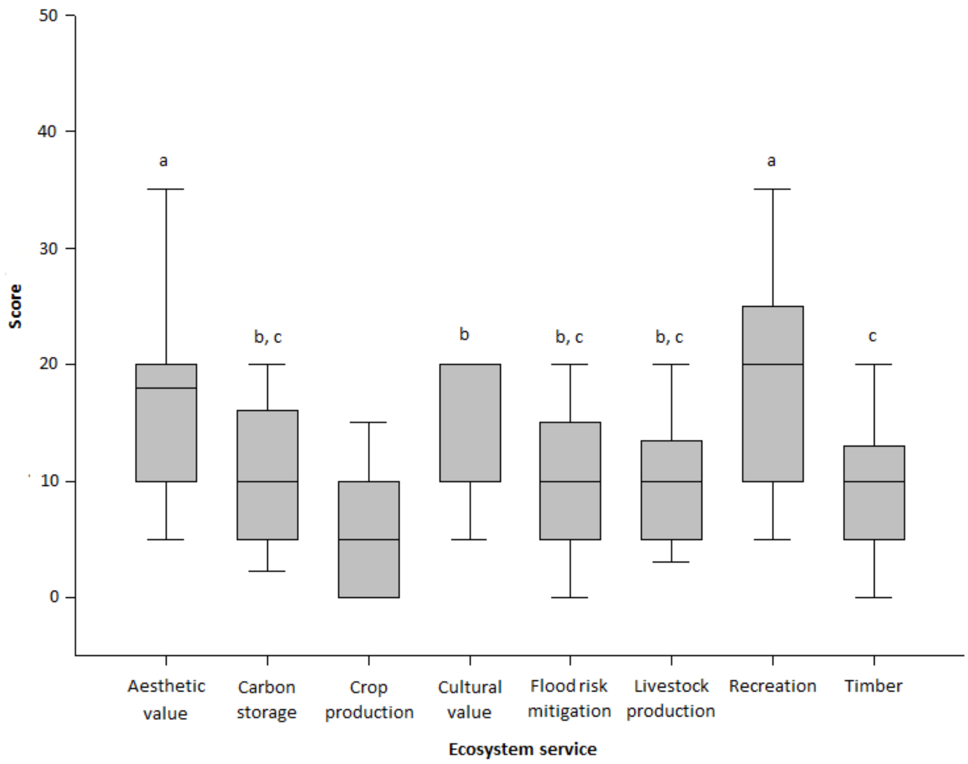

Figure 2.4.2: Boxplot illustrating the values assigned to different ecosystem services by participants. The overall difference between the median ranks of ecosystem services was significant (Kruskal-Wallis H test, $\chi^2(7) = 235.582$, $p < 0.001$, 2-sided). Bars grouped by the same letter are not significantly different from each other (pairwise comparisons, $p < 0.05$).

Difference in participant sub-groups scoring the importance of ecosystem services

Participants were asked whether they owned or managed any property in the New Forest National Park, with multiple choices being allowed, hence percentages add up to over 100%. Most participants did not own any property or land at 65.2%, with 16.7% of participants owning private property, 6.1% having commoner's rights and 1.5% managing private property. Commercial property was owed by 5.3% and managed by 0.8%.

Interest group and organisation affiliation was common; these were again multiple choice questions, hence percentages do not add up to 100%. Affiliation with groups was highest with 31.1% nature conservation management, followed by 9.1 % local government, 8.3% tourism, 6.8% agricultural production, 6.8% forestry, 6.1% residential 4.5% fisheries, 3.0% game management and 1.5% water production.

The median scores for the importance of aesthetic value as an ecosystem service were different for gender with a median of 15 for females and 20 for men, with the distributions differing significantly (Mann-Whitney U = 2689.500, n1 = 68, n2 = 64, $P < 0.05$ two-sided). Median scores for the importance of livestock production value as an ecosystem value were a median of 10 for women and 7.5 for men, with the distributions differing significantly (Mann-Whitney U = 1603.500, n1 = 68, n2 = 64, $P < 0.01$ two-sided). No other significant differences across the remaining six ecosystem services.

Area of residence affected median scores for the importance of livestock production value with highest values (median 15) given by those living in the countryside and lowest values (median 5) given by those living in cities, with town and village dwellers giving a median value of 10 to this service (Kruskal-Wallis H test, $\chi2(3) = 10.052$, n = 132, $p < 0.05$ two-sided). Post hoc tests showed that those in the city and village significantly valued livestock differently (Mann-Whitney U = -29.246, p= 0.031) as did city and countryside dweller (Mann-Whitney U = -38.735, p = 0.040).

Proximity to the New Forest National Park was also found to impact the median scores for the importance of livestock production value as an ecosystem value were 10 for 10 – 30 km away, 8.5 for more than 30km away, 8.5 for within 10km of the NFNP and 11 within the NFNP, with the distributions differing significantly significant (Kruskal-Wallis H test, $\chi2(3) = 8.281$, n = 132, $p < 0.05$ two-sided). Post-hoc tests showed a significant difference in scoring of those who lived within 10 km of the NFNP and those within the boundaries of the NFNP (Mann-Whitney U = -24.995, p = 0.027).

Involvement with affiliations and interest groups had varied impacts on the value placed on various ecosystem services. An involvement with nature conservation management led to a cultural value median of 20, compared to 10 if not involved (Mann-Whitney U = 2444.500, n1=91, n2=41, P<0.01 two-sided). Agricultural production affiliations found the livestock median to be 15, compared to 10 if not affiliated (Mann-Whitney U = 817.500, n1 = 123, n2 = 9, P<0.05 two-sided). Forestry affiliations scored a median of 20, compared to 10 if not affiliated (Mann-Whitney U = 769.500, n1 = 123, n2 = 9, P<0.05 two-sided). If participants had an affiliation to water production, cultural value was rated 25 compared to 10 if not (Mann-Whitney U = 234.500, n1=130, n2 = 2, P<0.05 two-sided). Residential affiliations found flood risk median 18.50 compared to 10 if not (Mann-Whitney U = 716.000, n1 = 124, n2 = 8, P<0.05 two-sided) and recreation value rated 10, otherwise 20, differently from how you rated other services (Mann-Whitney U = 198.000, n1 = 124, n2 = 8, P < 0.01 two-sided)

Property and land ownership or management impacted weighting of ecosystem services with private property owners having a livestock median of 16, compared to 10 if not an owner (Mann-Whitney U = 1627.000, n1 = 110, n2 = 22, P < 0.01 two-sided). Commercial property owners had a median cultural value of 20 compared to 10 (Mann-Whitney U = 673.500, n1 = 125, n2 = 7, P < 0.05 two-sided). Participants who did not own property had a flood risk median of 7.5, compared to 10 (Mann-Whitney U = 1503.500, n1 = 46, n2 = 86, P < 0.05 two-sided).

Importance of aesthetic, recreation and conservation value by habitat type

The distributions of aesthetic, recreation and conservation value significantly deviating from normal in all habitats using Kolmogorov-Smirnov test for normality (Appendix I.5), so non-parametric tests were used to look for differences between habitats.

Importance of habitat types for aesthetic value

The distribution of scores was found to be significantly different between habitat types for aesthetic value (Kruskal-Wallis H test, $\chi2(10)$ = 650.326, p < 0.001, 2-sided) (Figure 2.4.3). Post-hoc pair-wise comparisons showed that broad-leaved and mixed woodland was the only category to be significantly different from all other habitat types (Figure 2.4.3). Arable cereal and arable horticulture, urban and suburban, rural developed and improved grassland did not different significantly from each other. Dwarf shrub heath was significantly different from all habitats types except for fen/marsh/swamp and coniferous woodland. Neutral grassland was not significantly different from acid grassland, though both were significantly different from improved grassland.

The highest values can be seen for broadleaved/mixed woodland and dwarf shrub heath with medians of 20 and 15 respectively. The lowest medians are for suburban/rural developed with a median of 2 and urban with 0, followed by arable cereals at 2.5 and arable horticulture at 4. Improved grassland had the lowest median of the grasslands compared to the neutral grassland median of 10 and acid grassland median of 9.

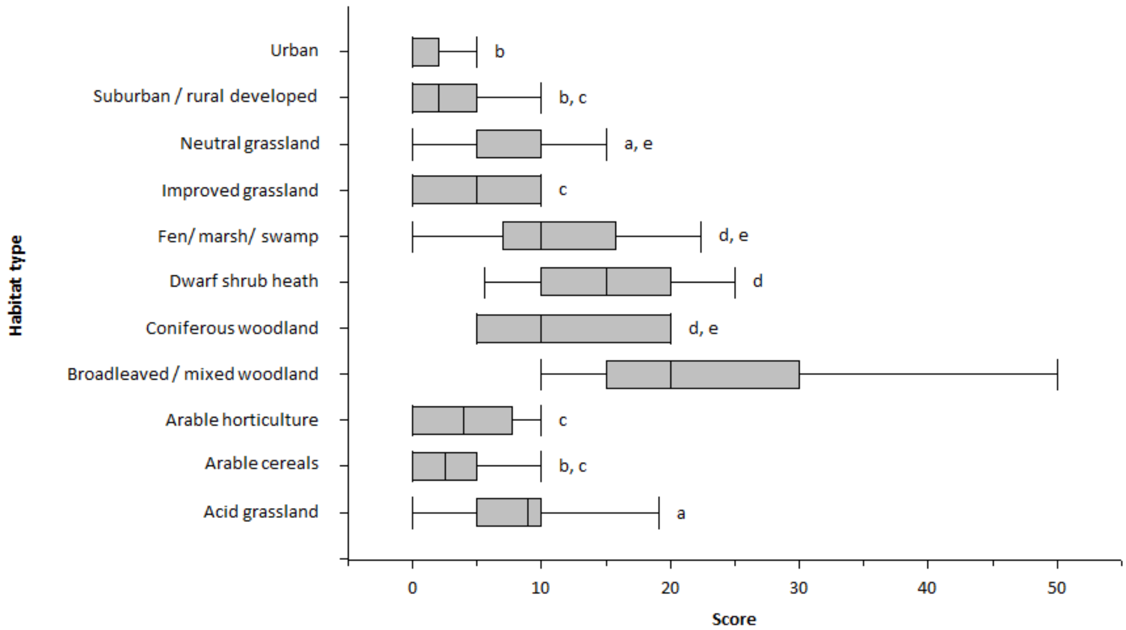

Figure 2.4.3: Boxplot illustrating the scores assigned to different habitat types for aesthetic value by participants. The overall difference between the median ranks of ecosystem services was significant (Kruskal-Wallis H test, $\chi2(10) = 650.326$, $p < 0.001$, 2-sided). Bars grouped by the same letter are not significantly different from each other (pairwise comparisons, $p < 0.05$).

Importance of habitat types for recreation value

The distribution of scores was found to be significantly different between habitat types for recreation value (Kruskal-Wallis H test, χ2(10) = 554.531, p < 0.001, 2-sided) (Figure 2.4.4). Post-hoc pair-wise comparisons showed broadleaved/mixed woodland and coniferous woodland did not differ significantly for recreation value. Neutral, improved and acid grassland were found to not significantly differ. Arable cereals and arable horticulture were found not to differ, though urban and suburban/rural developed did differ significantly.

Broadleaved/mixed woodland had the highest median of 20, followed by coniferous woodland with 15. The lowest medians belonged to urban, arable cereals and arable horticulture with 0, followed by suburban with 4. Improved and neutral grassland both had medians of 10, with acid grassland at 5.5.

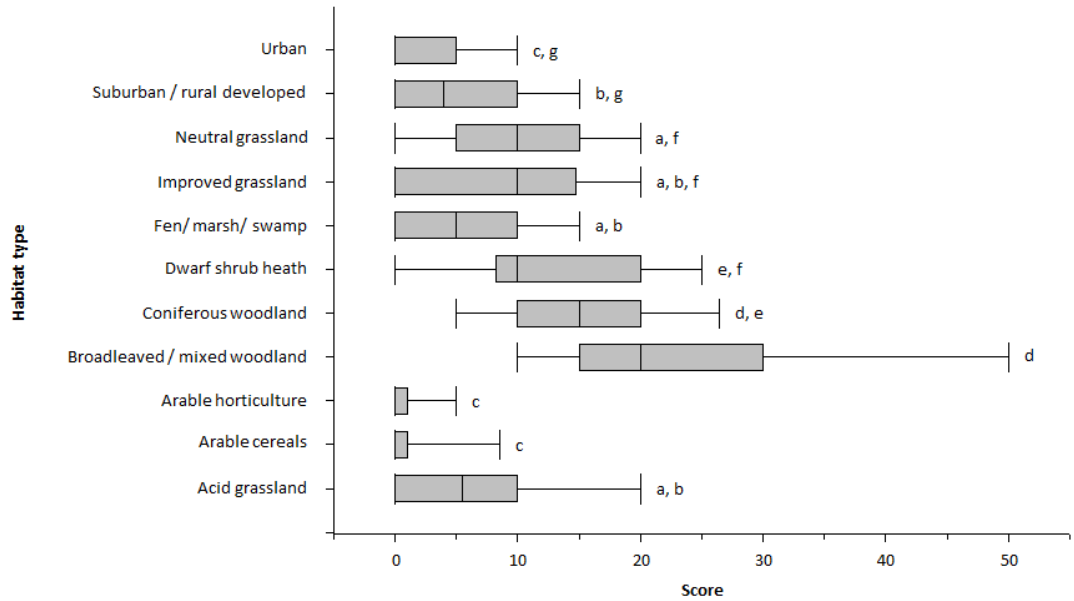

Figure 2.4.4: Boxplot illustrating the scores assigned to different habitat types for recreation value by participants. The overall difference between the median ranks of ecosystem services was significant (Kruskal-Wallis H test, χ2(10) = 554.531, p < 0.001, 2-sided). Bars grouped by the same letter are not significantly different from each other (pairwise comparisons, p <0.05).

Importance of habitat types for conservation value

The distribution of scores was found to be significantly different between habitat types for conservation value (Kruskal-Wallis H test, $\chi2(10) = 743.994$, $p < 0.001$, 2-sided) (Figure 2.4.4). Post-hoc pair-wise tests show that broadleaved/mixed woodland did differ significantly from coniferous woodland, though not dwarf shrub heath. Once again arable cereals and horticulture are not significantly different. Neutral grassland was not significantly different from acid grassland, though both were significantly different from improved grassland.

Broadleaved/mixed woodland had the highest median of 20, followed by fen, marsh and swamp with 15. Urban and suburban/rural developed scored 0 medians. Arable cereals and arable horticulture scored low medians of 0.5 and 2 respectively. Acid and neutral grassland both had medians of 10, with improved grassland at 5.

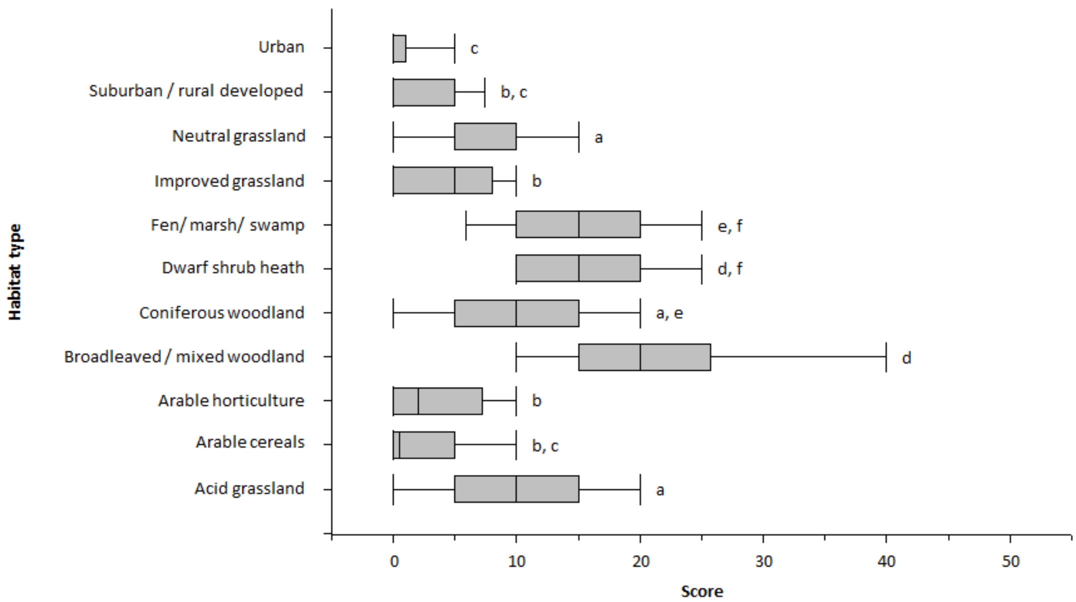

Figure 2.4.5: Boxplot illustrating the scores assigned to different habitat types for conservation value by participants. The overall difference between the median ranks of ecosystem services was significant (Kruskal-Wallis H test, $\chi2(10) = 743.994$, $p < 0.001$, 2-sided). Bars grouped by the same letter are not significantly different from each other (pairwise comparisons, $p < 0.05$).

Comparison and correlation of values for aesthetic, recreation and conservation value

Aesthetic, recreation and conservation medians appear to be consistent within habitat types (Figure 2.4.6). Broadleaved/mixed woodland had the highest medians of 20 across all three ecosystem services; neutral grassland can be seen to consistently score a lower median of 10 and urban a 0 median across all three ecosystem services, with no other habitats keep the same median across all ecosystem services. Arable cereals, arable horticulture suburban/rural developed and urban score had the lowest medians. Coniferous woodland and dwarf shrub heath have values between 10 and 15.

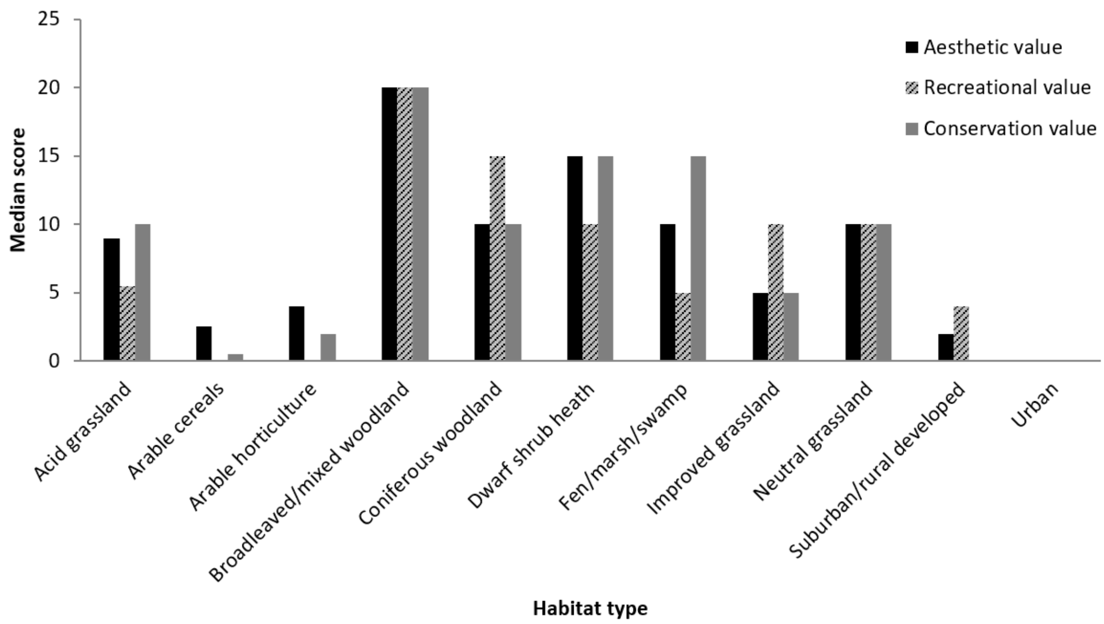

Figure 2.4.6: Clustered bar graph illustrating the aesthetic, recreation and conservation median values from fixed-sum survey questions.

Strength of association between aesthetic, recreation and conservation value were conducted using Spearman's rank-order correlation coefficient, with significant positive monotonic relationships being reported for all combinations (Table 2.4.1). Aesthetic and conservation value have a strong relationship (rho = 0.750) with aesthetic and recreation, and conservation and recreation having moderate relationships (rho = 0.591 and rho = 0.569 respectively). Participants placed high values for all correlation combinations for broadleaved and mixed woodland, with low scores for all services for urban areas (Figure 2.4.7 - Figure 2.4.9).

Further testing for strength of association using Spearman's rank-order correlation coefficient shows several moderate to strong relationships between habitats (Table 2.4.2). Arable cereals and arable horticulture can be seen to exhibit the same pattern of relationship, being strong between aesthetic/conservation, moderate between aesthetic/recreation and recreation/conservation. Acid grassland shows a strong relationship between aesthetic/conservation, moderate relationships between aesthetic/recreation and recreation/conservation. Dwarf shrub heath shows a strong relationship between aesthetic/conservation, moderate relationship between aesthetic/recreation and weak between recreation/conservation. Broadleaved/mixed woodland, coniferous woodland and urban show a moderate relationship across all combinations. Suburban/rural developed and improved grassland show a moderate relationship between aesthetic/ conservation and weak relationships between aesthetic/recreation and recreation/conservation. Fen, marsh and swamp exhibits a moderate relationship between aesthetic/conservation, though is weak between aesthetic/recreation and indeed the only very weak relationship of the entire matrix is between recreation/conservation. Neutral grassland consistently correlates weakly for all combinations of the services. All habitat types show a moderate to strong relationship for aesthetic/conservation apart from neutral grassland. Moderate to weak relationships can be seen for all aesthetic/recreation and recreation/conservation, apart from fen, marsh and swamp that exhibits a very weak relationship

Table 2.4.1: Correlations between aesthetic, recreation and conservation scores (n=1452) for all combined habitat types. Results show significant (* P < 0.05; ** P < 0.01; *** P < 0.001) or non-significant (n.s.) relationships.

Spearman's rank-order correlation (rho)	*Aesthetic*	*Recreation*	*Conservation*
Aesthetic	-	0.591**	0.750**
Recreation	0.591**	-	0.569**
Conservation	0.750**	0.569**	-

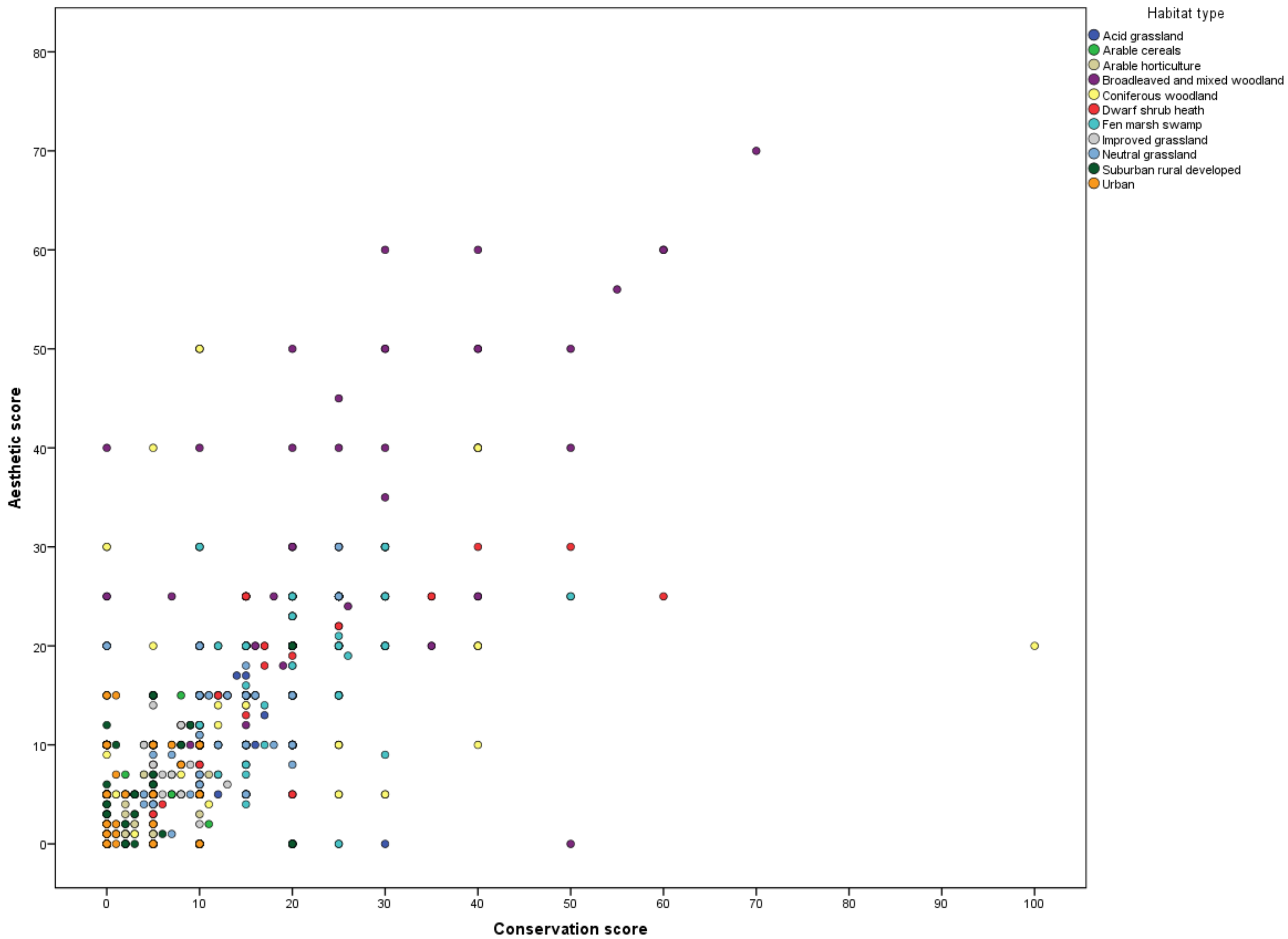

Figure 2.4.7: Scatterplot illustrating the correlation between aesthetic score and conservation score given by participants for each habitat type in the fixed sum scoring survey question.

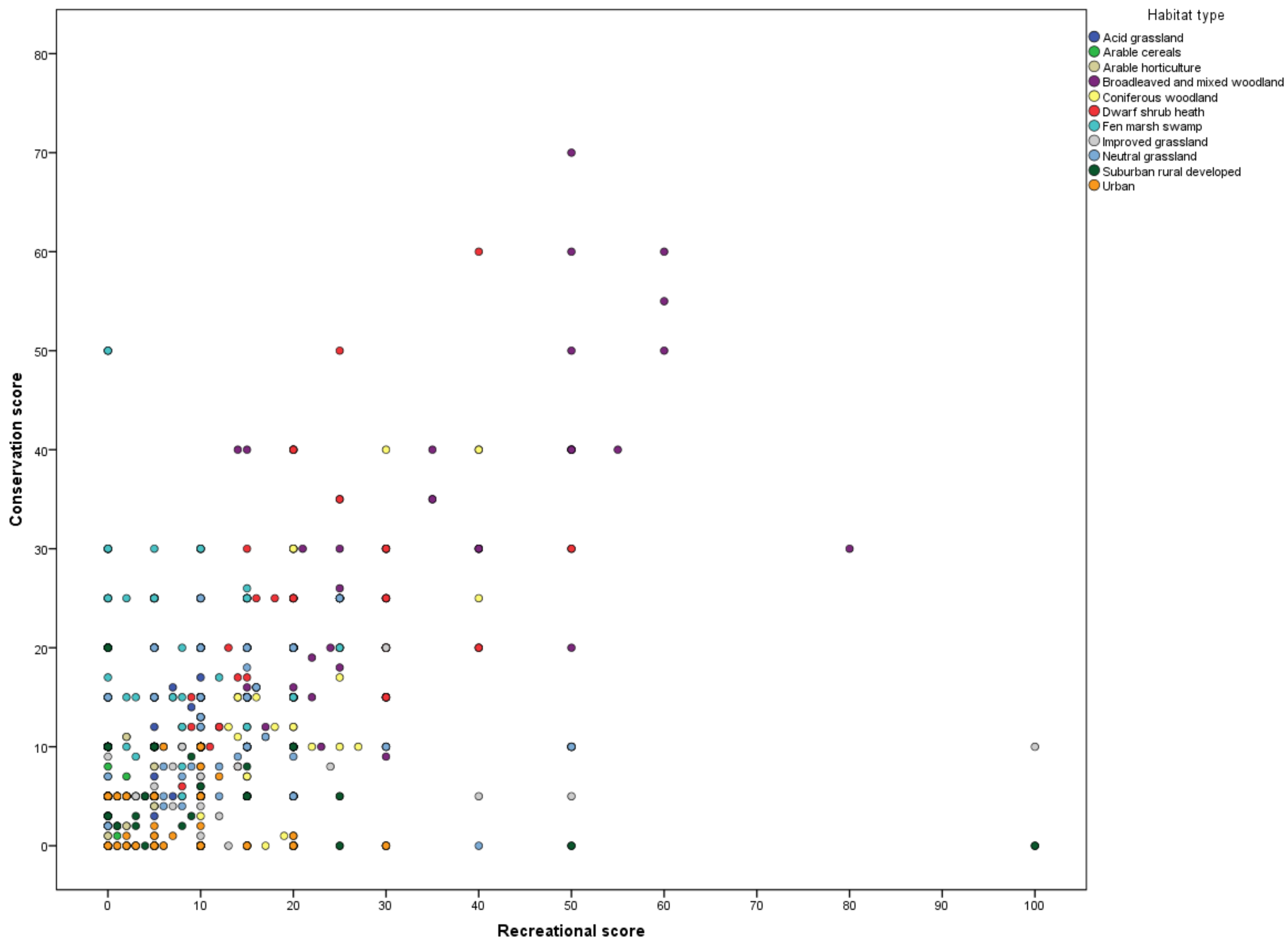

Figure 2.4.8: Scatterplot illustrating the correlation between conservation and recreation score given by participants for each habitat type in the fixed sum scoring survey question.

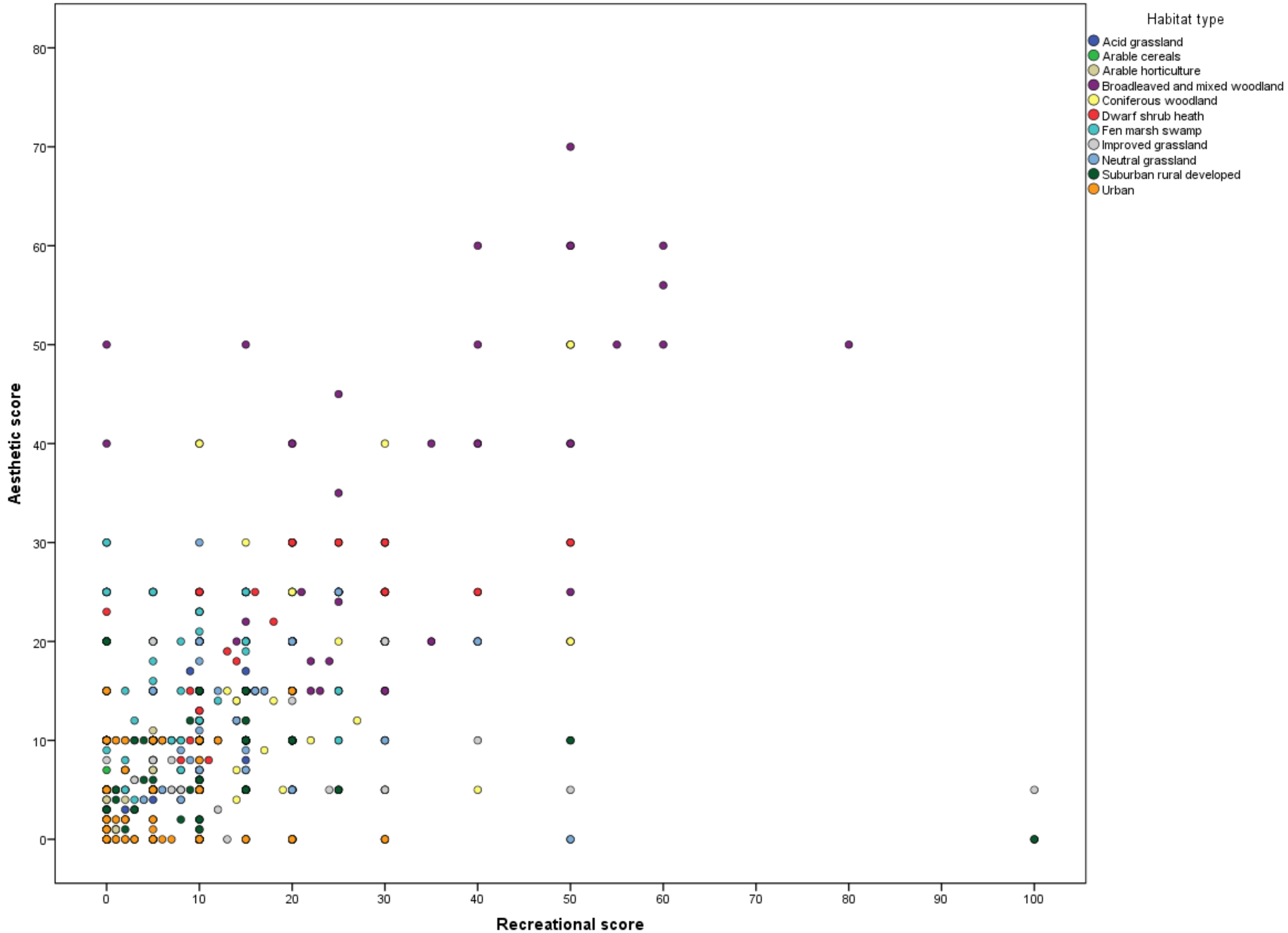

Figure 2.4.9: Scatterplot illustrating the correlation between aesthetic and recreation score given by participants for each habitat type in the fixed sum scoring survey question.

Table 2.4.2: Correlations between aesthetic, recreation and conservation scores were tested for significance using Spearman's rank correlation for each habitat type (n=132). Results show significance (* P < 0.05; ** P < 0.01; *** P < 0.001) or non-significant (n.s.) relationships.

	Spearman's rank-order correlation (rho)		
Habitat type	*Aesthetic/ Conservation*	*Aesthetic/ Recreation*	*Recreation/ Conservation*
Acid grassland	0.618 ***	0.584 ***	0.598 ***
Arable cereals	0.615 ***	0.433 ***	0.532 ***
Arable horticulture	0.682 ***	0.431 ***	0.519 ***
Broadleaved/mixed woodland	0.525 ***	0.443 ***	0.497 ***
Coniferous woodland	0.593 ***	0.462 ***	0.518 ***
Dwarf shrub heath	0.650 ***	0.469 ***	0.396 ***
Fen/marsh/swamp	0.579 ***	0.239 **	0.193 *
Improved grassland	0.527 ***	0.387 ***	0.351 ***
Neutral grassland	0.392 ***	0.382 ***	0.340 ***
Suburban/rural developed	0.565 ***	0.372 ***	0.340 ***
Urban	0.558 ***	0.440 ***	0.578 ***

Online participatory GIS (pGIS) task

Participants placed pins across the whole of the New Forest, though accumulations of points can be seen around towns, the central region and southern coastal area (Figure 2.4.10). Recreational scores can be seen to cluster around the towns of Burley, Lyndhurst, Brockenhurst and Beaulieu, along the coast and more open spaces (Figure *2.4.13*). Cultural scores show the largest clustering around towns, with Burley, Brockenhurst and Beaulieu featuring heavily. Though cultural value was also been placed in other areas away from towns, and focused along the southern tip of the coast (Figure 2.4.15). Aesthetic scores show less clustering around towns, though Beaulieu is the exception. The distribution of pins is prolific in areas away from towns and the coast (Figure 2.4.11). Multiple search radii for the kernel density analysis showed that the overall visual pattern remained unchanged (Figure 2.4.17.)

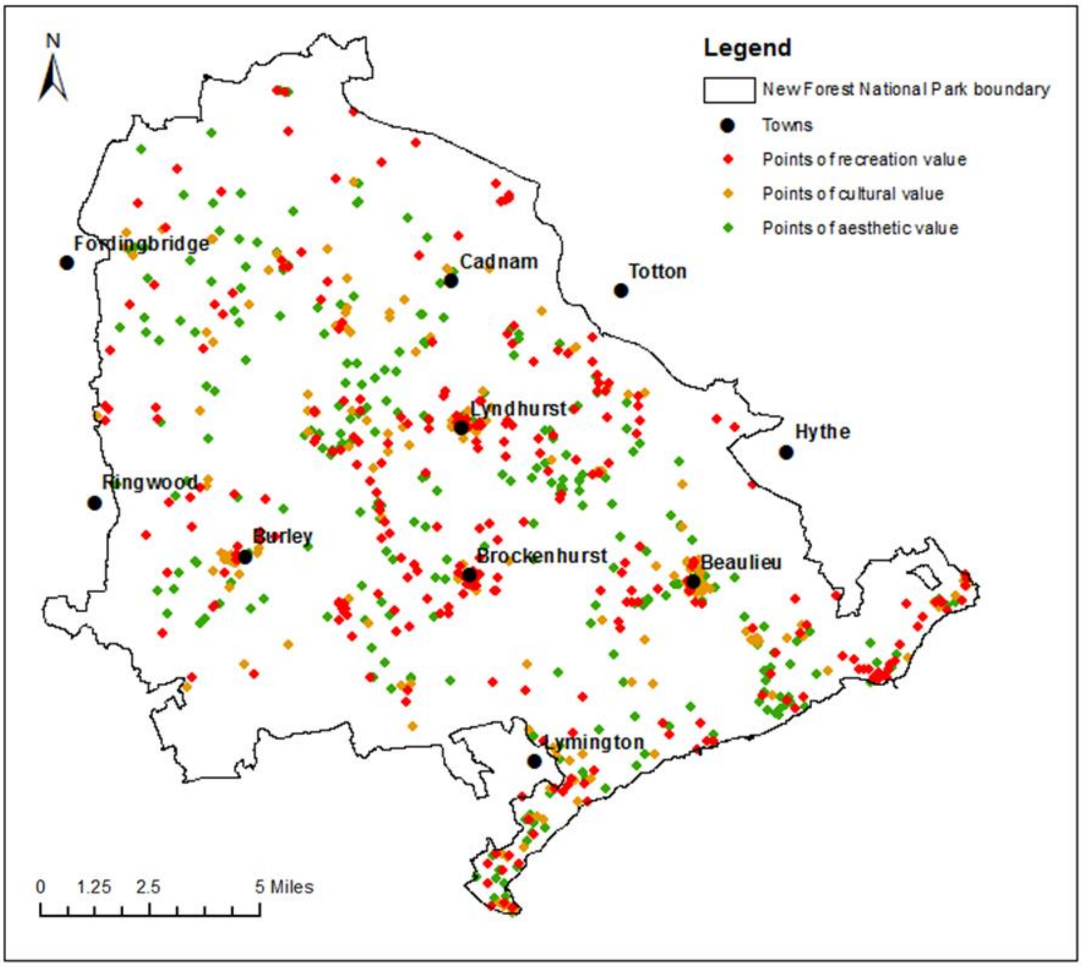

Figure 2.4.10: Map showing all aesthetic, recreation and conservation pins placed by participants across the New Forest National Park during the participatory GIS mapping exercise (n=786 markers across all services; 268 for aesthetic, 233 for cultural and 285 for recreation, with total point summed scores of 862, 775 and 896 respectively).

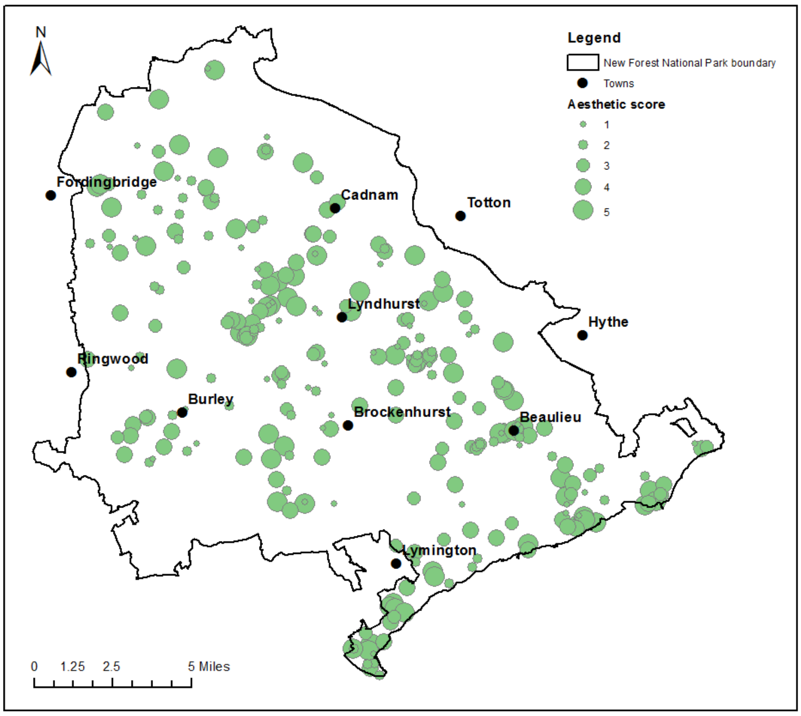

Figure 2.4.11: Map showing the aesthetic value pins placed by participants across the New Forest National Park during the participatory GIS mapping exercise (n=268, with a total summed point score of 862) represented with graduated coloured circles.

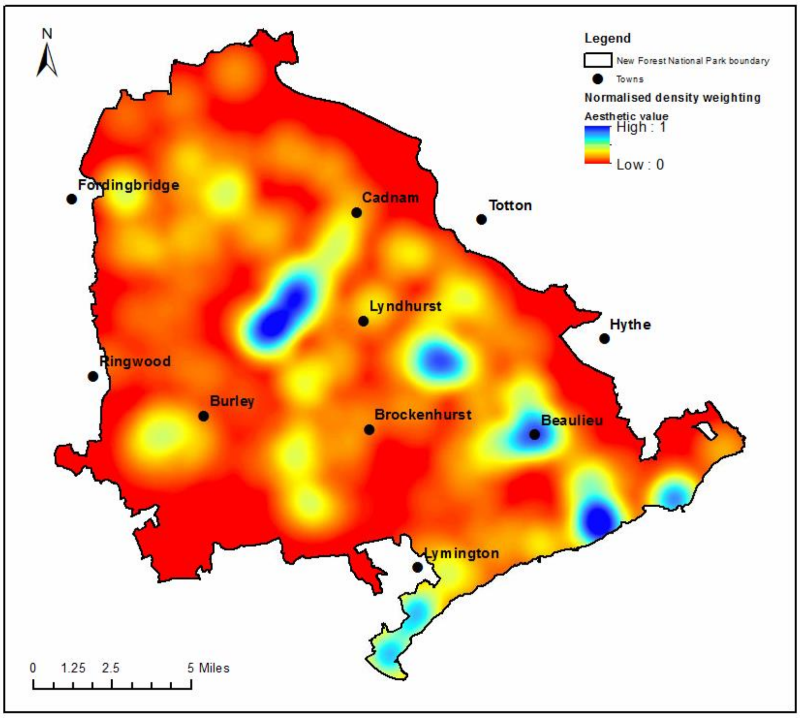

Figure 2.4.12: Kernel density map of normalised (0 - 1) values for aesthetic value showing areas with low to high value in the New Forest National Park based on a 25m grid cell size and 2000m search radius.

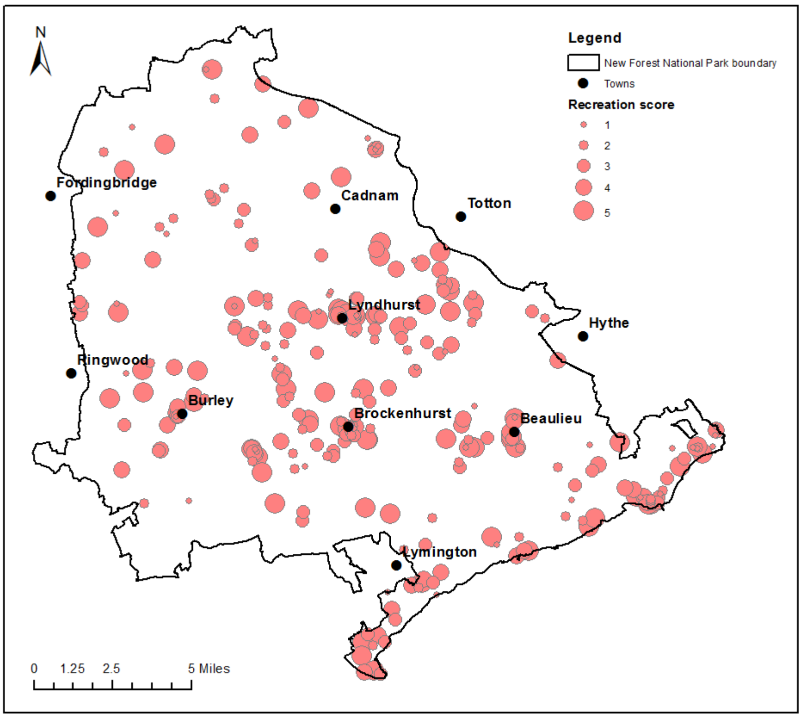

Figure 2.4.13: Map showing the recreation value pins placed by participants across the New Forest National Park during the participatory GIS mapping exercise (n=285, with a total summed point score of 896) represented with graduated coloured circles.

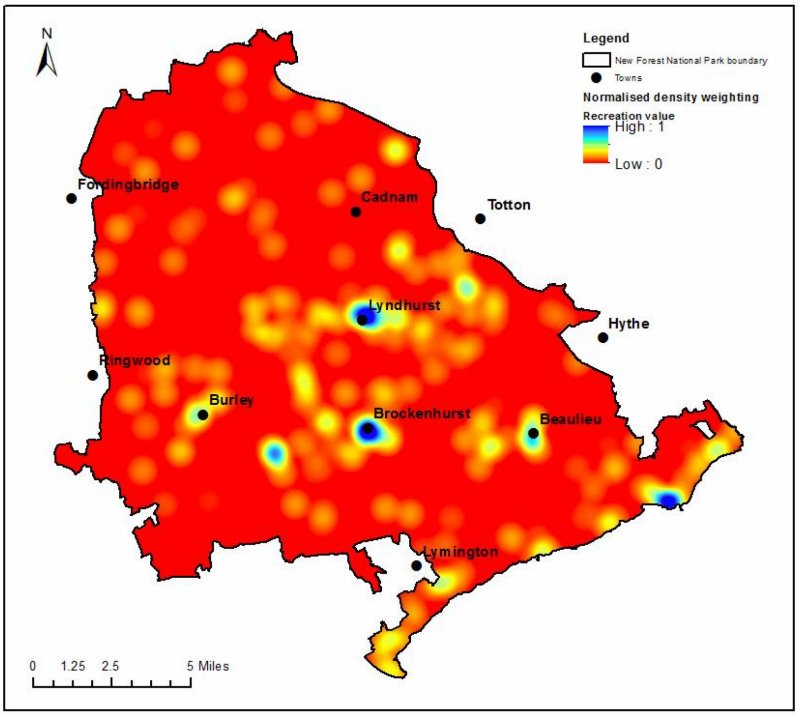

Figure 2.4.14: Kernel density map of normalised (0 - 1) values for recreation value showing areas with low to high value in the New Forest National Park based on a 25m grid cell size and 1000m search radius.

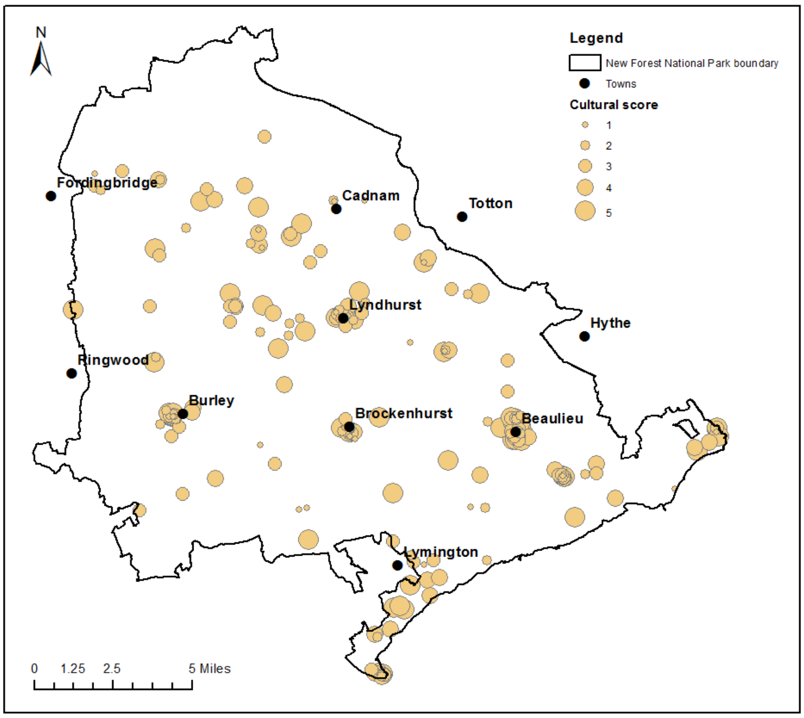

Figure 2.4.15: Map showing the cultural value pins placed by participants across the New Forest National Park during the participatory GIS mapping exercise (n=233, with a total summed point score of 775) represented with graduated coloured circles.

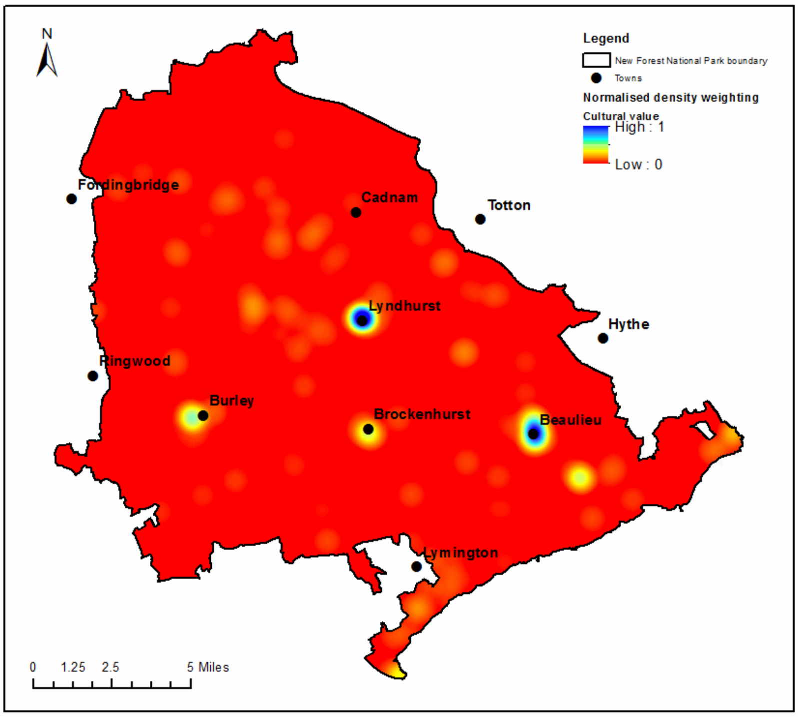

Figure 2.4.16: Kernel density map of normalised (0 - 1) values for cultural value showing areas with low to high value in the New Forest National Park based on a 25m grid cell size and 1000m search radius.

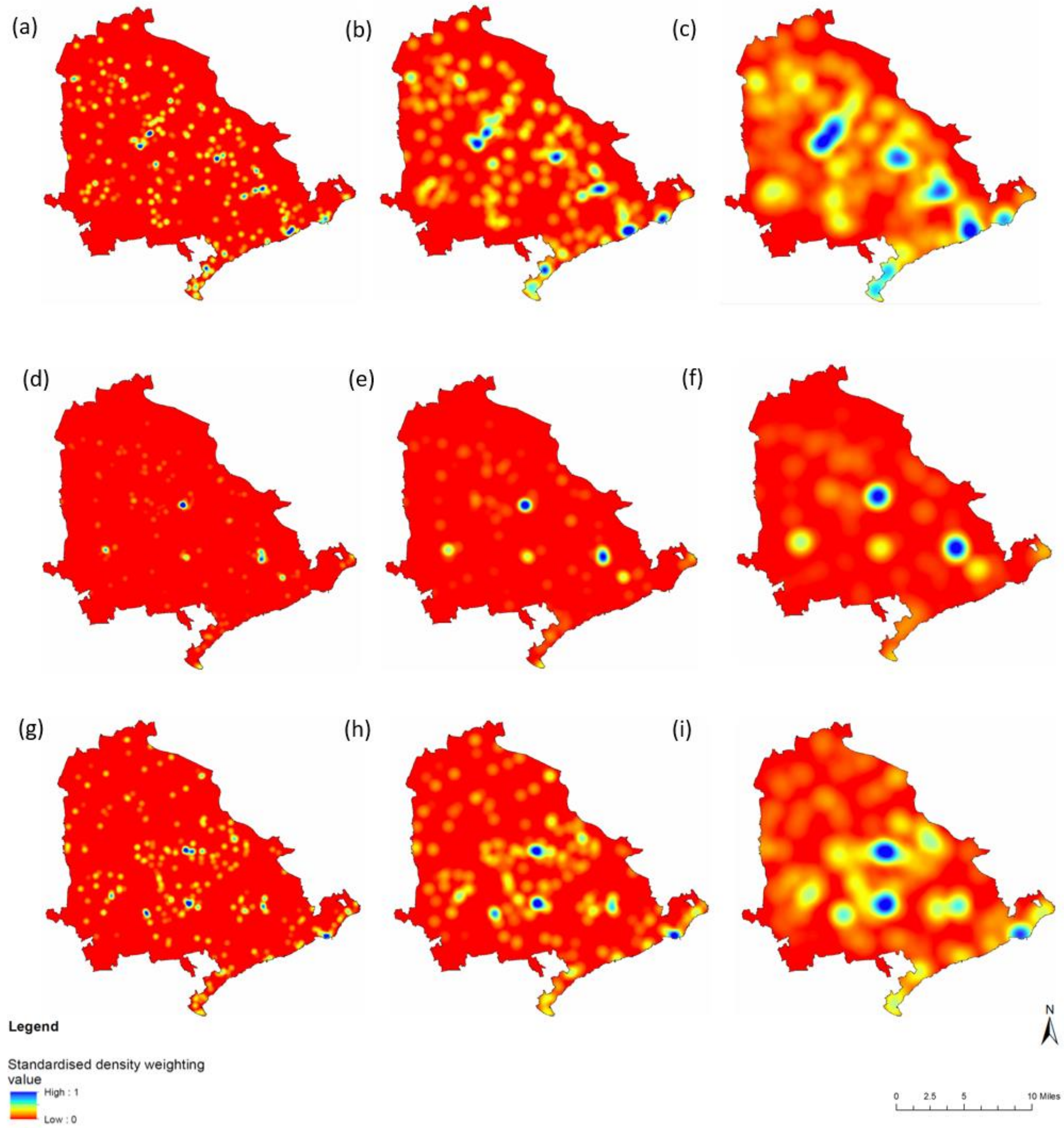

Figure 2.4.17: The effect of radius size on kernel density analysis on ecosystem services in the New Forest. The three columns illustrate the three radius sizes: (a), (d), (g), 500m, (b), (e), (h), 1000m; and (c), (f), (i), 2000m. The three rows illustrate the three ecosystem services: (a), (b), (c), aesthetic, (d), (e), (f), cultural; and (g), (h), (i), recreation. Original boundary data derived from Ordnance Survey (© Crown Copyright and Database right 2015).

Table 2.4.3: Extracted mean values from participatory GIS standardised kernel density maps. [1] This value is one-quarter of the original extracted value, as the search area was four times as large compared to recreation and cultural heritage value (see methodology).

Broad habitat type	Aesthetic[1]	Recreation	Cultural heritage
Acid grassland	0.034	0.043	0.009
Arable and horticulture	0.034	0.044	0.019
Boundary and linear features	0.029	0.067	0.033
Bracken	0.044	0.063	0.006
Broadleaved, mixed and yew woodland	0.038	0.038	0.010
Built-up areas and gardens	0.030	0.083	0.047
Calcareous grassland	0.080	0.239	0.248
Coniferous woodland	0.039	0.041	0.005
Dwarf shrub heath	0.035	0.036	0.006
Fen, marsh and swamp	0.038	0.042	0.011
Improved grassland	0.025	0.037	0.014
Inland rock	0.012	0.012	0.001
Inshore sublittoral sediment	0.136	0.147	0.048
Littoral Rock	0.108	0.114	0.071
Littoral Sediment	0.085	0.111	0.042
Neutral grassland	0.069	0.086	0.017
Rivers and streams	0.118	0.085	0.062
Standing open water and canals	0.087	0.102	0.062
Supralittoral Rock	0.034	0.052	0.052
Supralittoral Sediment	0.091	0.149	0.082
Unidentified habitat	0.032	0.057	0.020
Unidentified water	0.027	0.046	0.009

Kolmogorov-Smirnov testing for normality found that Aesthetic and cultural values deviated from normality, with recreation showing a normal distribution (Appendix I.5). A Friedman test was conducted across all habitat types for aesthetic, recreational and cultural value and tested significant for all post-hoc tests($x^2(2) = 27.909$, $p<0.001$). Post hoc analysis with Wilcoxon signed-rank revealed that all pairwise differences were significant (cultural heritage and aesthetic, ($Z= 0.727$, $P=0.048$; cultural heritage and recreation $Z=1.591$, $P<0.001$; aesthetic and recreation $Z=-0.864$, $p=0.013$). The median value for Recreation was the highest at 0.060, aesthetic was next at 0.038 and the lowest value was for cultural heritage services at 0.020.

Participants scored calcareous grassland and supralittoral rock for the highest cultural value with 0.248 and 0.082. Cultural services had the overall lowest scores across all three ecosystem services, with the lowest for inland rock, coniferous woodland and dwarf shrub heath at 0.001, 0.005 and 0.006 respectively.

The highest overall values can be seen for recreation with calcareous grassland, supralittoral rock and inshore sublittoral sediment at 0.239, 0.149 and 0.147 respectively. Inland rock (0.012), improved grassland (0.025) and unidentified water (0.027) scored lowest.

For aesthetic values inshore sublittoral sediment, rivers and streams and littoral rocks scored highest with 0.136, 0.118 and 0.108 respectively. Improved grassland (0.025), unidentified water (0.027), boundary and linear features (0.029), and built-up areas and gardens (0.030) scored the lowest for mean value.

It must be noted that calcareous grassland, supralittoral rock and littoral rock (with 2, 9 and 42 cells respectively) may disproportionally have high values for certain ecosystem services due to falling within a search radius, and not necessarily due to participants scoring them highly.

Comparing aesthetic and recreation value by fixed-sum and pGIS methods

Statistical difference in value of these selected habitat types between fixed-sum scoring and participatory GIS were not found using Wilcoxon signed-rank test for either aesthetic value (Z = 1.362, p = 0.173) or recreation value (Z = 1.007, p = 0.314.).

Table 2.4.4: Normalised values from fixed-sum scoring survey responses and values extracted from participatory GIS (pGIS) mapping task. [1]pGIS values are standardised within this sub-group of habitat type to allow comparison against fixed-sum values, [2]The fixed-sum categories of arable cereals and arable horticulture and [3]Suburban/ rural developed & Urban were combined to allow comparison against the pGIS categories.

	Aesthetic value		Recreation value	
Habitat	Fixed-sum	pGIS[1]	Fixed-sum	pGIS[1]
Acid grassland	0.334	0.489	0.319	0.501
Arable and horticulture ([2])	0.163	0.492	0.071	0.509
Broadleaved/mixed woodland	1.000	0.552	1.000	0.436
Built-up areas and gardens ([3])	0.110	0.440	0.214	0.965
Coniferous woodland	0.521	0.563	0.663	0.470
Dwarf shrub heath	0.618	0.507	0.561	0.414
Fen/marsh/swamp	0.485	0.557	0.279	0.481
Improved grassland	0.210	0.359	0.431	0.426
Neutral grassland	0.353	1.000	0.420	1.000

Strength of association between fixed-sum scoring pGIS values was conducted using Spearman's rank order correlation coefficient, with a close to significant and strong positive relationship aesthetic ($n = 9$, $rho = 0.617$, $p = 0.077$) (Figure 2.4.18). Broadleaved woodland is scored much higher as part of the fixed-sum scoring method at a normalised score of 1 compared to 0.552 score from the pGIS method. Natural grassland scores much lower with fixed-sum methods at 0.353 compared to a normalised pGIS score of 1.

A significant strong negative relationship between recreation methods ($n = 9$, $rho = -0.667$, $p = 0.0499$) (Figure 2.4.19). Broadleaved and mixed woodland is once again scored higher in the fixed-sum method at a normalised value of 1, compared to 0.436. Build-up areas and neutral grassland score much lower in fixed sum at 0.214 and 0.420 compared to pGIS normalised values of 0.965 and 1.

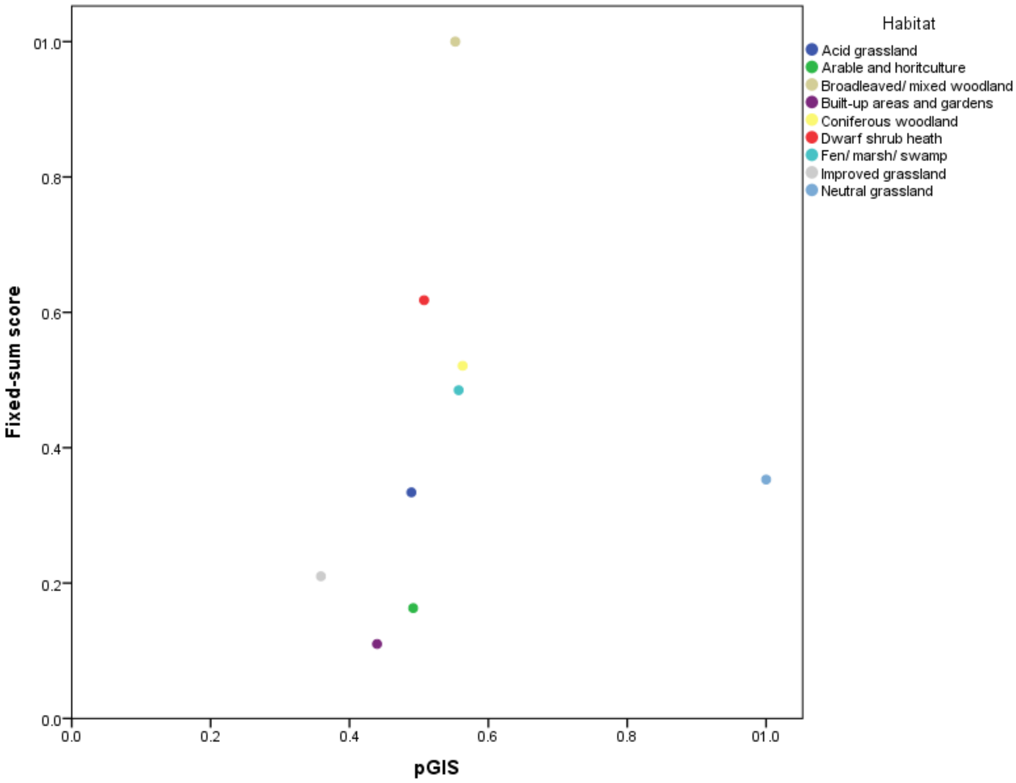

Figure 2.4.18: Scatterplot illustrating the correlation between normalised aesthetic fixed-sum and pGIS scores given by participants for each habitat type.

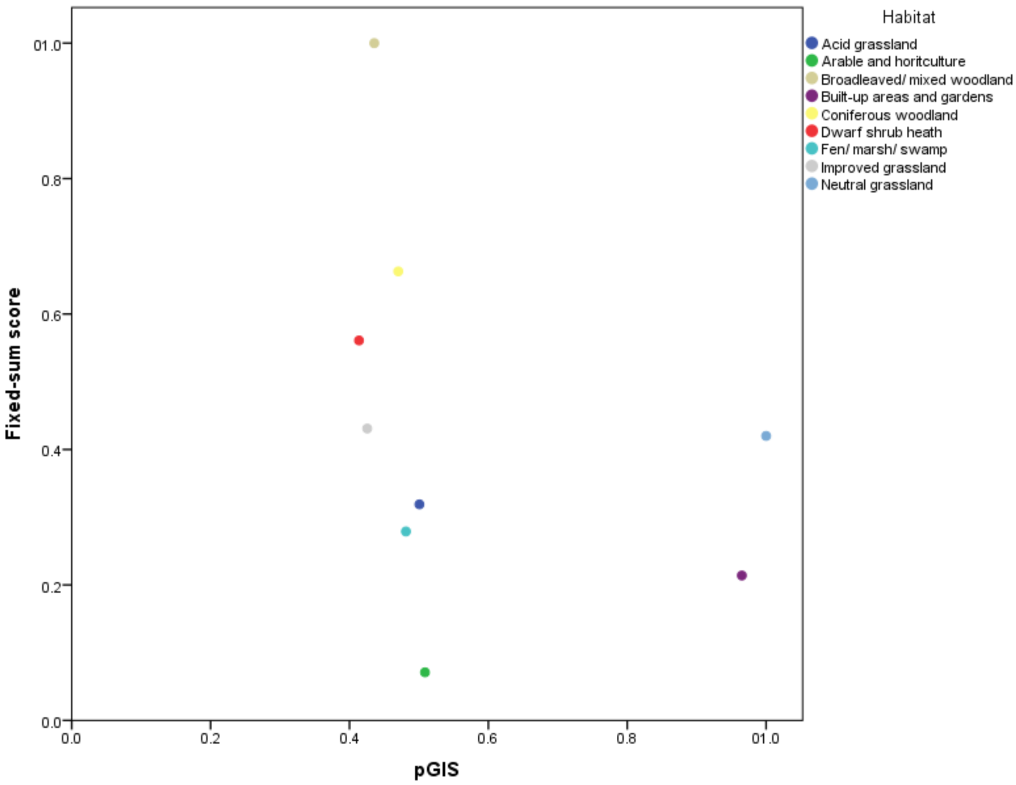

Figure 2.4.19: Scatterplot illustrating the correlation between normalised recreation fixed-sum and pGIS scores given by participants for each habitat type.

2.5. Discussion

The participants surveyed here ranked cultural ecosystem values (aesthetic, heritage/ cultural heritage and recreation), the very values often ignored, as more important than the other groups of ecosystem services, confirming the importance of this group of services. This mirrors research by Quétier et al. (2010) whom found that in mountain grasslands in the French Alps people talked in terms of cultural ecosystem services when interviewed about the various benefits that the grasslands provided. This was explained as industrialised societies material and access needs are secured through 'trade and the welfare state', and people not needing to rely on ecosystems for provisioning ecosystem services. Palomo et al. (2011) reports that 97% of users of Doñana social-ecological system in Spain used cultural ecosystem services the most. With cultural ecosystem services being so important to the public, they should be included in the planning process, e.g. Tielbörger et al. (2010) have suggested using cultural ecosystem services to help guide optimal land allocation schemes in Israel.

Of the habitats present in the study site, broadleaved and mixed woodland, followed by dwarf shrub heath, were valued highest for aesthetic importance using the fixed-sum questions, with suburban and urban scoring lowest. This was similar for recreational value, with broadleaved and mixed woodland and coniferous woodland valued the highest, with urban and arable cereals and horticulture scoring the lowest. A coincidence of services can therefore be seen, often referred to as ecosystem service (ES) bundles (Raudsepp-Hearne et al., 2010). Raudsepp-Hearne et al. (2010) has found that areas with lots of different ecosystem services (high ecosystem service diversity) often had high cultural ecosystem service values. They also identified several ES bundles; the most appropriate for this research being that of 'Destination Tourism' with high levels of tourism linking with moderate levels of natural appreciation and forest recreation. Interactions between ecosystems services can occur when the provision of one service affects the other, or when they both respond to the same driver of change (Bennett et al., 2009). ES bundles and associated synergies and trade-offs in the New Forest will be explored further in Chapter 4. Participants consistently placed higher values on broadleaved and mixed woodland and low values for urban areas across aesthetic, recreation and conservation fixed-sum scoring questions. This highlights the need to keep woodlands, to maintain the high level of cultural ecosystem services they provide. The preference for woodland has been demonstrated by Dandy and Van Der Wal (2011), who demonstrated a shared appreciation of woodland landscapes in three oak woodlands in the UK, across both lay and professional people. Further reasons for the high values of aesthetics and recreation in woodland landscapes are explored in Chapter 3.

The lack of recreational and aesthetic preference for urban habitats has also been shown in other studies. American and European adults have been found to prefer nature views compared to urban views (Ulrich, 1986, Kaplan et al., 1972). Even within an urban environment, 'naturalistic' landscapes are preferred by the public (Özgüner and Kendle, 2006), with time spent in urban parks and woodlands having a positive influence on stress (Tyrväinen et al., 2014). Ulrich (1981) found that nature scenes have a positive influence on psychophysiological state (heart rate, alpha amplitude ['relaxed' brain waves] and emotional stress). People's desire for contact with nature has been seen to serve as an adaptive function – to help psychologically restore people (Van den Berg et al., 2007). Nature has been seen as an important foundation for culture and folklore for humans, and though nature is important for inspiration for many things, including literature, art and national symbols, humanity does not seem overly conscious of this fact (De Groot et al., 2002). Nature scenes are important to people, as demonstrated by, for example, houses near national parks or with ocean views being more expensive than similar houses in other locations (Costanza et al., 1997).

Contrary to this, participatory GIS results indicated that heritage points clustered strongly around towns, and along the coast. It is worth noting that heritage values were called cultural values in this study, owing to the definition provided to participants in the study for ease of understanding. Heritage values were not included in the fixed-sum questions; if this research was replicated, they would be included. Recreational values were less clustered, though still had clustering around the major towns, with aesthetic points showing the least clustering around towns. Interestingly, distinct areas of high value can be seen along the coast for recreation and aesthetic value. All three ecosystem services are heterogeneous across the New Forest landscape, though overlapping in areas, they do not have the exact same distribution across the landscape.

Using participatory GIS methods, recreation was valued more highly than aesthetics, followed by heritage value, with no other services being mapped. More people chose to place pins for recreation and aesthetic value over heritage. Heritage services had the overall lowest scores across all three ecosystem services, with the lowest for inland rock, coniferous woodland and dwarf shrub heath. This could potentially infer that recreational services are more important, or that heritage values were the most difficult for participants to understand due to their transient nature. This mirrors Daniel et al. (2012) whom states that cultural services are often thought of as intangible and subjective. Whereas many examples of studies looking at ecosystem-based recreation or scenic beauty can be found; other cultural services such as those associated with heritage value are less well characterised (Chan et al., 2012). It may be this unfamiliarity with creating associations with explicitly valuing cultural heritage that resulted in participants placing fewer pins for heritage value. This highlights a further question for research: as participants rated heritage as an important ecosystem service, though placed the least pGIS pins for this ecosystem service, do people understand where areas of cultural heritage are in the New Forest National Park? Does management need to promote places of cultural interest, as well as those for recreational activity?

Using pGIS, calcareous grassland, supralittoral rock and inshore sublittoral sediment (coastal land types) have the highest mean score for recreation, with improved grassland having the lowest (apart from unidentified water). This contrasts with the results of the fixed-sum questions, which gave the highest values to woodlands. However, these habitat types were not included in fixed sum questionnaire, apart from improved grassland, which does not score lowest for any fixed-sum question for any service. Fixed-sum question habitats were chosen as the being the most dominant terrestrial habitats in the New Forest. This highlights a limitation of using a fixed-sum question that was limited by the number of habitats that could be included (as high numbers of habitats would make the question difficult for the participants to process, hence needed to be kept relatively simple for ease of completion). PGIS methods allow all habitats to be included, as the habitats are not explicitly labelled to the participant. For aesthetic values inshore sublittoral sediment, rivers and streams and littoral rocks scored highest. Apart from unidentified water, improved grassland, boundary and linear features and built-up up areas and gardens scored lowest. This clearly highlights an advantage to this method, as surveys alone would miss the rarer but more important habitats.

Investigating only those habitats that can be compared between the two different methods using normalised scores, aesthetic scores were most dissimilar for broadleaved/mixed woodland and neutral grassland. Using fixed-sum scoring participants scored woodland as the most important at 1.0 but with pGIS scored it at mid-range at 0.552. Surprisingly, whereas neutral grassland had a low score of 0.352, whereas using pGIS it was 1.0. Results for recreation were striking, with an almost reversed preference between the methods, with woodland scoring 1.0 using fixed-sum questions, though only 0.436 using pGIS, with neutral grassland scoring 0.420 with fixed-sum and 1.0 using pGIS.

People showed a preference for the coastal land types and water in the pGIS task. A preference for water has been well documented, also called ‘Hydrophilia’ (Herzog, 1985), from wildland scenes containing water in Røros, southern Norway (Kaltenborn and Bjerke, 2002) to water acting as a preference predictor (with the presence of water being more important than the size of the water body) in natural landscapes in Scotland (Wherrett, 2000). People did not value woodland as highly in the pGIS task, as in the fixed sum task. This may have been due to people not understanding where the tree stands were on the map.

Most people thought that they understood the conservation and environmental issues in the New Forest. However, the possession of environmental knowledge does not necessitate pro-environmental behaviour, and indeed is a well-known phenomenon, lacking an English name but known as *Umweltpsychologie* in German (Kollmuss and Agyeman, 2002). Schultz (2000) has argued that concern for environmental issues is an expression of how much people feel themselves to be part of the natural environment. As all surveys were of people whom had at least visited the New Forest, it could be inferred that an enjoyment of being in nature, or the New Forest predicated a sense of being part of nature, and hence a concern for nature, or the New Forest.

There were several limitations to this research, including bias. The participants were self-selecting and needed access to appropriate online hardware and software due to the nature on the online format of the questionnaire and pGIS task. There was a high level of participation from postgraduate students, that may have created bias. It has been found that knowledge of environmental issues increases with increased education (Kollmuss and Agyeman, 2002). This is further illustrated through the expression of most participants agreeing to the environmental statement. This may have resulted in those people having an understanding and interest in outdoor activities being overly represented. As these are the individuals that would be most likely to actively use the New Forest National Park, it can be assumed that users whom use the Park were not under-represented. A further consideration is the dis-connect between the ‘fixed sum’ and ‘pGIS’ methods, whereby participants may have ranked rare habitats highly, though might not have been able to access them. Accessibility of habitats in further explored in Chapter 3. In line with current research, the limitation of not having considered both supply and demand mapping could be investigated in future research in the New Forest National Park.

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Chapter 3

Do as I say, not as I do: recreational preferences differ from measured behaviour in an English National Park

3.1. Abstract

The assessment of cultural ecosystem services (CES) is limited in comparison to other ecosystem services, despite regularly being highlighted as needing further clarification and research. CES, seen as being intangible and difficult to measure, have no consensus on assessment methodologies in literature. This study investigated using stated-preference surveys and tracked visitor behaviour ascertain any differences in values for aesthetic appeal and recreational usage between visitors and between methods. Visitors to the New Forest National Park, in the south of England, were surveyed to elicit values for aesthetic appeal and recreation usage in relation to woodland characteristics, using a multi-faceted approach. Face-to-face surveys, ranking of photorealistic images of landscape structure and GPS tracking technology were used to explore the value people placed in these landscapes and ascertain whether interview methods elicited a different response to the tracking method.

Scoring of images showed that visitors showed a greater preference for more wooded areas, with increasing scores for woodland cover from 0% to 100%. As aesthetic score increased, there was a positive relationship with recreation score. Interestingly, 100% woodland was scored higher with a median of 4, compared to males scoring 3.5. Survey results indicate that despite visitors aesthetic preference for a 100% wooded landscape; they still understood the high aesthetic, recreation and conservation value of these landscapes. GPS results showed compared to availability of habitat versus the time spent (within a 3000 m radius buffer), 429% more time was spent in boundary and linear features followed by acid grassland and built-up areas and gardens, with 133% and 119% more time respectively. A deduction over expected time can be seen for improved grassland and bracken, 91.5% and 79% respectively. Surprising, if the whole of the New Forest National Park is treated as equally accessible, 3.5% time over expected is spent in coniferous woodland and 22% less than expected for broadleaved, mixed and yew. Although people

preferred woodland in the surveys, and images of woodland with a higher canopy density, the trackers revealed that they spent less time than would be expected in this habitat.

This study highlights the difference between stated and measured behavioural response in visitors. The obvious implication for future research in this area is the awareness that stated responses methodologies are best utilised for aesthetic assessment and behavioural monitoring methodologies for recreational assessment of these cultural services. The implications of these results for management of National Parks such as the New Forest are explored.

3.2. Introduction

In recent years, the concept of ecosystem services has become an important element of environmental policy, as illustrated by the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES), the EU Biodiversity Strategy to 2020, and national policies such as the UK's Biodiversity Strategy (European Union, 2011). These policy initiatives have been supported by major assessments of ecosystem services, such as the Millennium Ecosystem Assessment (2005) undertaken at global and national-scale assessments such as that undertaken in the UK (UK National Ecosystem Assessment (UK NEA), 2011).

The Millennium Ecosystem Assessment (2005) defined ecosystem services as the benefits that humanity gains from ecosystems. This was based on earlier definitions, such as that provided by Daily (1997) who defined ecosystem services as 'the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life', whereas Costanza et al. (1997) indicated that 'ecosystem goods (such as food) and services (such as waste assimilation) represent the benefits human populations derive, directly or indirectly, from ecosystem functions'. Similarly, Fisher et al. (2008) defined ecosystem services as 'the aspects of ecosystems utilized (actively or passively) to produce human well-being'.

Growth in adoption of ecosystem services as a policy concept has been accompanied by a rapid increase in research effort (Carpenter et al., 2009, Nicholson et al., 2009, Seppelt et al., 2011). This research has documented the factors influencing the provision of ecosystem services to people, and the different values of these services, in a wide range of ecosystems and socio-economic contexts. Progress has been made on a variety of different issues including land use dynamics, earth system modelling, human and earth system history and governance of common-property systems (Carpenter et al., 2009). However, relatively little progress has been made in assessing cultural ecosystem services (CES), which are often neglected owing to methodological challenges (Plieninger et al., 2013), (Daniel et al., 2012, Seppelt et al., 2011, Feld et al., 2009). This has led to a representational bias in ecosystem assessments and landscape planning decisions, in which CES are often poorly represented (Hernández-Morcillo et al., 2013). CES are often referred to as 'intangible' and 'subjective' and pose difficulties for economic or biophysical assessment (Daniel et al., 2012, Millennium Ecosystem Assessment, 2005).

CES were defined by Costanza et al. (1997) as being “aesthetic, artistic, educational, spiritual and/or scientific values of ecosystems”, whereas the Millennium Ecosystem Assessment (2005) used the term ‘non-material benefits’ instead of the term ‘values’, which were taken to include ‘spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience’. Both recreation and aesthetic value are consistently considered as CES by a wide range of authors (de Groot et al., 2010, Costanza et al., 1997). de Groot et al. (2010) describe aesthetic value as being the appreciation of natural scenery, with recreational value being the opportunities for tourism and recreational activities. Whereas ‘nature-based tourism’ encompasses those visits which include an overnight stay, ‘outdoor recreation’ refers to day activities that make up part of a visitors daily routine in nature or green areas (Bell et al., 2007). Some of these benefits can be tangible, for example, by measuring the time spent in an ecosystem by visitors.

Despite the intangible nature of CES, their importance has widely been acknowledged in previous research. For example, Guo et al. (2010) found that the dependence of people on cultural services has increased as economic development has taken place, and often exceeds that of regulating services (which include climate regulation and flood regulation). They have also been found to be important motivators for land management and ownership (Plieninger et al., 2012). de Groot et al. (2010) argue that it is important to develop socio-cultural reporting and measure practices that are robust to ensure comparability. The importance of CES can be seen through the discourse in various academic circles, from critical theorists’ concerns with ‘commodification of nature’ to ecologists thoughts on intrinsic values of nature (Chan et al., 2012).

In order for the factors influencing provision of CES to be understood, their spatial and temporal dynamics need to be examined at the landscape scale. A shift from single site to landscape scale management in conservation has occurred in recent years (Hodder et al., 2014). This has been supported by landscape ecology developing as a scientific discipline with its focus on the influence of land-use history and natural disturbances, dynamics of metapopulations and metacommunities (Turner, 2005, Leibold et al., 2004, Hanski, 2001). The importance of the ‘landscape’ approach can be seen through the adoption of the ‘Ten Principles of Landscape Approach’, a document summarising best practice implementation developed by the Convention on Biology Diversity (CBD) (Sayer et al., 2013). A limited body of research has examined CES at the landscape scale. In a recent review, only 23% of CES studies were explicitly represented spatially, those that did primarily focused on the mapping of biophysical variables overlaid with stakeholder’s cultural ecosystem service values (Guo et al., 2010). The traditional approach of focussing on intrinsic values has been disputed by Chan et al. (2016), whom infers that values that are derivative of relationships such as virtue and justice also need to be considered.

A variety of approaches have been used to assess CES, using both quantitative and qualitative measures, with 38% of studies utilising combined data (Hernández-Morcillo et al., 2013). Research has taken place on both finer, 'near-view' scales, and broader 'landscape' scale (Daniel et al., 2012). Yet no consistent methodological approach has been developed for assessing CES at the landscape scale (Hernández-Morcillo et al., 2013, Seppelt et al., 2011). Methods have included using monetary evaluation for scenic beauty; for example in ponderosa pine woodland (Brown, 1987) and managed forests in New Zealand (Thorn et al., 1997). Empirical model studies have also been supported by perception based survey methods (Buhyoff et al., 1982, Silvennoinen et al., 2001), often using computer visualisation of changes in landscape structure or pattern (Daniel et al., 2012, Meitner et al., 2005). Two approaches have been used for visual landscape quality assessment. The first utilises experts and is often the approach adopted by environmental managers. The second is a perception-based approach that is often used by researchers. In both approaches landscape quality is seen to derive from an interaction between human perception and the biophysical features present in the landscape (Daniel, 2001). The technique of assessing visual quality of a landscape through the use of public surveys to score photographs is often used for landscape valuation (Arriaza et al., 2004). A comparative study on the value of landscapes found that the two approaches provided different results. Inhabitants based value on a set of criteria including emotion, intimate local knowledge and everyday experience. The formal visual criteria experts used for assessment was of less importance to the local inhabitants (Vouligny et al., 2009). Structured interviews were used by Plieninger et al. (2013) to elicit aesthetic value, social relations and educational values in Eastern Germany, using GIS to create hotspots and cold spots of ecosystem service provision. Other research has included the assessment of agricultural heritage values using stated preference surveys (using contingent valuation) in Chiloé Island, southern Chile, demonstrating the importance of agriculture heritage as being highly valuable (Barrena et al., 2014).

Many previous studies of CES have used stated-preference methods. There are two groups of stated preference methods; contingent valuation and choice modelling techniques. Contingent valuation surveys asks participants their 'willingness to act' or 'willingness to pay' for changes, choice modelling surveys present participants with several alternative and elicit the most preferred from them (Riera et al., 2012). Surveys are designed so that the answers to the questions can be designed as a choice experiment, contingent ranking, rating or grouping or the best-worst approach that asks the participant to select the best and worst from a choice set – ultimately using an appropriate model of preference (Riera et al., 2012, Brown, 2003). Various studies have used direct, or stated preference methods, for example Newton et al. (2012) used an online survey tool in isolation to elicit cultural [heritage], aesthetic and recreation value in the river catchment of the River Frome, Dorset, Southern England.

For analysis of spatial patterns in CES, the use of photo-manipulation software to alter photos and remove non-target landscapes features can be seen in a number of recent studies. For example, Holgén et al. (2000) estimated economic value of recreation in various stand types within boreal forest using a questionnaire survey. Preference values regarding different forest attributes were collected by showing photographic series with a forest stand at a specific phase of rotation of the forest to participants (Holgén et al., 2000). Lindemann-Matthies et al. (2010) manipulated images to remove elements (such as power lines) that caused negative influence on scenic beauty ratings. This allowed specific components of the landscape to be examined (Lindemann-Matthies et al., 2010). Photo-realistic visualisations of future landscape developments have also been used successfully, for example by Soliva and Hunziker (2009), who used Adobe Photoshop to create four future visualisations of the Surses valley in the Swiss Alps in questionnaires sent to inhabitants of the area. A similar approach was used by Cordingley et al. (2015) whereby photo-realistic images of different heathland vegetation cover and gradients of dwarf shrub heath, to scrub and woodland were presented to heathland visitors. The utilisation of photo-realistic images have also been found to be valid representations of real landscapes (Lange, 2001).

Despite this assemblage of research that focuses on stated-preferences, these methods have rarely been compared with actual measurements of human behaviour to see whether they correspond. There are various methods that can be used to directly survey movement patterns of people in an area by visitors. These include use of field observations, self-registration approaches, video cameras, GPS trackers, smartphones, social media derived information (e.g. twitter feeds), thermal cameras and GIS approaches (del Rosario et al., 2015, Deadman and Gimblett, 1994, O'Connor et al., 2005, Lau and McKercher, 2006, Shoval and Isaacson, 2007, Garthe, 2010, Pettersson and Zillinger, 2011, Birkin and Malleson, 2012, Orellana et al., 2012). Examples of the application of such methods include

O'Connor et al. (2005), who used geo-spatial ankle transmitters on 900 individuals in Twelve Apostles National Park in Victoria, Australia to determine tourist behaviours. Pettersson and Zillinger (2011) used a combined method of GPS tracking and surveys on background and experiences for visitors to the Biathlon World Championships 2008 in Östersund, Sweden, finding that the combined approached allowed both collection of information on experience and mobility for event management. Other examples include analysis of human movement in space-time in the Old City of Akko (Israel) to identify time geographies of visitors (Shoval and Isaacson, 2007), factors affecting tourist movement visiting Hong Kong using a GIS approach (Lau and McKercher, 2006) and use of large GPS datasets to discover commonalties of visitors preferences in natural recreational areas (Orellana et al., 2012). However, as far as I am aware, no CES based studies have used such approaches for assessing human movement patterns in rural landscapes.

Although stated preference methods have been widely used to assess CES, there is a risk that stated preferences do not necessarily correspond with human behaviour, with a common criticism that the choice sets are hypothetical so the participant only states they would behave in a certain fashion, rather than actually making any behavioural changes (Adamowicz et al., 1994, Cummings et al., 1986). Furthermore Jacquemet et al. (2013) suggests that sincere responses are often difficult to elicit for non-market goods. Stated preference methods are a type of preference satisfaction approach, inferring utility from choices due to 'what is best for someone is what would best fulfil all of his desires' (Parfit, 1984). The very process of decision making is very much context-dependent, with preference being constructed at the time the person is asked (Fujiwara and Campbell, 2011). Stated preference methods can lead individuals to overstate their valuation for a good by two or three factors, and has been attributed to non-commitment bias (Fujiwara and Campbell, 2011) and even strategic bias, whereby individuals overstate their valuation of a good as they believe it may influence its provision (Fujiwara and Campbell, 2011).

Stated preference techniques have rarely been tested with other techniques, in the context of assessing CES at the landscape scale. Despite the highlighted importance of CES, and the limited use of stated preference or behavioural research within CES, these studies rarely incorporate both methods. Pettersson and Zillinger (2011) suggest that GPS monitoring techniques are suitable for understanding mobility and personal experience, with surveys allowing collection of background information.

This chapter investigated recreational and aesthetic appeal of woodland areas in the New Forest National Park. This area is a major resource for recreation, attracting over 13 million day visits per year. Previous studies examining recreational visits to the New Forest (Sharp et al., 2008) found that 40% of visitors were staying tourists, 25% were day visitors coming from over 5 miles away and 35% were local visitors living with 5 miles of the park. Visitation rates were higher in the summer, with non-local visitors tending to visit honey-pot sites and locals avoiding these sites (Sharp et al., 2008). However, little information is available on spatial patterns of visitor movement within this National Park, and how this relates to landscape characteristics. The objectives of this research were to investigate whether visitor recreation and aesthetic values were related to the extent of woodland cover, and whether visitors showed a preference for recreation in certain habitats. Stated-responses for aesthetic and recreation values were measured and behaviour tracked to understand if there was a difference between the two types of measurement on the same group of visitors. This was achieved by (i) using on-site questionnaire surveys to gather information on visitor values on aesthetic and recreational qualities of woodland in varying stages of decline and (ii) using GPS tracking technology to see where visitors went and how this related to available habitat.

3.3. Materials and methods

Ethical statement

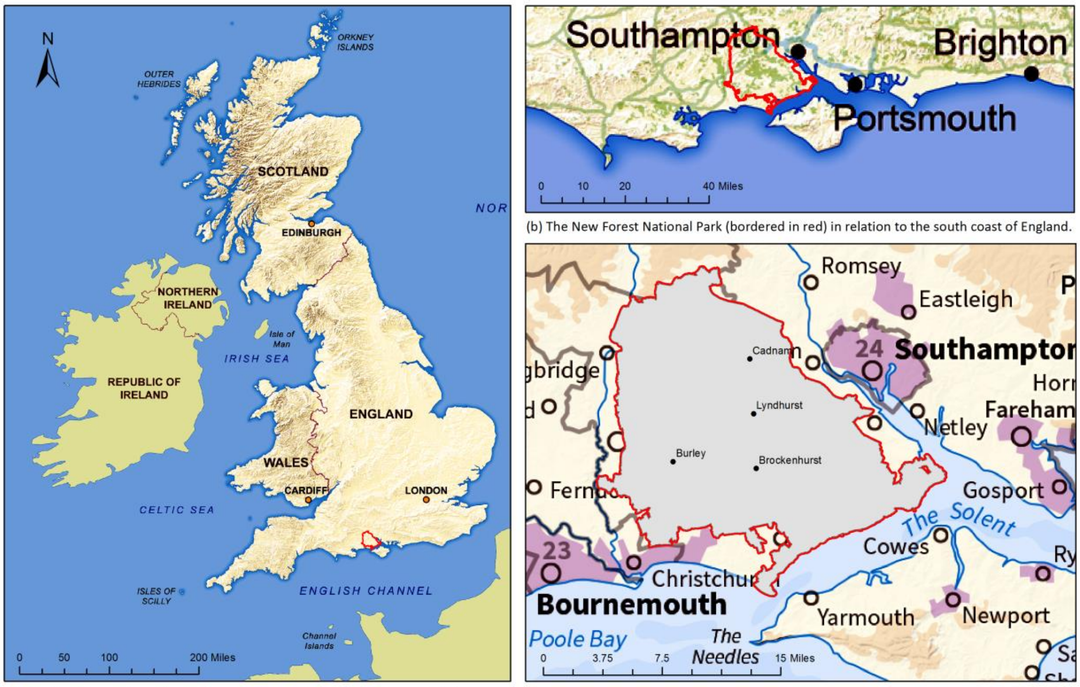
Permission to survey at each site was given by the Forestry Commission – South England Forest District (Permit number 017485/2015). Individuals interviewed were provided with a Participation Information sheet (Appendix II.1) and signed a physical consent form (Appendix II.2) before the survey was conducted. Data were anonymised, and all participants were made aware of how the data would be used, prior to requesting their voluntary participation. This study was approved by the Bournemouth University Research Ethics Committee.

Study area

The New Forest covers 56,658 ha on the south coast of England, latitude 50°51'59" N, longitude 01°40'50" W (see Figure 3.3.1) (New Forest Park Authority (NFPA), 2010). The 'perambulation', which has byelaws allowing livestock grazing (Newton, 2010), covers 37,907 ha spanning the counties of Hampshire and Wiltshire. The New Forest Special Area of Conservation comprises a large number of habitats including broad-leaved deciduous woodland (29%), coniferous woodland (17%) and heath (34%) (JNCC, 2011). This complex mosaic of habitats is home to a wealth of biodiversity, with high species richness in groups such as insects, birds, mammals, reptiles, fungi and lichens amongst others (Newton, 2010). As with any landscape, its basic structure is a result of its underlying relief and geology. In the New Forest, the geology is comprised of gravel terraces, divided by valleys that eroded as a result of falling sea levels in the Pleistocene and are filled with clays and sand of Tertiary deposits (New Forest Park Authority (NFPA), 2008).

Large populations of free-ranging large herbivores, including livestock and deer, have made a significant contribution to the functioning of the New Forest as an ecological system. The Forest originated as a Royal hunting reserve in 1079, and a traditional commoning system survives to the present day. Currently, nearly a quarter of the land is used for farmland and settlements. The remaining three-quarters are 'Crown lands', being of Royal Forest status (Newton, 2010). Designated in 2005, the New Forest National Park includes 20 SSSIs, six Natura 2000 sites, and two Ramsar Convention sites at least partly within its boundaries. Approximately 50% of the Park's land is covered by unenclosed vegetation, often referred to as the 'Open Forest', the structure and composition of which is strongly influenced by the action of large herbivores (Newton, 2010, New Forest Park Authority (NFPA), 2008).

The New Forest has also been a major recreational resource since the mid-19th century, and today receives around 13 million day of visits per year (New Forest Park Authority (NFPA), 2008).



(a) The New Forest National Park (bordered in red) in relation to Great Britain.

(b) The New Forest National Park (bordered in red) in relation to the south coast of England.

(c) The New Forest National Park (bordered in red) in relation to the local area, with larger towns highlighted.

Figure 3.3.1: Maps of the New Forest National Park on the south coast of England, with the Park boundary overlaid in red on a series of increasing scale Ordnance Survey maps (© Crown Copyright and Database right 2015): (a) the Park in relation to the United Kingdom, (b) the south coast of England, and (c) in relation to the local area.

Survey locations

Owing to the large area and spatial heterogeneity of the New Forest, random stratified sampling was used. The New Forest SAC is divided into ten New Forest Park Authority's management units (see Figure 3.3.2); each unit was used as an individual stratum. Cochran (1977) states that there are four reasons to adopt random stratified sampling: (1) estimated means may be needed for each stratum, (2) sampling problems may directly relate to the stratum, (3) more precise estimates of population parameters can be obtained, therefore confidence intervals can be narrower and (4) more than one member of field crew can undertake sampling concurrently using stratification (Young and Young, 2013). Stratification is common in wildlife studies, though this is often used to estimate parameters within strata for comparative purposes. The main objective here of using stratification was to minimise variance of summary statistics for all combined strata (Morrison et al., 2008).

From each stratum (management unit), a random car park was chosen from a list of car parks, which were extracted from an Ordnance Survey (OS) map. Car parks were chosen for two reasons; firstly, as this study examined the aesthetic value and recreation usage of the New Forest, the aim was to target the majority of visitors which previous studies (conducted in the Forest National Park) found that 15 million people reside within a 90 minute drive to the New Forest National Park, and the majority of visitors to the New Forest use a car or private vehicle (New Forest Park Authority (NFPA), 2007a). 88% of local residents and 94% of non-local day visitors used cars, compared to 67% of staying visitors (though 96% of these staying visitors used a car to reach their accommodation) (New Forest Park Authority (NFPA), 2007a). Such high numbers of recreational visitors using car parks as access points was a justification for using them as the survey points in this study. The second reason concerned the high value of the GPS data loggers, thus the survey design allowed for the best possible security and retrieval of the units by being in attendance at the location the participants parked their vehicles.

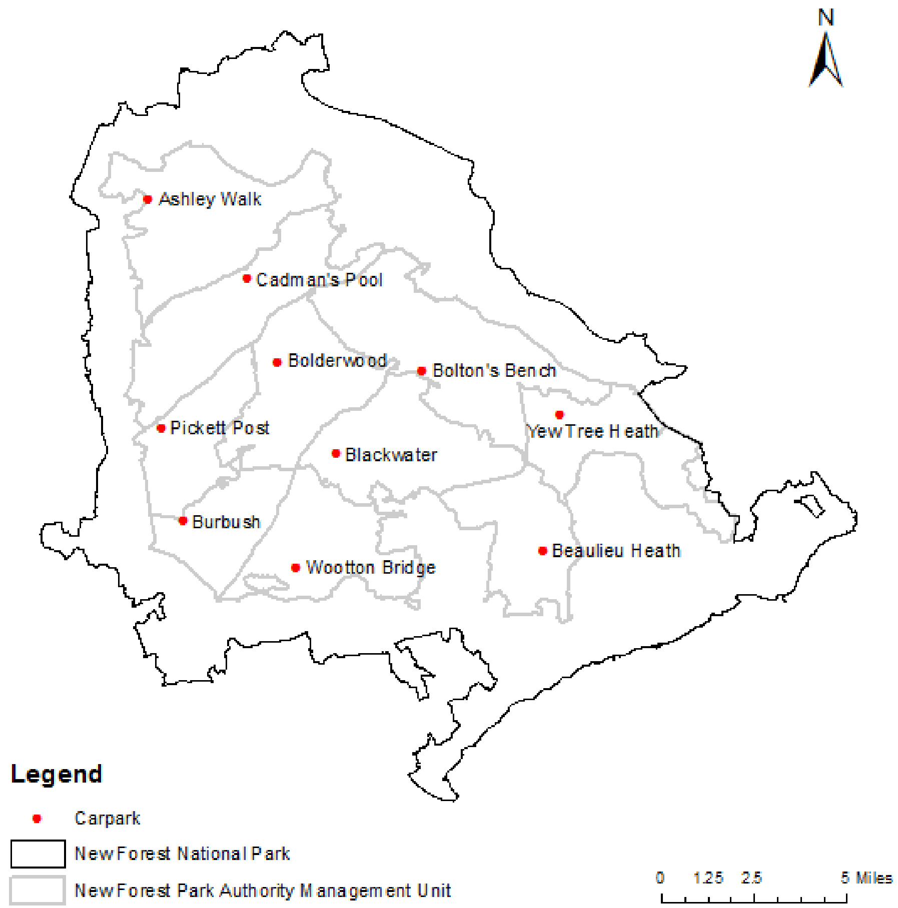


Figure 3.3.2: Map of the New Forest National Park showing surveyed car park locations and management units.

Questionnaire design

Questionnaires were designed to collect information on (i) recreation values for landscape images with different degrees of woodland cover, (ii) aesthetic values for landscape images with different degrees of woodland cover, (iii) demographic and personal information of participants, (iv) the importance of woodland for conservation, aesthetics and recreation and (v) aesthetic preferences of participants regarding tree species and size.

For (i) and (ii), images (12.9 cm x 6.5 cm) were laser printed in full colour as part of the questionnaire survey. Laser copier paper of 80 GSM in white was used to print the surveys in a matt finish. The images were presented in a set order, from the most woodland cover (100%) in steps of 25% to least woodland cover (0%). Participants were given the paper survey to hold on a clipboard. This allowed participants to examine the images and complete the survey at their own pace. The participants were asked to rate the relative value of different degrees of woodland cover, as a visitor might encounter during a walk through the New Forest (see Figure 4) on an adapted Likert scale of 1 to 5 (with 1 being very unappealing to 5 being very appealing; unappealing, neutral and appealing were accorded a score of 2, 3 and 4 respectively.).

Likert scales (Likert, 1932) are a method of capturing judgments from an individual being studied, and are one of the most widely used scales to measure attitudes. Likert scales are also known as summated rating scales; they are used by presenting a statement and asking the respondent whether they strongly agree, agree, are undecided, disagree, or strongly disagree (Ary, 2009). The rest of the survey comprised questions to elucidate information for (iii), (iv) and (v). For (iii), the following demographic and personal data was collected: (1) Age, (2) Gender, (3) Type of area of residence, (4) Proximity to the New Forest National Park, (5) Mode of travel, (6) Activities being undertaken.

For (iv), participants were asked to score the importance of woodlands for providing conservation value, recreation value and aesthetic value, on an adapted Likert scale of 1-5 for each, with 1 being very unimportant to 5 being very important. Definitions of each type of value were given. Conservation value was described as being a measure of how important a habitat is for wildlife, it was explained that it was reflection of how strongly the participant felt the habitat should be protected, or managed, for the purposes of conserving wildlife. Recreation value was described as the importance the participant felt a habitat was for recreation including hiking, dog-walking, horse-riding, cycling and picnicking. Lastly, aesthetic value was described as the value placed on a habitat for its beauty or visual appearance. A further question asked the participant how concerned they were that the mature beech woods of the New Forest were dying because of climate change on an adapted Likert scale of 1-5, with 1 being not at all concerned to 5 being very concerned. The final section (v), focused on the aesthetic value of trees, to collect information on whether the participant preferred broadleaved or conifer trees and smaller or larger (veteran) trees, with an option to declare no preference.

An initial version of the questionnaire was designed and trialled on 20 volunteer students from Bournemouth University. The students were asked to complete the survey and provide feedback on how easy the questionnaire was to comprehend and understand. The students were a mixture of male (8) and female (12), and undergraduate (6) and postgraduate (14) students from mixed disciplines studying at Bournemouth University in April 2015. This feedback was used to redevelop the questionnaire, improve the questionnaire and streamline the whole design. The resulting finalised questionnaire was deployed across the chosen sampling locations across the New Forest National Park. Please see Appendix II.3 for details of the survey.

Woodland images

Photo-realistic images were developed to represent a landscape in the New Forest, with a range of woodland cover (Figure 3.3.3). The images were designed to collect recreational usage and aesthetic appeal scores as well as to see how these values would be affected by declining woodland cover. Photos of beech woodland in Ridley Wood of the New Forest were taken on the 7th June 2015, a clear day, between the hours of 10:00 and 14:00. A Canon EOS 40D single lens reflect camera, with no flash, was used. A Canon EF-S 18-55 mm f/3.5-5.6 IS II lens was used, consistently set to 18 mm, to maintain a wide-angle. It has been found that photos are most realistic when a focal length of less than 50 mm is used (Lindemann-Matthies et al., 2010). A range of photos at varying distances from beech woodland and of different stands were taken. Photos of heathland in the distance were also taken using the same camera settings. Two separate photographs of heathland at a distance and a generic sky were chosen to create the base image for the background. This allowed the skyline to be standardised. The photo-editing software Adobe Photoshop CS 6 (Adobe Systems Europe Ltd, Berkshire, UK) was used to alter this base image to create five separate images for individual woodland cover for 100%, 75%, 50%, 25% and 0%.

It is not often the depiction of an actual site *per se*, though rather the character of the site that the participant will rate (Ode et al., 2008). For this reason Ode et al. (2008) state that photographs can be manipulated to allow selected landscape features to be assessed. Consequently, the images of the varying woodland cover types were created by clipping sections of beech tree stand photos and using a grid to make up the percentage woodland cover. To maintain realism, the edges of the trees were 'feathered' and the gaps between trees filled with a background beech stand image. The outline of the woodland stand in each photo was purposefully manipulated to look uneven and not repetitive, again to maintain realism. The modification of the woodland cover alone, with standardised land and sky ensured that the preference values for the different woodland cover types for both recreation and aesthetic value could be assumed to be based on the varying woodland cover, rather than any other features (Cordingley, 2012).

(a)

(b)

(c)

(d)

(e)

Figure 3.3.3: Illustration of survey images of a sequence of photorealistic images representing different degrees of woodland cover for recreation usage and aesthetic appeal, (a) 100% woodland, (b) 75% woodland, (c) 50% woodland, (d) 25% woodland and (e) 0% woodland to elicit responses from survey participants.

Survey method

The surveys were conducted during July and August 2015 in the New Forest National Park across ten car parks. To obtain a representative range of participants, each car park was visited between 08:00 and 18:00, until 20 respondents had completed the survey at each location. Understanding visitation patterns was not an aim in this study, so set times, days and weather patterns were not taken into account. The same two surveyors were always present during the collection of data. Visitors were approached and invited to participate in the survey, and the study explained to them. If they wished to proceed, a 'Participant information sheet' was given to them, and consent signed. Anonymity to participants was guaranteed to all participants. The following particulars were explained to the participants:

the format of the questionnaire, the amount of questions, how to assess the images for aesthetic and recreational value and the time taken to complete the study would be in the region of 10-15 minutes. Each participant was given the option of taking a GPS data logger with them during their visit, though was not a requisite of taking the survey.

Tracking visitor movement using GPS data loggers

The GPS data loggers used, the Qstarz BT-Q1000XT (Qstarz International Company Limited, Taipei, Taiwan) were small and easily transported; having a belt loop that could be attached to clothing or bags if the participant wanted. This method allows for very accurate geospatial information to be collected, including the time that was spent in different areas (Fearnley, 2013). The Qstarz BT-Q1000XT is built with a MTK II GPS module, featuring 66 channels with a sensitivity of –165 dBm, with a stated horizontal accuracy of 3.0 m. With an operating **temperature of – 10 °C to 60 °C, the environmental conditions of the New Forest National Park were suitable for use of the equipment (Qstarz, 2013). The unit had the ability to record the location of a waypoint at set intervals, allowing it to be set to record every 5 seconds for this study. Initial testing of the units found that drift of the waypoints and signal loss occurred under dense canopy and inside buildings.**

The Qstarz BT-Q1000XT unit has been used in various studies that track movement patterns, from physical activity in adolescents (Moore et al., 2014), food access patterns in midlife and older adults (Huang et al., 2012) to tracking exposure to elevated concentrations of ultrafine particles that have negative consequences for health (Quiros et al., 2013). Duncan et al. (2013), found compared to 6 other commercially available GPS units, the Qstarz BT-Q1000XT showed the smallest mean error, with the accuracy of the unit under canopy cover being superior to all those tested. The unit was also overall the most accurate across six conditions (open sky, under beacon, residential, mixed use, under canopy and high rise) with a circular error probability (CEP) of 5.0 m (the radius centred on a geodetic point, within which 50% of recorded points have to fall) (Duncan et al., 2013).

Data were collected as a series of waypoints. The data points recorded were therefore positive location data - as heavily wooded areas with a lot of cover may be underrepresented in recorded points as **testing of the units found that drift of the waypoints and signal loss occurred in these areas.** The units were set to record the latitude and longitude every 5 seconds, for the full duration of each visit, the default for the unit.

GIS analysis

The data was extracted using the default software supplied with the units - Qstarz QTravel 1.46 (Qstarz International Company Limited, Taipei, Taiwan). The waypoints were then exported as a 'gpx' (GPS Exchange Format) files and imported into ArcMap 10.1 (ESRI UK Limited, Aylesbury, UK) using the tool 'GPX to Features'. The waypoints were projected using British National Grid (OSGB 36) using the Ordnance Survey OSTN02 transformation (OSGB_1936_To_WGS_1984_7)' (University of Edinburgh, 2015). Broad Habitat data was used to ascertain the location of points in relation to different habitat types. This was acquired under a data supply agreement from Hampshire Biodiversity Information Centre (Hampshire County Council, Winchester) dated 1 February 2015 (job number: 5335). The 'Combine' tool was used to ascertain the number of GPS points that fell into each habitat type, giving the time spent in each habitat.

To compare the visitation data in relation to the availability of different habitat types, buffer zones of 500 m, 1000 m, 2000 m, 3000 m, 4000 m and 5000 m radii around car parks was calculated using the 'buffer' tool in ArcMap 10.1 to determine the area of each habitat within different distances from the car parks (Figure 3.3.4). The maximum size of the 5000 m buffer radius was chosen to capture all the GPS data points, as no user was found to travel more than 5000 m (as the crow flies) from the car park or origin. The resulting buffer shapefile was then used to 'clip' the visitation point shapefile and land cover polygon shapefile. The 'Combine' function was then repeated to count the amount of points in each habitat type, for each buffer area.

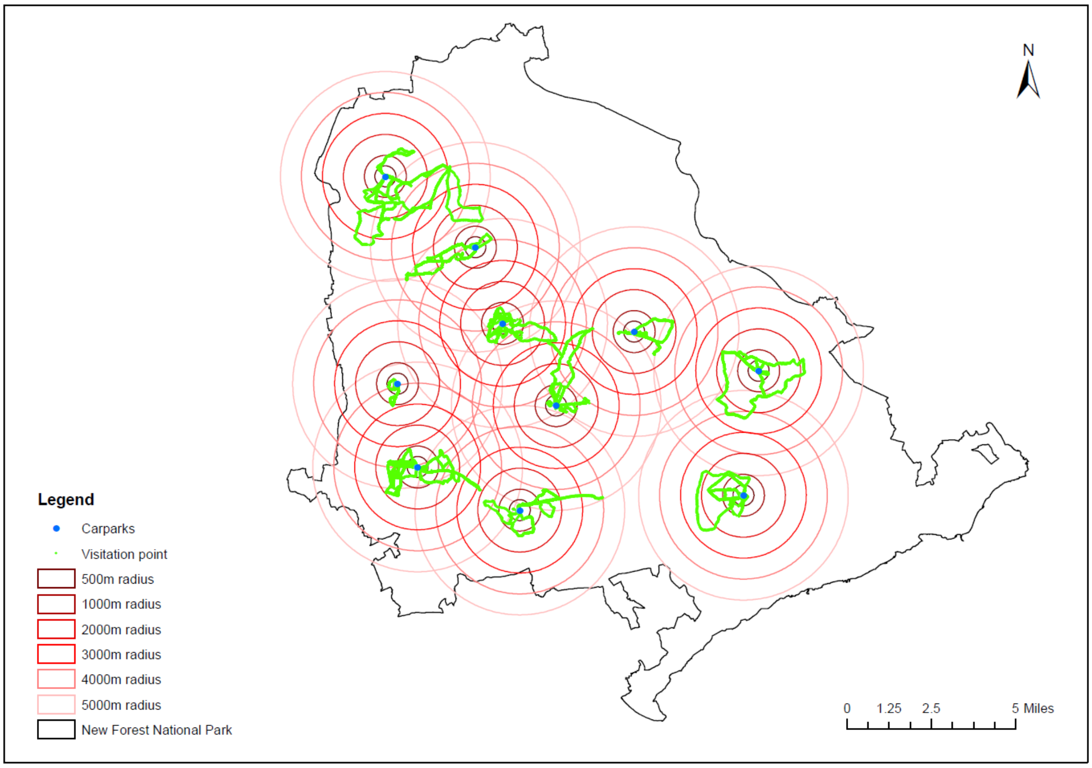


Figure 3.3.4: Car parks surveyed with buffer zones of 500 m, 1000 m, 2000 m, 3000 m, 4000 m and 5000 m, whereby GPS points recorded and habitat data were extracted to understand the effect on preference over varying distances from the car parks. Due to the scale of the map, the aggregated points appear as tracks, though no polyline data was used.

Data and statistical analysis

Statistical analysis was conducted using IBM SPSS Statistics v.22 (IBM United Kingdom Limited, Portsmouth, UK) with data being visually represented using SigmaPlot 13 (Systat Software Inc, London, UK) and SPSS.

The scoring of woodland images for recreation and aesthetic values were ordinal, hence it was treated as non-parametric data. The Friedman test is a non-parametric equivalent of the repeated measures analysis of variance (ANOVA) test for parametric data. When the results of the Friedman test are significant, at least one of the samples is significantly different from one of the other samples. To identify where the differences were present, sample contrasts or post hoc tests were used. The post hoc test used was the Wilcoxon signed ranks test (Corder, 2009). As all participants scored all of the woodland cover images, any mean differences between the images could not be explained by individual differences. Therefore, using repeated measures statistics, individual differences are eliminated from the test (Gravetter and Wallnau, 2012).

Strength of association testing for the woodland cover and preference score for both aesthetic appeal and recreational usage as well as strength of association between aesthetic score and recreation score was conducted using Spearman's rank-order correlation coefficient (r_s). Spearman's rank-order correlation coefficient is appropriate when both variables are ranked values.

Views from subsets of users for whether woodland was viewed as important for conservation, recreation value and aesthetic appeal were analysed using Mann-Whitney U tests. The categories used were mutually exclusive for users who scored a landscape devoid of woodland highly for aesthetic appeal and recreational usage (participants that scored the 0% woodland cover photo-realistic image as a 4 or 5) or whether they scored it low (participants that scored the % woodland cover 1 or 2), therefore repeated measures statistical tests were not required. Mann-Whitney U tests were also used to compare male and female scores.

For each buffer zone, chi-square test statistics were calculated summing the squared difference between observed and expected time spent across the land covers. Arable and horticulture, inland rock and littoral sediment were removed from the analysis, as all observed data across all buffer zones was zero. Rivers and streams, standing open water and canals plus unidentified water were grouped so expected values were above 5 for the test (though removed from 500 m analysis, as were bracken and neutral grassland due to the expected values being under 5 even after grouping). Only neutral grassland was removed from the 1000 m analysis due to the expected values being under 5.

Linear regression in SPSS allowed examination of how habitat availability and amount of time a visitor spent in a habitat were related. The adjusted R^2 value was reported, in lieu of R^2, as it corrected for positive bias, thus the resulting value was more indicative of a value that would be expected in the population. The adjusted R^2 value was also an estimate of the effect size (*d*) (Lund and Lund, 2013). Cohen (1988) defined effect size as: "small, d = .2," "medium, d = .5," and "large, d = .8". Effect sizes were taken into consideration to allow interpretation of the magnitude of the effect in the population, allowing 'real world' interpretation (Ellis, 2010).

3.4. Results

A total of 200 questionnaire surveys were completed, with 128 visitors choosing to additionally take part in the GPS tracking experiment.

Demography of participants

A wide spread of ages completed the survey (55% female/45% male), with 25% between 61-70 years old, 20.5% between 51-60 years old and 15.5%, 15%, 12.5% and 10% were 41-50, 31-40, 18-30 and over 71 years of age respectively. Only 1.5% were between 10-17 years old.

Participants were asked the main activity that they were undertaking on that visit. Multiple activities were allowed and recorded, hence percentages add up to over 100% (Cordingley, 2012). The majority visited for dog walking (37%) or walking (36.5%). Family outings were the next main reason (11.5%). Orienteering/geocaching, cycling and wildlife watching were also key reasons for a number of visitors (8%, 8% and 7% respectively). For a minority of visitors, exercising, picnicking, rest stops and pub lunching were the main reason for visiting (5%, 2.5%, 1.5%, and 1.5% respectively). The least recorded responses for visitation were watching aircraft, photography, horse riding and for educational purposes (all 0.5%).

The majority of participants lived in towns (45.5%), followed by villages (28%), cities (18%) and the countryside (8.5%). The majority also lived more than 30 km away (42%), followed by within 10 km (30%), within the New Forest Park boundary (18.5%) and between 10 – 30 km away (9.5%). Most people travelled by either car or van (99.5%) and the remaining by bicycle (0.5%).

Aesthetic appeal value of woodland in various stages of decline

The median aesthetic scores for the photo-realistic images of the different woodland cover types differed significantly (Friedman Test $\chi2= 263.666$, $P < 0.001$) between cover categories (Figure 3.4.1) with 100% woodland scoring highest, followed by 75% woodland. Post hoc Wilcoxon Signed Ranks Pairwise tests showed 100%, 50% and 0% woodland cover types had significantly differing median scores (all $p < 0.001$). The median scores for 100%, 75% and 50% woodland cover also significantly differed from 0% woodland cover (all $p < 0.001$), see Figure 3.4.1.

The mean scores decreased with each consecutive reduction in woodland cover (100% to 0%). The score distribution was skewed towards the ‘Very appealing’ end of the scale for

100% woodland (scores 4 and 5 totalling 83%) and 75% woodland (scores 4 and 5 totalling 76%). The lowest mean score was for 0% woodland, showing a more even distribution of 22.5%, 21%, 24, 14% and 18.5% for scores 1, 2, 3, 4 and 5 respectively.

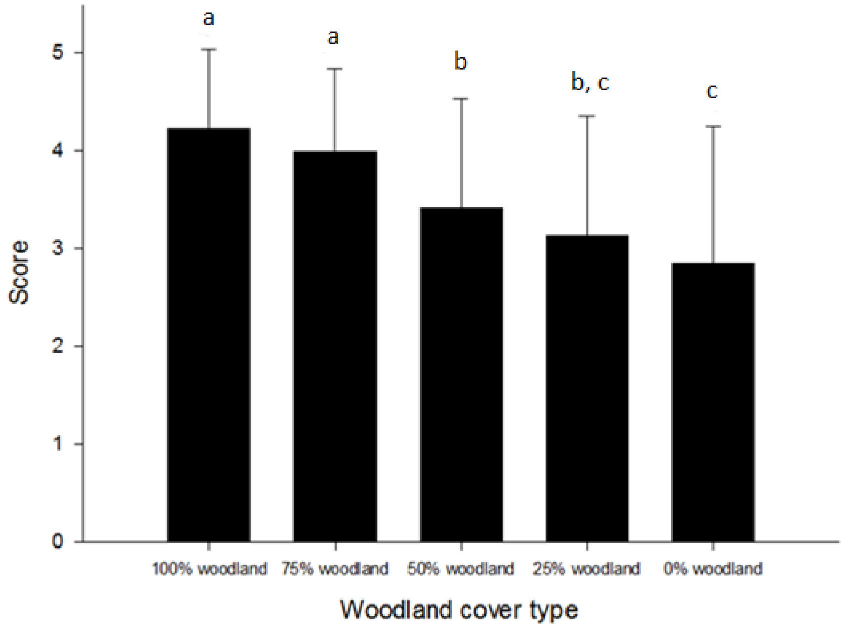


Figure 3.4.1: Bar graph illustrating mean aesthetic value scores for woodland cover types. Aesthetic score was rated on a scale of 1 to 5, with 5 meaning 'very appealing' and 1 meaning 'very unappealing'. Error bars present the standard deviation for each woodland cover type. The overall difference between the median ranks of aesthetic scores for different cover types (Figure 6) was significant (Friedman Test $\chi^{2(}4) = 263.666$, $P < 0.001$, n = 200). Bars grouped by the same letter are not significantly different from each other, using a Wilcoxon Signed Ranks Pairwise comparison performed with a Bonferroni correction for multiple comparisons, $P < 0.001$).

Strength of association between the percentage of woodland cover and aesthetic score were conducted (Table 3.4.1). As the percentage of woodland cover increased from 0% wood to 100% woodland, there was a positive relationship with both aesthetic appeal and recreational usage scores, with the aesthetic score being much stronger.

Table 3.4.1: Friedman tests and correlation between aesthetic appeal and recreational usage scores of visitors (n=200) and increasing percentage of woodland cover in presented photo-realistic images. Results show significant (* P < 0.05; ** P < 0.01; *** P < 0.001) or non-significant (n.s.) relationships.

Statistical test	**Aesthetic appeal score**	**Recreational usage score**
Friedman's test	263.666 ***	71.693 ***
Spearman's rank-order correlation coefficient (r_s)	0.402 **	0.172 **

Recreational usage value of woodland in various stages of decline

The highest mean score was for 75% woodland cover, followed by 100% woodland cover. The mean scores decrease in order between 75% and 0% woodland cover. There was a difference of 0.72 between the lowest and highest mean score. The score frequency distribution was skewed towards the 'Very appealing' end of the score scale for all woodland cover types except for 0% woodland cover, where it showed a more even distribution.

Median recreation scores were significantly different between different photo-realistic images of varying woodland cover (Friedman Test $\chi 2(4) = 71.693$, $p < 0.005$) (Figure 3.4.2). Post hoc Wilcoxon Signed Ranks Pairwise tests showed the densest woodland cover type (100%) had a significantly differing median score compared to no woodland cover (0%). Excluding 100% woodland, all other woodland cover types (75%, 50%, and 25%) had significantly differing median scores to 0% woodland cover (Figure 3.4.2). Strength of association between the percentage of woodland cover and recreation usage score were conducted using Spearman's rank-order correlation coefficient ($rs = 0.665$, $p < 0.01$). As the percentage of woodland cover increased, there was a positive relationship to recreation score. Test values, whilst positive and significant, were substantially weaker than those for aesthetic scores.

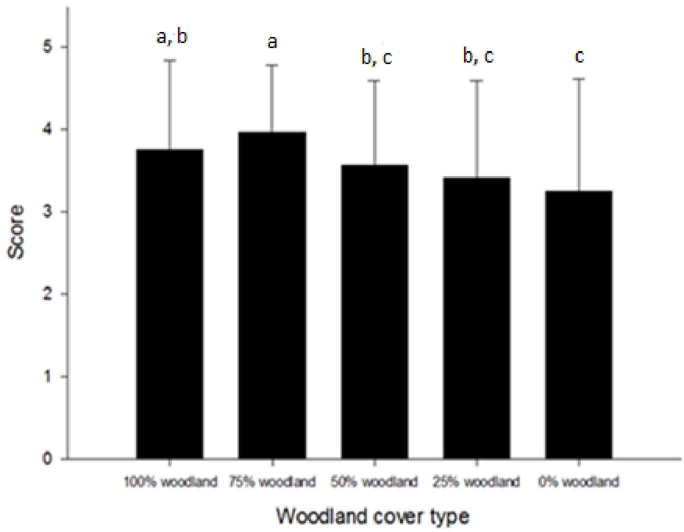


Figure 3.4.2: Bar graph illustrating mean recreation value scores for woodland cover types. Recreation score was rated on a scale of 1 to 5, with 5 meaning 'very appealing' and 1 meaning 'very unappealing'. Error bars present the standard deviation for each woodland cover type. The overall difference between the median ranks of recreation scores for different cover types was significant (Friedman Test $\chi^2(4) = 71.693$, $P < 0.005$, $n = 200$). Bars grouped by the same letter are not significantly different from each other (Wilcoxon Signed ranks Pairwise comparisons were performed with a Bonferroni correction for multiple comparisons, $P < 0.001$).

Relationship between aesthetic appeal scores and recreational usage scores

The relationship between aesthetic score and recreation score indicated by participants for the photo-realistic images of woodland cover were explored by testing for strength of association between both score types using Spearman's rank-order correlation coefficient scoring a r_s statistic of 0.665 ($P < 0.01$). As the aesthetic score increased, there was a positive relationship with recreation score.

Landscapes devoid of woodland

To investigate the relationship between visual aesthetic values of woodland and other features that woodlands provide, a comparison between two subsets of users was conducted. The 0% woodland image was used to divide the participants into two groups: (a) those that found landscape a devoid of woodland aesthetically appealing (participants that scored the image as a 4 or 5, n = 65) (Figure 3.4.3 HL) and (b) those who did not find it appealing (scoring the image as a 1 or 2, n = 87) (Figure 3.4.3 LL). Participants who scored the image of landscape devoid of woodland as highly aesthetically appealing were significantly more likely to rate woodland as being more important for recreation and aesthetic value, but there was no difference between the groups in the conservation value rating.

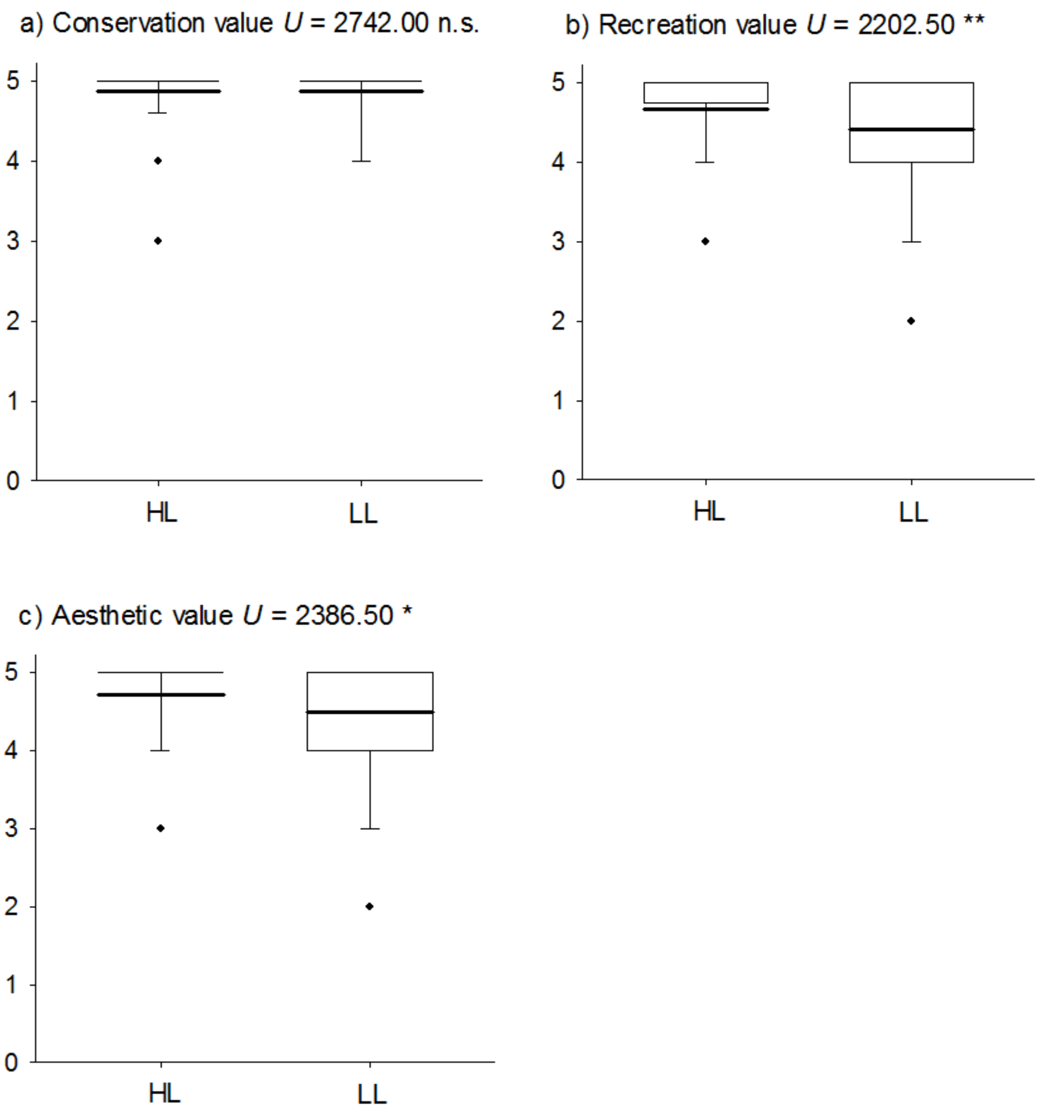


Figure 3.4.3: Box plots illustrating how important woodland is for providing (a) conservation value, (b) recreation value and (c) aesthetic value, for participants who found a landscape with no woodland aesthetically appealing, scoring 4 or 5 (HL) and those who did not find it appealing, scoring it 1 or 2 (LL). Mann-Whitney U tests show either significant (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significant (n.s.) differences between participants with different preferences. Mean line shown in bold.

Male and female comparisons

Aesthetic and recreational values were compared between male and females (Table 3.4.2). Females were found to significantly score denser wooded area higher for recreational usage (with mean scores of 4.01 for 75% and 3.95 for 100% woodland; mean male scores were 3.83 and 3.52 respectively). No further differences were found.

Table 3.4.2: Comparison of differences between males and females in scoring all woodland cover type images, for both aesthetic appeal and recreational usage. Mann-Whitney U tests show either significant (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significant (n.s.) differences between respondents with different preferences.

	Aesthetic appeal			**Recreational usage**		
Woodland cover type	**Male median score**	***Female median score***	***U***	**Male median score**	***Female median score***	***U***
100%	4	4	4697.00 n.s.	3.5	4	3929.00 **
75%	4	4	4511.00 n.s.	4	4	4188.50 *
50%	3	3	4941.50 n.s.	4	4	4865.00 n.s.
25%	3	4	4727.50 n.s.	4	3	4865.00 n.s.
0%	3	4	4913.50 n.s.	4	3	4493.00 n.s.

Tree preferences

Participants specified a preference to broadleaved trees, followed by holding no preference on species type, with 56% and 38% respectively (Figure 3.4.4). Larger or veteran trees were preferred by the majority at 70.5% compared to smaller trees (Figure 3.4.5).

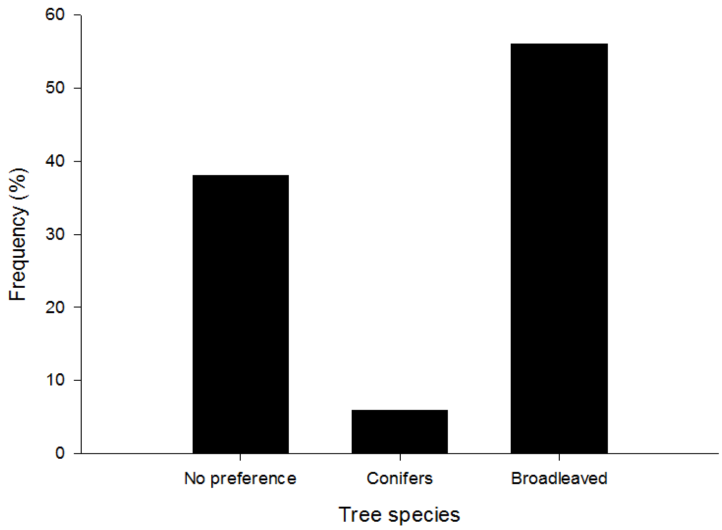


Figure 3.4.4: Tree species preferences in response to questionnaire.

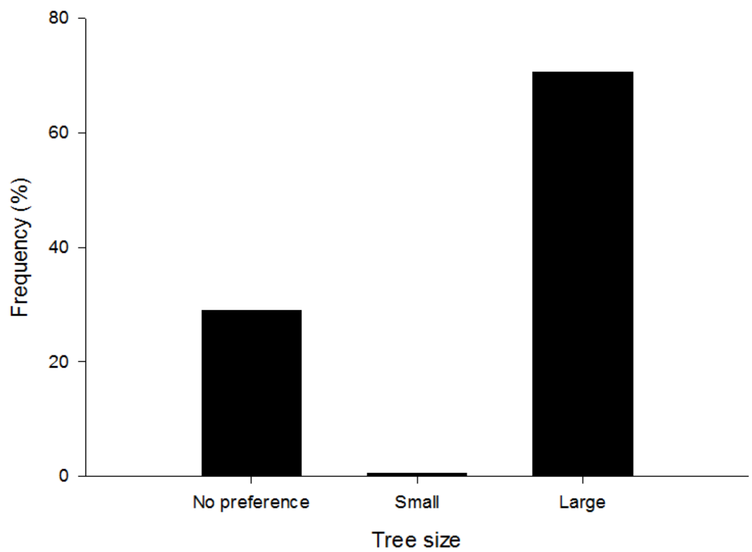


Figure 3.4.5: Tree size preferences in response to questionnaire.

Conservation, recreation and aesthetic value of woodlands

The majority of participants indicated that woodland is very important for conservation, with 87.5% selecting the highest score and a further 11% selecting a score of 4 (Figure 3.4.6). Similarly, participants indicated that woodland is very important for recreation, with 60% scoring 5 and 28% scoring 4 (Figure 3.4.7). Aesthetic value of woodlands was also deemed to be of high importance for woodlands to provide, with 68% scoring 5 and 22.5% scoring 4 (Figure 3.4.8).

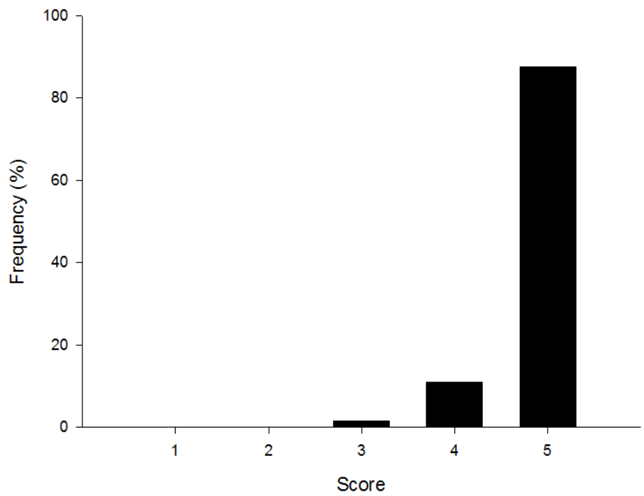


Figure 3.4.6: Bar chart illustrating frequency (%) scores for conservation value of woodlands. Conservation score was rated on a scale of 1 to 5, with 5 meaning ‘very unimportant’ and 1 meaning ‘very important’.

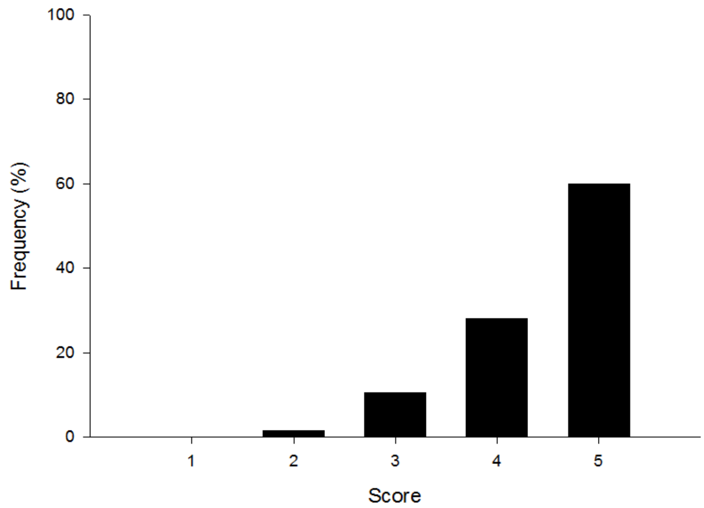


Figure 3.4.7: Bar chart illustrating frequency (%) scores for recreation value of woodlands. Recreation score was rated on a scale of 1 to 5, with 5 meaning 'very unimportant' and 1 meaning 'very important'.

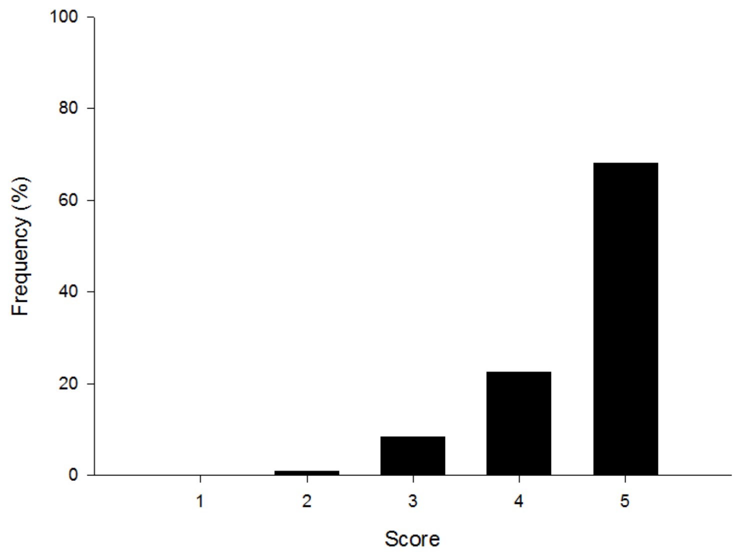


Figure 3.4.8: Bar chart illustrating frequency (%) scores for aesthetic value of woodlands. Aesthetic score was rated on a scale of 1 to 5, with 5 meaning 'very unimportant' and 1 meaning 'very important'.

Dieback due to climate change

Participants were concerned that mature beech woods of the New Forest were dying due to climate change (mean of 4.20), with the majority of participants scoring a 4 (36.5%) or 5 (44%) (Figure 3.4.9).

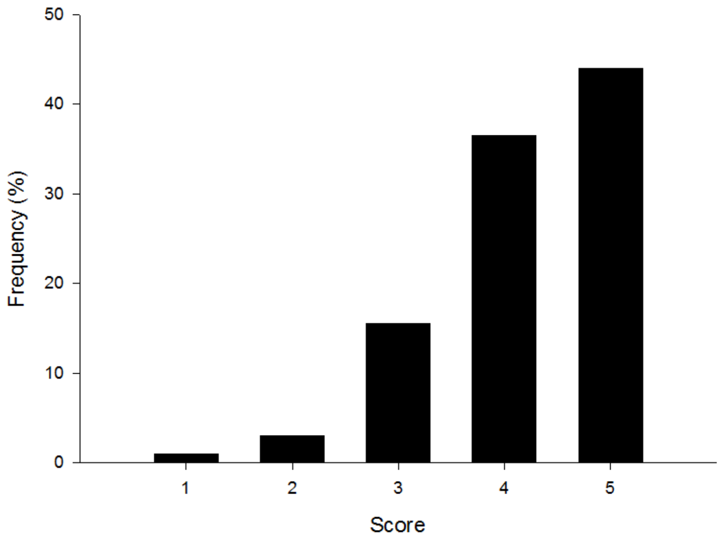


Figure 3.4.9: Bar chart illustrating frequency (%) scores for concern that mature beech woods of the New Forest are dying because of climate change. The concern score was rated on a scale of 1 to 5, with 1 meaning ‘not at all concerned’ and 5 meaning ‘very concerned.

GPS survey

In total, 128 participants completed the additional GPS tracking part of the survey across all ten car parks (Figure 3.4.10).

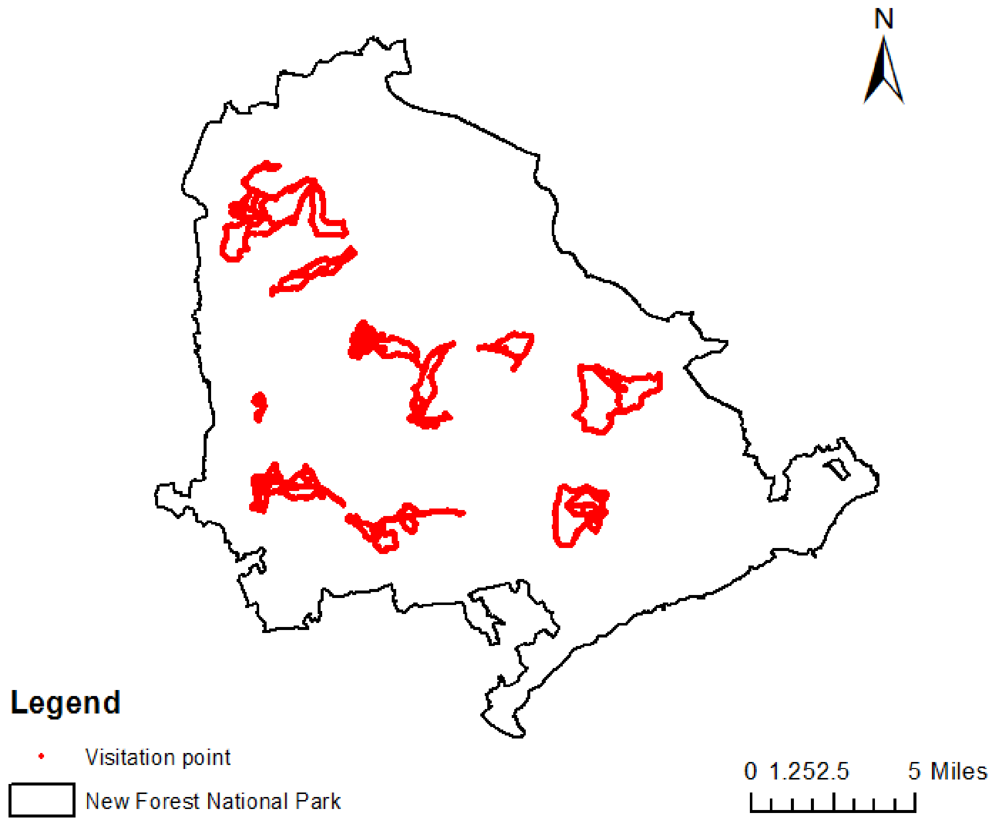


Figure 3.4.10: Map showing the GPS points (taken at 5 second intervals) from 128 participants who took part in the optional GPS tracking part of the survey. Visitation points appear to create polylines; this is only an artefact of the extent of the map and the density of the points.

Time spent in habitats

Overall, the majority of time spent by visitors was in dwarf shrub heath. This was followed by acid grassland and broadleaved, mixed and yew woodland (Figure 3.4.11).

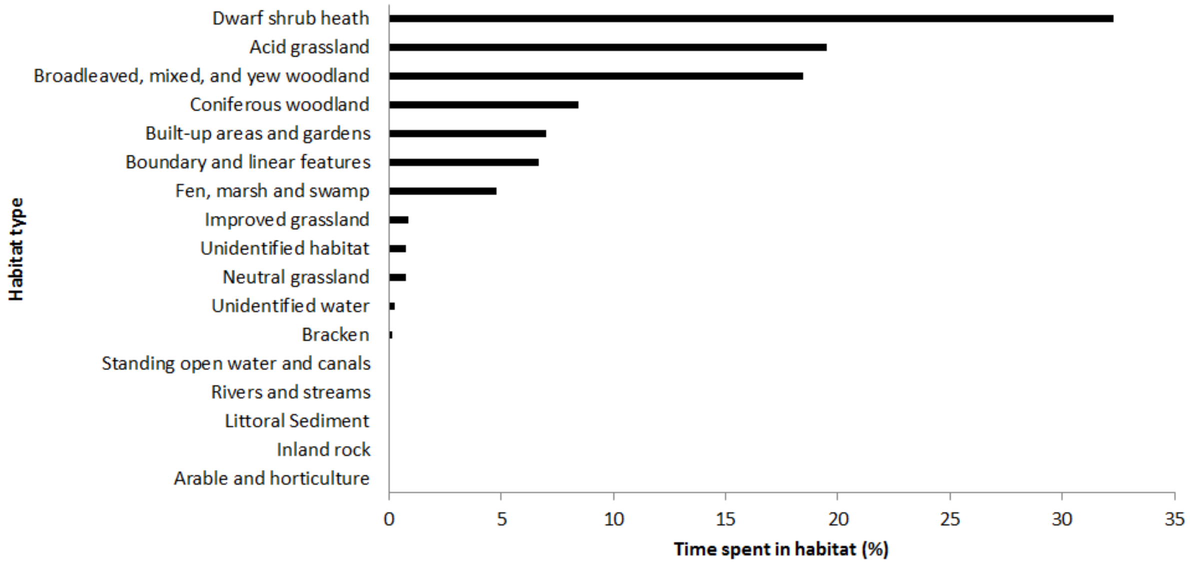


Figure 3.4.11: Overall time spent (%) in different habitats of the New Forest, for all users.

To investigate the relationship of distance travelled from the access point (car park) and the effect on preference for habitat, multiple buffer zone radii were analysed. This data (collated in Table 3.4.3) was then used to plot scatterplots of habitat area available (%) against time spent in habitat (%) and linear regression analysis was conducted for each zone. The resulting R^2 values were then plotted, and a further linear regression analysis conducted (Figure 3.4.14).

Table 3.4.3: Time spent and habitat available in buffers of varying radii.

Habitat	500 m buffer		1000 m buffer		2000 m buffer		3000 m buffer		4000 m buffer		5000 m buffer		No buffer	
	Area (ha)	Time (mins)	Area (ha)	Time (mins)	Area (ha)	Time (mins)	Area (ha)	Time (mins)	Area (ha)	Time (mins)	Area (ha)	Time (mins)	Area (ha)	Time (mins)
Acid grassland	126.40	506.00	398.09	1000.92	1254.15	1389.42	2128.04	1472.42	2602.72	1562.25	2855.81	1562.25	3150.55	1562.25
Arable and horticulture	0.00	0.00	2.79	0.00	120.12	0.00	731.78	0.00	1850.75	0.00	3325.13	0.00	5608.47	0.00
Boundary and linear features	16.08	301.83	42.57	390.83	163.22	472.17	325.59	509.58	512.25	533.67	659.71	533.67	940.48	533.67
Bracken	4.92	3.42	30.62	6.58	135.06	9.33	195.68	12.25	263.23	12.25	286.01	12.25	306.41	12.25
Broadleaved, mixed, and yew woodland	139.42	217.00	566.67	519.33	2776.00	1033.50	6514.29	1317.92	9518.38	1477.75	11547.26	1477.75	14255.19	1477.75
Built-up areas and gardens	25.90	181.00	96.75	319.00	384.72	440.25	847.02	548.83	1351.23	561.75	1829.86	561.75	2687.64	561.75
Coniferous woodland	62.50	187.67	311.38	318.83	1283.95	505.08	2730.08	674.00	3883.53	676.00	4307.53	676.00	4906.42	676.00
Dwarf shrub heath	295.01	862.17	1174.97	1560.67	4081.18	2237.42	6930.50	2560.00	8168.57	2581.25	8881.77	2581.50	9881.55	2581.50
Fen, marsh and swamp	57.39	146.92	304.31	262.50	1108.89	352.42	2046.86	384.33	2554.99	384.33	2751.26	384.33	3032.06	384.33
Improved grassland	38.20	28.50	139.87	44.08	938.72	69.42	2788.59	70.58	4812.52	70.58	6808.80	70.58	10898.47	70.58
Inland rock	0.00	0.00	0.00	0.00	0.00	0.00	3.72	0.00	13.50	0.00	21.75	0.00	50.24	0.00
Littoral Sediment	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	12.29	0.00	45.95	0.00	1247.21	0.00
Neutral grassland	4.37	1.17	9.18	1.17	82.09	60.25	226.49	60.25	399.43	60.25	534.80	60.25	1214.31	60.25
Rivers and streams	0.00	0.00	0.00	0.00	0.07	0.00	3.81	0.00	8.57	0.00	23.41	0.00	118.18	0.00
Standing open water and canals	0.07	0.00	0.31	0.00	1.65	1.92	6.75	1.92	21.27	1.92	46.52	1.92	98.15	1.92
Unidentified habitat	11.70	17.33	52.22	51.75	177.65	60.75	406.88	60.75	602.59	60.75	807.80	60.75	1419.37	60.75
Unidentified water	3.24	3.00	11.43	7.92	48.44	13.92	116.45	18.67	177.26	18.67	218.46	18.67	288.72	18.67

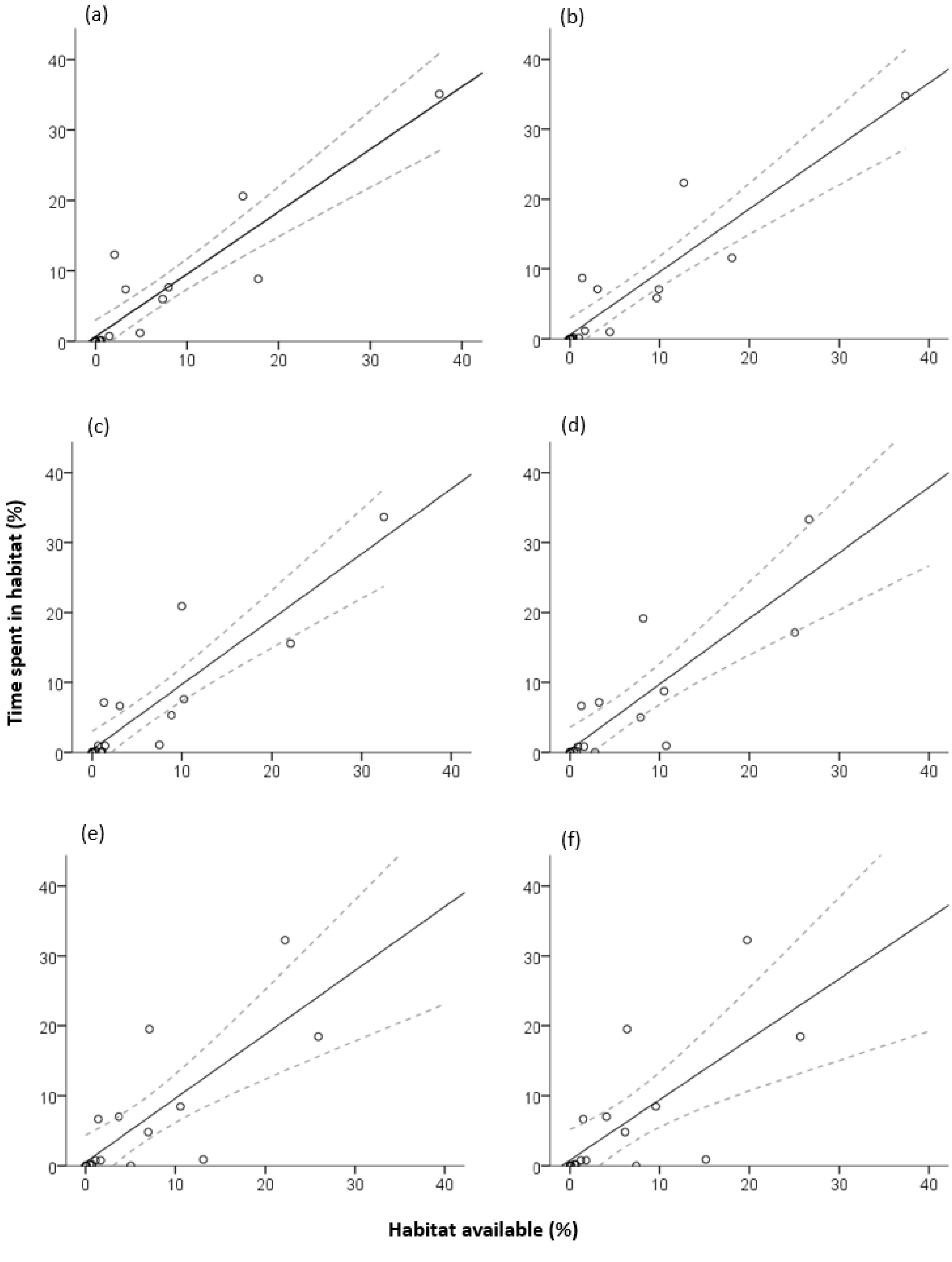


Figure 3.4.12: Scatterplots illustrating the relationship between habitat available and time spent in habitat by visitors across multiple buffer radii: (a) 500 m, (b) 1000 m, (c) 2000 m, (d) 3000 m, (e) 4000 m and (f) 5000 m. Lines of best fit (R^2) with mean prediction interval lines shown as dashed lines.

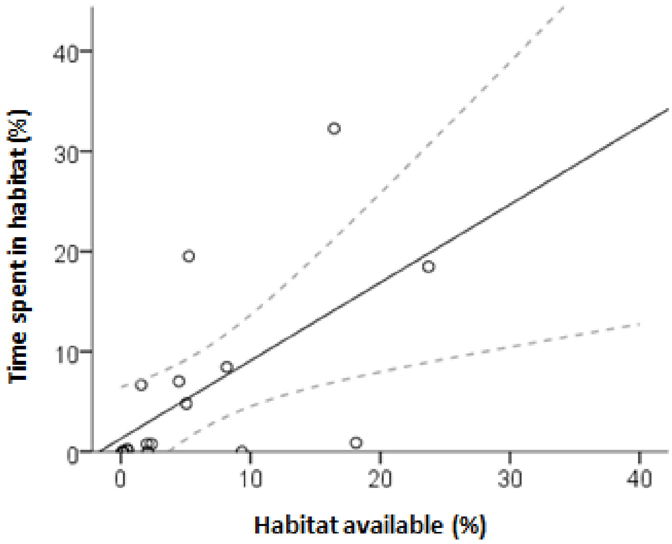


Figure 3.4.13: Scatterplot illustrating the relationship between habitat available and time spent in habitat by visitors across the full extent of the New Forest National Park with no buffer. Line of best fit (R^2) with mean prediction interval lines (dashed) are shown.

Visual assessment of the scatterplots of the relationship between time spent in habitats by participants and buffer radii centred on the car park of access showed a linear relationship (Figure 3.4.12, Figure 3.4.13). A linear regression showed that habitat availability could significantly ($P < 0.001$) predict the time visitors would spent in each habitat type, between 500 m and 4000 m radii from the car park with habitat availability accounting for 58.8% - 83.5% of the explained variability in time visitors spent in habitats (using adjusted R^2 values) (Table 3.4.4). To a lesser extent, a 5000 m buffer radii and no buffer could significantly predict the same. The decreasing adjusted R^2, F and increasing p- values between 500 m and 5000 m, and no buffer, inferred that using habitat availability as a predictor for visitation become less reliable as the visitor travels further from the car park. A curvilinear relationship existed between the buffer radius and R^2 values (Figure 3.4.14) hence regression analysis using multiple equations was conducted (Table 3.4.5). A significant quadratic relationship was seen, confirming that the further the distance a visitor travelled the relationship between available habitat and time spent in that habitat gets weaker. The further visitors travelled, the less limited they were by what was available.

Table 3.4.4: Linear regression between habitat availability and time spent in habitat by visitors across multiple buffer radii, T = Time spent in habitat (%), h = Habitat availability (%).

Buffer radii	**Adjusted R^2**	***F*(1,15)**	**Regression equation**
500 m	0.835	81.868 ***	T = 0.667 + (0.887 x h)
1000 m	0.831	79.739 ***	T = 0.584 + (0.901 x h)
2000 m	0.798	64.104 ***	T = 0.396 + (0.933 x h)
3000 m	0.704	38.996 ***	T = 0.357 + (0.939 x h)
4000 m	0.588	23.842 ***	T = 0.507 + (0.914 x h)
5000 m	0.482	15.876 **	T = 0.799 + (0.864 x h)
No buffer[a]	0.322	8.582 *	T = 1.296 + (0.780 x h)

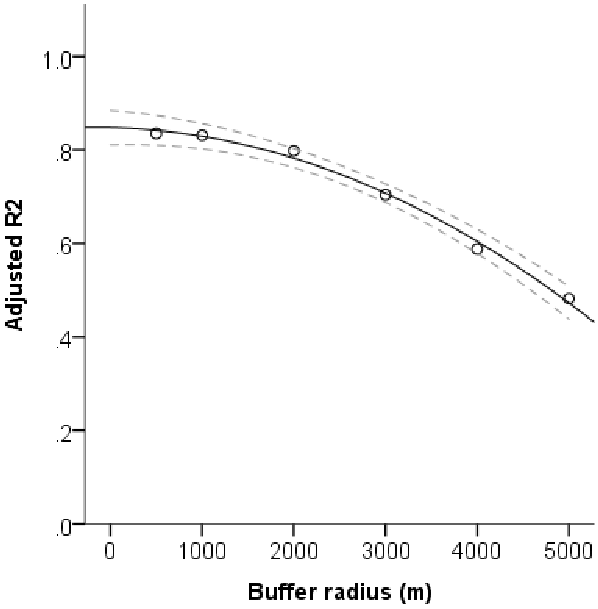


Figure 3.4.14: Scatterplot of R^2 vs. proportional time spent in habitat as a function of the available habitat varying radii of the surveyed car parks. Lines of best fit (R2 quadratic) with mean prediction interval lines (dashed) are shown.

Table 3.4.5: Results of regression analysis for the R^2 values across all buffer radii. Regressions show significant (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significant (n.s.). F statistic degrees of freedom are given in brackets.

Equation	**R^2**	**F**
Linear	0.948 **	(1,4) 73.527
Quadratic	0.994 ***	(2,3) 238.734

The individual relationship between individual habitat types and time spent in the habitat was explored. Chi-squared statistics (Table 3.4.6) showed increasing test values as the buffer zone increases; this shows the observed time spent across land cover types was significantly different from those expected.

Table 3.4.6: Chi square test statistic for all habitat types for all buffer zones. Habitat types were grouped for ‘rivers and streams, standing open water and canals and unidentified water’ and ‘’so that no value was under 5. Arable and horticulture, inland rock and littoral sediment were removed from the analysis, as all observed data across all buffer zones was zero. [a] The habitats ‘bracken’ and ‘neutral grassland’ and the grouping ‘rivers and streams, standing open water and canals and unidentified water’ were removed from the 500 m analysis due to values being under 5. [b]Only neutral grassland was removed from the 1000 m analysis due to the value being under 5. Results show significant (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significant (n.s.).

Buffer zone (radius)	**Chi-square test statistic**
500 m[a]	1612.04 ***
1000 m[b]	2739.04 ***
2000 m	3505.63 ***
3000 m	4443.12 ***
4000 m	5216.95 ***
5000 m	5843.97 ***
New Forest National Park	7391.55 ***

The majority of participants did not go beyond 3000 m (Table 3.4.3), hence the 3000 m buffer provides a robust area to consider. Participants spent 429.11% more time in boundary and linear features than would be expected, followed by 133.19% and 119.05% above the expected time for acid grassland and built-up areas and gardens. Improved grassland and bracken show a 91.44% and 78.84% deduction in the expected time, signalling a preference to avoid these habitats. If the entirety of the New Forest National Park is considered, 3.49% over expected time was spent in coniferous woodland compared to a decrease of 22.13% of time spent in expected for broadleaved, mixed and yew woodland. Overall people are spending more time in coniferous forest than broadleaved than was expected. As people spent longer in the New Forest National Park, and travel further, they spent longer in dwarf shrub heath, with an increase over expected time occurring after 2000 m. Within 500 m of the car park people show a preference for staying on footpaths as demonstrated by a 500.11% increase in time over expected spent on boundary and linear features (Table 3.4.7).

Table 3.4.7: Percentage difference in observed and expected time spent in habitats by participants. *Categories not included due to expected values being under 5, [a]New Forest National Park

BAP Broad habitat	Buffer zone (radius)						
	500 m	1000 m	2000 m	3000 m	4000 m	5000 m	NFNP[a]
Acid grassland	27.98	76.15	109.31	133.91	175.71	207.33	272.46
Boundary and linear features	500.11	543.21	446.54	429.11	378.54	354.46	326.23
Bracken	*	-84.94	-86.94	-78.84	-78.62	-75.94	-69.97
Broadleaved, mixed, and yew woodland	-50.24	-35.79	-29.66	-31.60	-28.69	-28.10	-22.13
Built-up areas and gardens	123.42	131.00	116.20	119.05	90.96	72.47	57.00
Coniferous woodland	-4.00	-28.26	-25.68	-16.54	-20.04	-11.83	3.49
Dwarf shrub heath	-6.57	-6.94	3.58	24.88	45.15	63.29	96.23
Fen, marsh and swamp	-18.16	-39.57	-39.96	-36.52	-30.91	-21.52	-4.79
Improved grassland	-76.15	-77.92	-86.03	-91.44	-93.26	-94.18	-95.14
Neutral grassland	*	*	38.66	-10.07	-30.71	-36.71	-62.73
Unidentified habitat	-52.64	-30.57	-35.39	-49.52	-53.69	-57.75	-67.85
Water (River and streams, standing open water and canals, unidentified water)	*	-52.76	-40.36	-45.21	-54.35	-59.90	-69.39

3.5. Discussion

Results were striking and unexpected: visitors indicated that they prefer wooded landscape in survey responses; tracked behaviour indicated otherwise. Behaviourally much less time than expected was spent in woodland, with tracked participants spending more time in boundary and linear features, acid grassland and built-up areas and gardens than expected given their relative availability.

Stated survey responses indicated a positive relationship between the amount of woodland cover and both the recreational usage and aesthetic appeal. Aesthetic appeal was the stronger relationship. Average recreational scores were highest for 75% woodland that 100% woodland; confirming those by Hunziker (1995) whom found that people preferred partially re-afforested landscapes and a study on urban woodland in Sweden for recreation preferences found that open forests were considered most attractive (Heyman et al., 2011). This supports research findings from stated responses that highlights a preference for wooded over open landscapes in natural and semi-natural systems (Cordingley, 2012, Schroeder and Orland, 1994, Cordingley et al., 2015). Psycho-physiological responses to desert, grassland, tundra and three forest biomes (coniferous, deciduous and tropical forest) have also shown that people prefer tundra and coniferous forest the most, with desert and grassland being the least popular (Han, 2007). A study eliciting responses for individuals for 71 structurally varying landscapes found that tall and dense vegetation was judged as being more natural than low, open vegetation (Lamb and Purcell, 1990). Considered in isolation, the stated survey response findings in this study support an assemblage of stated preference research that has taken place over the last few decades. However, the GPS monitoring results support previous research that has found that people prefer open savannah-like natural environments (Falk and Balling, 2010, Sharp et al., 2012, Cordingley, 2012).

The difference in the stated and monitored methods may be explained by people preferring images with trees or nature to those without (Lohr and Pearson-Mims, 2006). Natural settings have been found to elicit moderate fascination and aesthetic pleasure, having restorative effects on people (Herzog et al., 1997), with Kaplan (1977) suggesting that people are fascinated by and value nature, despite being potentially freighted by it. Research has also shown that individuals exposed to stressful situations will recover faster and more completely when exposed to natural, compared to urban video scenes (Ulrich et al., 1991). Even the addition of plants in windowless rooms has been found to make people more attentive (Lohr et al., 1996), and even pain control is improved in patients exposed to nature murals and nature sounds (Diette et al., 2003).

In the stated survey responses visitors preferred larger, broadleaved trees. The value of broad leafed trees has also been demonstrated in a study in Sweden by Bostedt and Mattsson (1995) analysing two forest areas of tourism value. The study found that value of the forests was attributable to forest characteristics and that by increasing the proportion of broadleaved trees in forest stands, value could be effectively increased. It has been suggested that although the timber values of broadleaves are lower than conifers in Sweden, the non-monetary value of broadleaves is still high (Holgén and Bostedt, 2004). Though in contrast, people surveyed in a national household survey in Denmark found that more than 60% of the population in Northern Zealand preferred coniferous to broad-leaved forests (Paracchini et al., 2014).

Participants here may have preferred looking at the survey images, though danger and fear responses were activated when present outdoor, faced with multiple habitat types. This may also have been guided by familiarity with the area. Heyman et al. (2011) found that in Västra Götaland, Sweden in boreal forests infrequent visitors preferred more open forests, whereby regular visors preferred more closed forests. Research has shown that people prefer open forests with areas of cleared small shrubs (Ribe, 1989). Tyrväinen et al. (2003) found during the participatory planning process of Helsinki City forests, that adults and children valued open forests more than closed forests, with youth appreciating dense wild forests. Gundersen and Frivold (2008) reviewed 53 studies investigating mostly boreal coniferous forest landscapes in Finland, Sweden and Norway. The study found that most studies reported that people preferred older stands of forest and larger trees, with the feeling of accessibility being very important. The contradictory nature of the results can be seen in other studies where participants scored open woodland highly, yet a majority claimed they preferred a mix of both open and closed woodland (Heyman et al., 2011). However, regardless of visual and recreation preferences for woodland in landscapes, visitors believed that woodland was highly important for conservation, recreation and aesthetic value.

A sense of danger during the GPS monitored method may account for the difference in results between techniques. Herzog and Kutzli (2002) investigated perceived danger and fear in a variety of field and forest settings, and found access, both visual and for movement possibility (locomotor), an important predictor for danger. Herzog and Smith (1988) undertook a study on danger, mystery, and environmental preference using urban and non-urban field and forest setting finding that danger negatively, and mystery positively predicted preference. A study by Herzog and Kropscott (2004) found that legibility (the apparent ease with which features and parts of a scene can be recognised), landmarks and visual access were positive predictors of preference. Historically, our ancestors would have sought access in natural environments for survival (surveillance, exploration, and escape) (Appleton, 1975). Hence people may have scored woodland highly, but when movements were tracked decided not to venture into denser, less accessible areas such as woodland as danger or fear were perceived. Alternatively, it may simply been the desire to move, or exercise that caused visitors to use more open habitats, with more time being spent in boundary and linear features. Hence it may be accessibility and paths that were an important factor in visitor's decisions, though this would not be of the same magnitude across all habitat types.

The presence of footpaths may have biased the results, with participants potentially preferring east access routes. This can be seen within 500 m of the car park people showing a preference for staying on footpaths (with a 500% increase in time expected to be spent on linear features. In future research the footpath network of the New Forest could be buffered and different habitat types with these buffers measures and compared to ascertain participant preferences.

A clear aversion to spending time in improved grassland and bracken was also noted, despite the accessibility of the former. Distance travelled was a major factor in preference for habitats; those travelling only short distances tended to use habitat as available, those who travelled longer distances sought out particular habitats. Human evolution in open-savannah landscapes has led to theories that humans prefer open landscapes (Appleton, 1975). This has been supported by various studies, for example Falk and Balling (2009) found that individuals overwhelmingly selected savannah scenes over rainforest, deciduous forest, coniferous forest and desert. Previous research showed that people found large, spreading trees (such as those found on the African savannah) more attractive than rounded or column shaped trees, supporting the savannah hypothesis (Lohr and Pearson-Mims, 2006). As this survey question had no images, it was assumed that participants viewed broadleaved trees as large and spreading, though potentially rounded too. Surprisingly, people spent more time than expected in coniferous woodland, if we look at the scale of the whole national park, by nearly 3.5% whereby over 22% deduction in proportional time can be seen for broad-leaved woodland, despite people stating they preferred broad-leaved woodland.

The nature of this study covered a large area so sampling resources were spread out; an expansion of this research design, with an increased sample size would offer more robust results as out of the 142 car parks that are present in the New Forest National Park (New Forest Park Authority (NFPA), 2007b) less than 10% were surveyed in this study. A slight limitation in regards to using high value GPS equipment was using carparks as access points, as train and foot visitors were not surveyed, but it has been estimated that 96% of visitors travel by car (New Forest Park Authority (NFPA), 2007b), hence the bias was minor. As the data points recorded were positive location data, heavily wooded areas with a lot of cover may be underrepresented in recorded points since GPS devices are less precise in these conditions, this could partially account for the lower than expected time spent in woodlands.
The use of manipulated images in this study allowed the variable of interest, woodland cover, to be assessed independently by keeping other variables such as sky, distance, etc. unchanged. Though the process of deciphering visitor's preferences for visual scenes alone is troublesome; a percentage of habitat cover represented in an image does not mean that that percentage should be maintained as a habitat in the a wider landscape (Cordingley, 2012).

If the landscape is to be managed for aesthetic appeal, then it is difficult to translate images of Beech woodland in varying percentages to a heterogeneous landscape, though the results do indicate the aesthetic preferences visitors hold. From a management perspective it is important to understand which landscapes people prefer; as these gain the most acceptance and support (Sharp et al., 2012, Vouligny et al., 2009, McFarlane and Boxall, 2000) with strong aesthetic responses to landscape being an excellent starting point to consider actions for landscape change. For example, aesthetic appreciation has been found to be the strongest motivator for non-industrial private forest (NIPFs) purchases in the Midwestern USA landscape (Erickson et al., 2002). Eagles and McCool (2002) states that national park designations are often placed on those that people attach aesthetic and cultural meaning to. It has been suggested that a wide range of values including aesthetic and cultural values be used in making balanced management decisions on visitor use of national parks (Eagles and McCool, 2002).

It is suggested that the key is to couple aesthetic appeal with other cultural ecosystem services such as recreational behavioural monitoring to have a holistic informed approach. This study has highlighted the need for utilising appropriate methodologies for the assessment of different kinds of cultural ecosystem services, ultimately if trade-offs with cultural services against other ecosystem services are to be considered, it is of utmost importance that they are correctly assessed. As multiple-methods are not usually used when collecting opinions and views of stakeholders, it is important that it be understood how the method utilised will impact the results, be it stated or behavioural. Further research is needed to understand this difference, and whether it impacts other types of land classification and areas. Further research is needed on the extent and interdependence of aesthetic appeal and recreation usage, and trade-offs that visitors to a landscape are willing to accept as part of the process of managing for cultural use.

Aesthetics are seen to be a highly important value of forest landscapes, and is often taken in consideration in public forest management (Gobster, 1999), with aesthetics being a key feature in the people-landscape interface (Kaplan and Kaplan, 1989). Perceptions of landscapes are often based on the aesthetic of the area (Gobster, 1999), with this especially being true of forest landscapes (Ribe, 1994), thus is vitally important to consider aesthetics and recreation. If the precursor to recreation is aesthetics, although these two cultural ecosystem services can be decoupled; ultimately, one seems to affect the other as can be seen from the positively correlated status of the services. The implication of these results is clear; reliance on one method of assessment for cultural ecosystem services can lead to misleading results.

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Chapter 4

Does the New Forest National Park meet current management objectives?

4.1. Abstract

The use of spatial values for ecosystem services have been found to aid management decisions regarding land use. Multiple studies have now examined trade-offs and synergies amongst multiple ecosystem services and biodiversity, though a limited amount of studies have investigated these in protected areas. In this study a comprehensive of ecosystem services are mapped including cultural, supporting, provisioning and regulating services, with biodiversity included as a supporting service. Multiple methods of mapping ecosystem services were adopted, including the use of the spatially explicit modelling tool InVEST, and GIS techniques. These services were compared within and outside Sites of Special Scientific Interest (SSSIs), using the New Forest National Park as a case study. The SSSI acts as the 'core' area and the National Park as the 'buffer'. All ecosystem services analysed were significantly different in and outside the SSSI area, with most services being higher inside the SSSI. The high ecosystem provisioning areas were identified using a technique to determine percentiles of combined ecosystem service value, with the SSSI showing an increase 23% over expected for the 90th percentile. The high provisioning of ecosystem services adds another dimension to the value of protected areas, moving beyond just conservation of biodiversity. The New Forest Partnership Plan was utilised to ascertain whether management objectives were being met in the park. This study further highlights the need for ecosystem service assessments to include multiple services and the role protected areas play in preserving high-value ecosystem service provision. Using spatially explicit methods to understand the spread of ecosystem services and biodiversity can lead to more efficient land management decisions.

4.2. Introduction

The Millennium Ecosystem Assessment (2003) defines ecosystem services as ‘the benefits people obtain either directly or indirectly from ecological systems’. Ecosystem service identification, quantification and mapping is increasingly used as a tool for the management of environmental resources (Troy and Wilson, 2006) and is important to understand where conservation efforts impact further than just a biodiversity focus (Naidoo et al., 2008). Where ecosystem services are not quantified, they are often ignored by those making land-management decisions (Nelson et al., 2009). Mapping multiple ecosystem services allows the provision from a broad range of ecosystem services to be visualised spatially and analysed.

A wide array of benefits from nature are enjoyed by people, and a large number of these ecosystem services need to be considered to identify the overall provision we get from different areas. Spatially explicit values for ecosystem services can aid in management decisions regarding land-use (Nelson et al., 2009). It is important to look at multiple ecosystem services so that synergies and trade-offs, or hot and cold spots, can be identified. High areas of ecosystem service provision and biodiversity can lead to ‘multiple wins’ in conservation land management by protecting both. The impacts of conservation on ecosystem services have been investigated by several studies. Bradbury et al. (2010) suggest that the land management option chosen for farmland bird conservation can change the ecosystem service delivery in that area. Benayas et al. (2009) found that despite an increase of 44% in biodiversity and 25% in ecosystem services in ecologically restored ecosystems, intact ecosystems still had higher provision for both.

With the advent of increased usability and availability of Geographic Information Systems (GIS) and land cover data sets, a rise in mapping of ecosystem services can be seen (Troy and Wilson, 2006, Balmford et al., 2002). Two approaches can be noted in literature on mapping ecosystem services. The first is broad-scale, and uses values derived from habitat types and mapped to whole areas, a prime example being Costanza et al. (1997) whereby values for ecosystem services were mapped across the planet, estimating a total value of US$16-54 trillion for 17 ecosystem services across the world. Troy and Wilson (2006) undertook case studies using this method in multiple locations in the USA including Massachusetts, California and Maury Island, Washington, finding variability across case study sites. The second approach involves small scale modelling for single ecosystem services using ‘ecological production functions’, which calculate provision using local ecological variables (Nelson et al., 2009). Examples include Kaiser and Roumasset (2002) who investigated the conservation of trees in tropical watersheds, resulting in high payoffs in terms of water saved and the impact on groundwater discharge. Nelson et al. (2009) suggest that what are needed are approaches that build on using the rigour of small-scale approaches but at the larger scope of broad-scale approaches.

Examples of studies that have attempted to use this mixed approach include Boody et al. (2005) who investigated the impact of changes to farming practices in Minnesota, USA, on social, environmental and economic outcomes. It was found that the magnitude of changes was resultant on the magnitude of changes to agricultural practices; with improvements to the environment including improved water quality, carbon sequestration, lower greenhouse gas emissions. A study conducted in South Africa investigating surface water supply, water flow regulation, soil accumulation, soil retention and carbon storage found low levels of spatial congruence between ecosystem services (Egoh et al., 2008). Another study highlighted how carbon sequestration strategies in the USA using tree plantations don't often consider the full environmental impact, finding that plantations can help groundwater recharge but include consequences such as reduced stream flow and acidification of soils (Jackson et al., 2005). Antle and Valdivia (2006) have studied the provision of ecosystem services using a minimum-data approach for agriculture, investigating carbon sequestration in Montana, USA. They found that the minimum-data approach supply curve for carbon storage was similar to that from a more data intensive approach. Others have used spatially explicit models over larger scales.

A diversity of methods now exist to spatially quantify ecosystem services, and the choice of method is dependent on several factors including data availability and the question that needs to be answered (Mouchet et al., 2014). Mouchet et al. (2014) states that the complex socio-ecological should be viewed by characterising three types of interactions: ecosystem service delivery, stakeholders' needs and the benefits stakeholders derive from the ecosystem services that are delivered. In recent years, two different approaches can be seen, the first utilise static snapshots of ecosystem service associations at a certain time and space and the second evaluates ecosystem service associations across time and/or space (Mouchet et al., 2014). The latter approach is seen to be more robust, and follows Raudsepp-Hearne et al. (2010) in identifying ecosystem service bundles, and can be seen in research conducted on trade-offs on ecological relationships between ecosystem services (Egoh et al., 2008) and ecosystem service supply and demand congruence (García-Nieto et al., 2013). Though the ecological trade-offs need to be assessed for a more accurate predictions of demand-supply congruencies (Seppelt et al., 2011). Several methods are currently used for analysing ecosystem service associations, the group of methods belong to the detection and defining of the ecosystem service bundles and the second identifying the drivers of bundles.

Ecosystem service bundles have been detected and defined by several methods, though these can be grouped into association coefficients, ordination and clustering (Mouchet et al., 2014). Association coefficients include the use of correlation coefficients (Chan et al., 2006, Naidoo et al., 2008, Egoh et al., 2009, Egoh et al., 2008, Raudsepp-Hearne et al., 2010, Eigenbrod et al., 2010, Gos and Lavorel, 2012), Chi-squared tests, and overlap analysis (Egoh et al., 2009, Egoh et al., 2008, Swallow et al., 2009, Eigenbrod et al., 2010, O'Farrell et al., 2010). Ordination approaches include use of Multiple Correspondences Analysis (MCA) and Principal Component Analysis (PCA) (Raudsepp-Hearne et al., 2010, Maes et al., 2012). Clustering is also used, including self-organising maps and K means (Raudsepp-Hearne et al., 2010)

Methods for identifying drivers of bundles include the use of distance approaches such as Mantel tests and Analysis of Similarity (ANOSIM) and analysis of variance approaches such as ANOVA (Willemen et al., 2010), MANOVA and co-inertia. Regression based approaches have also been utilised including Generalised Linear Models (Steffan-Dewenter et al., 2007, Fisher et al., 2011), Generalised Additive Models and Autoregressive models. Other techniques include machine-learning (i.e. decision trees and Artificial Neural Networks), time-series models (such as VAR and ARMA) and canonical analysis (such as Canonical Correspondence Analysis used by Lamarque et al. (2014)).

A spatially explicit modelling tool called 'Integrated Valuation of Ecosystem Services and Tradeoffs' (InVEST) has been used by Nelson et al. (2009) in the Willamette Basin, Oregon, USA, to investigate several ecosystem services, with the benefit of both approaches: the small-scale detail based on ecological production functions and the broad scale mapping function. Nelson et al. (2009) looked at multiple ecosystem services including water quality, carbon sequestration, biodiversity conservation and agricultural/timber harvest. Spatial patterns of synergies and trade-offs between ecosystem services and biodiversity conservation were investigated. It was found that those scenarios with multiple high value ecosystem services had high biodiversity scores.

Other studies have been conducted looking at the spatial congruence of ecosystem services. Various studies have investigated the relationship between ecosystem services and biodiversity. Thomas et al. (2013) investigated conservations strategies in the New World and Britain, finding that conservation of carbon stocks does not necessarily lead to biodiversity conservation, but this varies by region. Though it was found that by combining carbon and species within the conservation planning process, substantial increases in biodiversity could be gained through modest reductions in carbons stocks being conserved. A study conducted in California, USA, found that biodiversity and ecosystem services had low correlation, with moderate overlap (Chan et al., 2006). Another study in South Africa investigated whether biodiversity priorities and hotspots overlapped with ecosystem services (Egoh et al., 2009). A moderate overlap was found with species richness higher in areas of water flow regulation and soil accumulation. Turner et al. (2007) , in a study looking at the relationship between biodiversity priorities and ecosystem services on a global scale, found that overlap between the two varied across regions, though pinpointing overlaps showed opportunities to maximise both biodiversity and ecosystem service provision. A global study found that areas selected to maximise biodiversity had the same ecosystem service provision as randomly selected areas, with spatial congruence between ecosystem services varying widely between regions (Naidoo et al., 2008). Anderson et al. (2009) showed similar results using Britain as a case study investigating spatial covariance between biodiversity and ecosystem services, with 'diametrically opposing conclusions' being drawn from different regions of Britain.

Both biodiversity and ecosystem services are seeing a decline, and it is thus highly important that that both opportunities and trade-offs are considered to maximise conservation of both (Cimon-Morin et al., 2013). The spatial congruence between ecosystem services and biodiversity has been studied for over a decade, with a positive relationship between biodiversity and ecosystem service hotspots being identified globally (despite often being weak to moderate) (Larsen et al., 2012, Luck et al., 2009, Strassburg et al., 2010), with any investment in preserving ecosystems services would probably also preserve biodiversity (Naidoo et al., 2008). Conversely, weak overlap between ecosystem services and biodiversity has been seen when looking at individual regions (e.g. polar and temperate regions) (Larsen et al., 2012, Strassburg et al., 2010). Though it is at the local scale whereby ecosystem services provide benefits, hence this is more appropriate that the global scale. An example can be seen with carbon storage, where global analysis reveals a strong positive correlation, though finer resolution analysis shows that biodiversity does not coincide with carbon rich habitats in tropical zones (Paoli et al., 2010, Anderson et al., 2009). Hence the importance of looking at the local scale, where trade-offs can be identified is vitally important (Naidoo et al., 2008).

The reason for the positive global correlation of ecosystem services and biodiversity has been attributed to several factors, including the actual data used in the analysis (Eigenbrod et al., 2009, Eigenbrod et al., 2010), using proxy data linked to land cover/ land use, where primary data is lacking (Haines-Young et al., 2012, Maes et al., 2012, Martínez-Harms and Balvanera, 2012, Yapp et al., 2010). Though proxy data is often obtained through the 'benefit-transfer method' (Eigenbrod et al., 2010) whereby values are derived from a small area and applied to a larger area, or altogether different region (Chan et al., 2006, Balmford et al., 2002, Sutton and Costanza, 2002), though it has been shown when compared to primary data this proxy approach can provide false spatial correlations, with multiple ecosystem services analysed can compound the biases (Eigenbrod et al., 2010). Despite the data used, the actual categories of ecosystem service that are analysed can have an effect of the correlation on biodiversity. Provisioning ecosystem services have been found to weakly or negatively correlate with biodiversity (as well as other ecosystem services), whereas regulating, supporting and cultural services show a greater spatial correlation with biodiversity (Cimon-Morin et al., 2013). Though most studies have only considered five or fewer ecosystem services (Seppelt et al., 2011), with those that are easily mapped, or those that have a direct role in decision making being analysed (Cimon-Morin et al., 2013). Though important ecosystem services relevant to ecosystem regulation and welfare are seldom included in these analyses (Martínez-Harms and Balvanera, 2012, Cimon-Morin et al., 2013). Finally, another reason for the weak congruence between ecosystem services and biodiversity is due to functional diversity is the most important biodiversity aspect to influence the production of a given ecosystem service at a specific location, and there may be a spatial inconsistency between functional diversity and species richness, or rather the link between these lacks consistency across various spatial scale, and trophic levels (Cadotte et al., 2011). Cimon-Morin et al. (2013) suggests that due the general lack of congruence seen in literature between ecosystem services and biodiversity, conservation choices should not hinge on priority areas for either, with the assumption the other would also benefit. Though the inclusion of ecosystem service assessment still has advantages in being included in reserve selection, rather than being ignored.

Eigenbrod et al. (2009) provided one of the first analyses of biodiversity and ecosystem service provision (carbon storage, agriculture and recreation) across multiple conservation strategies in England. They found that protected areas had high biodiversity (more than three-fold over expected) and carbon storage provision, with recreation scoring lowly. A study on ecosystem service delivery across multiple protected sites across the UK found that protected sites had higher levels of ecosystem service; especially with more cultural and regulating ecosystem services (Eastwood et al., 2016). Land cover classes have been investigated in terms of ecosystems service demand in the central region of Leipzig-Halle, Germany by Burkhard et al. (2012), finding that urban areas are sinks and rural areas showed oversupply. They also found that those habitat classes important for conservation such as natural grassland showed high supply of regulating services compared to demand, though low supply of provisioning services compared to demand. Castro et al. (2015) have investigated the supply of ecosystem services in Natura-2000 and RENPA protected areas in Andalucía, Spain. They found that 59% of regulating ecosystem services supply (erosion control, water flow maintenance and climate regulation) occurred within protected areas compared to non-protected areas. Castro et al. (2015) also found that a 14% increase in protected areas (by land area) would secure 83% of the total regulating ecosystem services.

The status of England's ecological network is dependent on a wide selection of statutory and non-statutory wildlife sites across a range of habitats. The Sites of Special Scientific Interest (SSSI) series of sites alone do not make a resilient ecological network. Lawton et al. (2010) found that across England, the size of wildlife sites was too small, with some habitats having reached a state of loss, whereby significant efforts would be required to prevent further biodiversity loss; the loss or degradation of natural connections in the countryside was also seen, resulting in isolation of sites. Under management and a lack of protection was noted across most of the semi-natural habitats (the exception being Natura 2000 and SSSIs sites) (Lawton et al., 2010). SSSI's are protected sites and have been designated as such due to being valuable and distinctive for habitat, wildlife, geology or geomorphology (National Audit Office, 2008, Natural England, 2012). The SSSI series of sites are designed to be the most valuable sites in the UK, alongside being 'resilient' to the effects of climate change (Natural England, 2012). Special Protection Areas (SPAs) and Special Areas of Conservation (SACs) form the Natura-2000 network, and are also designated as SSSI by Natural England (2012).

The New Forest National Park was designated as a National Park in 2005, and is the UK's second youngest (National Parks UK, 2016). Within the National Park, the New Forest SSSI includes lowland heath, mire and ancient pasture woodland (Natural England, 2016b). The National Parks and Access to the Countryside Act (1949) created the 'National Park' designation to conserve areas for wildlife, cultural heritage and for its natural beauty whilst allowing the public to enjoy those areas and often include villages and towns. This makes the New Forest ideal as a case study to examine the ecosystem service provision within the SSSI and wider National Park. With conservation and protection areas already securing protection status, it is important to understand the role these areas play in protecting wider ecological and socio-cultural benefits to society in terms of ecosystem services.

To the author's knowledge, no studies have yet compared ecosystem service provision, including biodiversity, inside and outside SSSI protected areas, those with a lower level of protection in terms of human activities allowed, within the New Forest National Park, in respect to their management goals. This work builds on previous work (Anderson et al. (2009), Eigenbrod et al. (2009), Eigenbrod et al. (2010)) whom investigated three ecosystem services with biodiversity and stated that data availability on ecosystem services was a major constraint. The aim of this chapter is to consider the role of the New Forest National Park SSSI and investigate the provision of ecosystems services in and outside the SSSI area, compared to the rest of the National Park in light of current management priorities. Various methods will be utilised, including the use of the model InVEST to address the need of utilising ecological production functions and working at a broad mapping scale. Eastwood et al. (2016) states there are several issues with large scale correlative analyses. Studies investigating biodiversity conservation and ecosystem services are usually limited in their scope of services assessed due to data availability, with limited services leading to a distorted impression of the overall ecosystem service provision. The ecosystem services were explicitly mapped to the New Forest National Park Partnership Plan 'priorities' (New Forest Park Authority, 2015):

- Tranquillity is actively managed for by the NFPA, with priorities 'ST1 and ST1' involved with monitoring overall tranquillity and 'ST3 – ST5' focusing on reducing visual intrusion. The New Forest Park Authority (2015) want to maintain and enhance tranquil areas of the National Park, particularly focussing on reducing low-flying aircraft disturbance and the visual impact of energy infrastructure such as power cables and pylons
- Priority 'LH1' states the need to improve the condition of at least 30 local nature conservation sites, with priority 'LH2' aiming to have 60% of SSSIs within the New Forest National Park as favourable condition by 2020. Hence separating and comparing the SSSI area is important in this study.

- Priority ‘LH7’ states the need to promote an integrated approach to river catchment management, including surface and ground water quality and flood prevention, with priority LH8 stating the need to improve water quality.
- Priority ‘LH14’ directly looks to establish methods for monitoring wildlife and habitats in line with the Government’s Biodiversity 2020 targets. Priority ‘LM3’ states the promotion of ecosystem service maps to aid land management and social wellbeing and the economy. Priority ‘CC3’ aims to maintain the carbon sink the New Forest provides.
- Priority ‘CC4’ focusses on producing and promoting a climate change adaption plan, through the likely future climate impacts. Storm and flood events have affected local communities in the New Forest, and hence priorities ‘CC5’ and ‘CC6’ aim to supporting preparation of Emergency Response Plans to plan for flooding and management to reduce flooding.
- Recreation usage is important, with the section ‘Enjoying the special qualities of the National Park’ (priorities ‘EP1 – EP8’) aiming to protecting and improve community involvement in access, open spaces, routes and opportunities for outdoor recreation. Livestock was also included, due to the traditional appeal of visitors to the New Forest ponies, horses and cattle.
- Economic well-being is actively managed, with priority ‘EW4’ aiming to encourage private sector investment in forestry and land management.

4.3 Materials and methods

Study site

The New Forest National Park is England's smallest national park covering 145 square miles in the south east (50°51'59" latitude and 01°40'50" longitude) across the counties of Hampshire and Wiltshire. Originally a royal hunting reserve, this mixed landscape is now revered for its ancient history, wildlife and beauty (Forestry Commission, 2016). The National Park has various protected designations including sites protected by European and UK law such as Special Protection Areas (SPA), Special Areas of Conservation (SAC), Ramsar sites and Sites of Special Scientific Interest (SSSI) (JNCC, 2006). The latter underpin all other designations and so I considered these to be protected areas (PAs).

The Crown lands encompass 26,756 ha of the New Forest, with the SAC comprising 29,214 ha, with the mosaic landscape comprising of grassland, mires, heathland, ancient and ornamental woodland, forestry inclosures and agricultural land (Forestry Commission, 2008). The New Forest is especially important as these habitats, once common in lowland Western Europe, are now rare and fragmented. In fact there are thirteen habitats of European interest with two priority habitats: bog woodland and riverine woodland (Forestry Commission, 2008). The New Forest has some of the most extensive lowland northern Atlantic wet heaths in southern England, is the largest lowland heathland in the entirety of the UK and holds the largest area of mature beech woodland in Britain (JNCC, 2016).

The New Forest is used regularly by visitors for recreation, with 235 km of public footpaths, 57 km of public bridleways, and having 27,000 hectares of land with open public access (NFPA, 2007). Sharp et al. (2008) found that 40% of visitors were staying tourists with the remainder comprising of 35% local and 25% non-local (from over 5 miles away) day trippers with an estimated total of 13 million visitors annually (NFPA, 2007).

Data

Various data were needed to produce the ecosystem services and biodiversity maps. The hydrological models; InVEST Water Yield and InVEST Nutrient Retention: Water Purification model, were the most resource intensive, and used watershed boundaries that did not extend to the full area of the New Forest. These have been separated here from the terrestrial models for ease of understanding (see Figure 4.3.1 for a flowchart of modelling and mapping operations). Due to the local scale, and the research question being concerned with the management goals of the New Forest Park Authority (2015), techniques utilised were chosen to provide a robust estimate of the ecosystem services analysed, using the best local data that I had available to me. For this reason, data from Mapping and Assessment of Ecosystems and their Services (Maes et al., 2017) and other European Union projects was not utilised. In light of the newer CICES classification (Haines-Young et al., 2012), ecosystem services were also chosen to fall into the three main themes of provisioning, regulating and maintenance, and cultural and social.

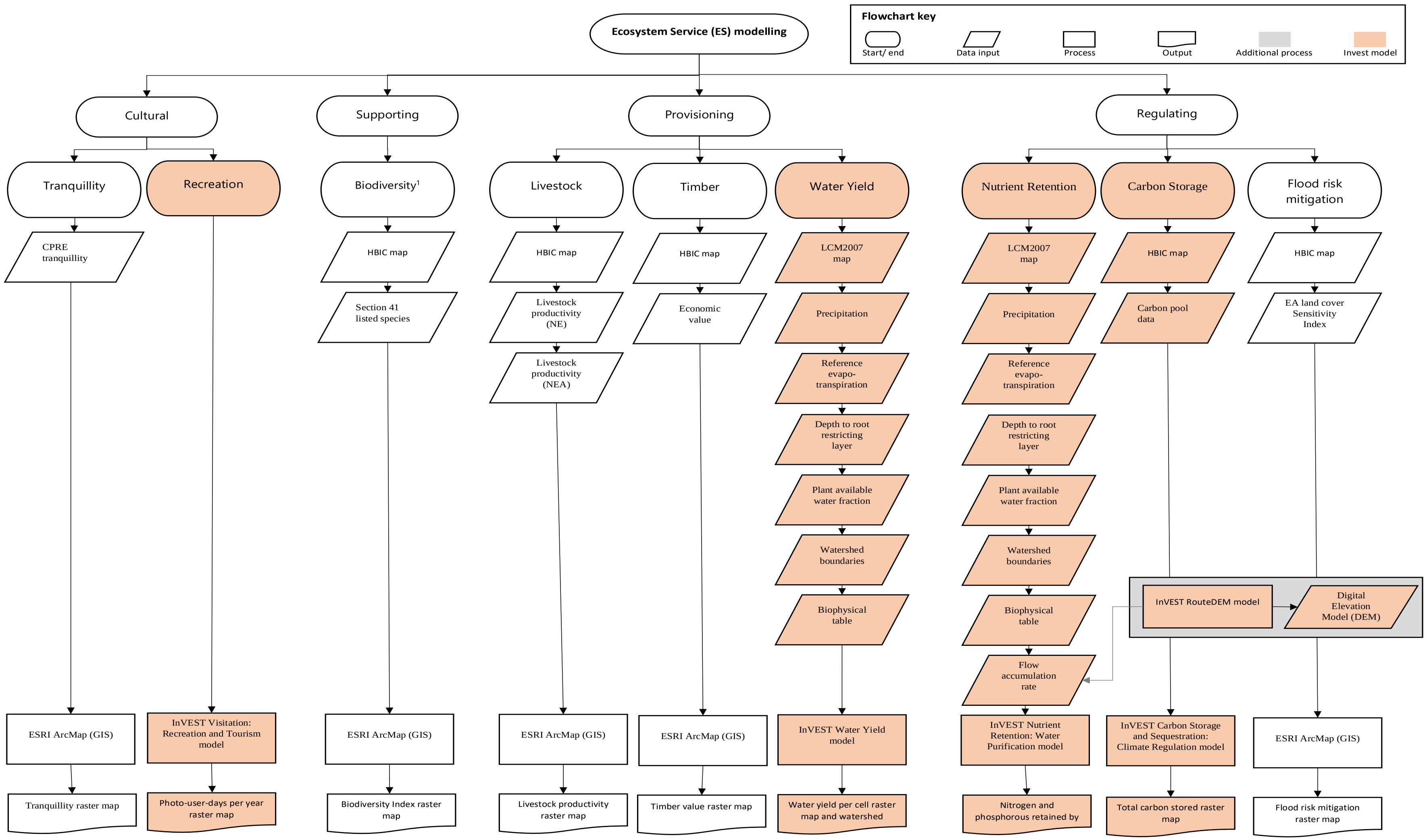

Figure 4.3.1: Flow chart illustrating the steps of modelling the ecosystem services; recreation, agricultural land quality, property value, livestock, timber, water yield, nutrient retention, carbon storage, flood risk mitigation and pollinator habitats in the New Forest National Park. InVEST models paths are shaded. Boundary data is not included as a data step, as it is common to all maps. [1]Biodiversity has been included as a supporting ecosystem service to simplify the mapping of the modelling task.

Hydrological modelling

The Academic UKCP09 (Met Office, 2016) gridded raster datasets were used as average annual precipitation in mm, though a long-term average total rainfall data was not available, hence 5 km resolution monthly average precipitation (1981-2010) was downloaded and processed. Data was imported as ASCII using the 'ASCII to Raster' conversion tool into ArcMap as float raster's, rather than integers. As each raster was the average rainfall in a specific month (the mean from 1981 to 2010), raster calculator was used to calculate the mean of all twelve monthly rasters to give a average annual rainfall between 1981-2010. Several boundary pixels were missing for the coast of the study area, hence the 'Focal statistics' tool was used to fill these in, using a rectangular neighbourhood 3 x 3 cell mean technique. The 'Mosaic to new Raster' tool was used to combine the mean precipitation raster with the new focal statistics layer to create a new raster, with the new perimeter pixels and all original pixel values remaining unaltered. Parameters for the new raster were set to match the original data: the spatial reference was set to British National Grid, a 32 bit float pixel type, 1 band, with a cell size of 5000 m, with the annual precipitation raster being used as the mosaic operator.

Reference evapotranspiration was required as a raster (as average annual value in mm). The CGIAR-CSI Global Potential Evapo-Transpiration (Global-PET) Geospatial Dataset (Trabucco and Zomer, 2009) was downloaded, re-projected from WGS_1984 to British National Grid using the 'Project raster' tool, with a cell size of 1 km x 1 km using the OSGB_1936_To_WGS_1984_7 geographic transformation and clipped for the New Forest National Park using the 'Extract by Mask' tool.

A depth to root restricting layer was downloaded from the European Soil Database (ESDB), as derived data for 'Depth available to roots' in cm (Hiederer, 2013a, Hiederer, 2013b, Panagos et al., 2012). The data was re-projected from ETRA_1989_LAEA to British National Grid using the 'Project raster' data management tool with a cell size of 1 km x 1 km using the 'ETRS_1989_To_WGS_1984 + OSGB_1936_To_WGS_1984_7' geographic transformation using the 'nearest' resampling technique. This was then reclassified using the 'Reclassify' spatial analyst tool to convert the cm measurements into mm, as the data needs of the model dictated, before being clipped by using the 'Extract by Mask' tool for the study area.

Plant available water content (PAWC) was needed as a raster as fraction values between 0 and 1. The fraction can be obtained through a division of the available water content (in mm) divided by soil depth (Sharp et al., 2015). Available water content was averaged from top soil and sub soil available water content (mm) 1 km x 1 km rasters from European Soil Database Derived data (Hiederer, 2013a, Hiederer, 2013b, Panagos et al., 2012). The data format was in the 'RST' format that was read in ArcMap, the projection defined as 'European Terrestrial Reference System 89, Lambert Azimuthal Equal Area (ETRS_89_LAEA)' using the 'Define projection' data management tool and was exported as a GRID format. The data was re-projected from ETRA_1989_LAEA to British National Grid using the 'Project raster' data management tool with a cell size of 1 km x 1 km using the 'ETRS_1989_To_WGS_1984' geographic transformation using the 'nearest' resampling technique. Raster calculator was used to calculate mean available water content across the top soil and sub soil rasters. This was divided by the root restricting layer to give a Plant Available Water Content (PAWC) fraction raster with all values between 0 and 1.

Watersheds data was obtained from the Environment Agency (2015) as the Water Framework Directive River Waterbody Catchments Cycle 2 version 2.0, as a polygon dataset, which includes 12 coastal catchments, allowing the full extent of the New Forest to be modelled. To prevent misrepresentation as the InVEST water yield model calculates at the watershed level, the watersheds that fully or partially intersected the study area boundary were selected using the 'Select by Location' tool and exported as a new data set, with a 'ws_id' column added to the attribute table with unique values, as dictated by the model. Land use/land cover (LULC) was obtained from the CEH Land Cover Map (LCM) 2007 (Morton et al., 2011), since this dataset is available for the whole country and hence covers the extent of watersheds outside the New Forest National Park. The Hampshire Biodiversity and Information Centre (HBIC) habitat map is not used as its extent is limited to the New Forest National Park boundary. The HBIC map is used for all other non-hydrological modelling and mapping due to its greater accuracy within the boundaries of the New Forest National Park. The biophysical table with evapotranspiration, rooting depth coefficients, phosphorous and nitrogen by habitat type were sourced by personal communication with the Centre for Ecology and Hydrology (J Redhead, 2016, pers. comm., 6 June).

The OS Terrain 50 m Digital Terrain Model (DTM) (Ordnance Survey (GB), 2015) was used due to free access and a cell size that allows practical processing. The 'Fill' spatial analyst tool was then used to remove any holes that would prevent the model working correctly. The InVEST RouteDEM v 3.2.0 (Natural Capital Project, 2015b) tool was then run with the updated DEM with multiple levels of threshold flow accumulation between 10 and 2500, a visual appraisal found that the 1350 threshold flow accumulation level being the most similar to the OS Mastermap in terms of waterways (OS Open Data, 2016).

The seasonality constant (*Z*) was calculated using the formula $Z = n * 0.20$ (as per Redhead et al. (2016)), where *n* was the historical daily rainfall event (> 1 mm) data for the South East England administrative area by summing the average monthly (between 1971-2000) from Met Office (2013) values. This gave a value of 23.958 for the whole of the South East of England, of which the New Forest is part.

To ascertain the water yield in the catchments in the New Forest, the InVEST (v.3.2.0) Water Yield model was used. The precipitation, reference evapotranspiration, depth to root restricting layer, plant available water fraction and land use raster layers, and a watersheds vector were inputs to this model. The seasonality constant and the biophysical table of evaporation coefficients and root depths were also included.

The InVEST Water Yield model outputted a shapefile that was opened in ArcMap v10.3 (ESRI UK Limited, Aylesbury) and converted to raster for mean water yield per pixel on the watershed (mm) and the volume of water yield in the watershed (m^3) using the 'Polygon to raster' tool, using a cell size of 25 m x 25 m using the 'maximum area combined' method and only processed to the extent of the New Forest (as some watersheds are not fully contained within the New Forest boundary). It must be noted that the model does not take into account upstream catchments that contribute their yield to those downstream. These implications are considered later in the discussion. The mean water yield per pixel on the watershed (mm) was used in the final assessment of ecosystem services, as it gives a mean value per pixel within the watershed (which is based on the land cover input map which was at 25 m x 25 m).

To calculate the retention of nitrogen and phosphorous across the New Forest area, InVEST (v.3.2.0) Nutrient Retention: Water Purification model was used. This model had the same data needs as the Water Yield model, with the addition of a biophysical table, with water quality coefficients including nitrogen and phosphorous loading, and vegetation filtering values for each pixel. Additionally, the Nutrient Retention model required the Flow Accumulation rate from InVEST (v.3.2.0) RouteDEM.

Whereas the 'per pixel' outputs cannot be used from the Water Yield model, this is not the case with the Nutrient Retention model. Both 'per pixel' and 'per watershed' outputs were generated, the former chosen due to the final ecosystem service provision assessment not being based on watersheds. The outputs used were the pixel level map showing how much nitrogen or phosphorous was absorbed by the habitat on the map. Maps were resampled to 25 m x 25 m resolution from 50 m x 50 m (due to input resolution of the Digital Elevation Model) using the 'Resample' tool using the 'nearest' technique and clipped to the extent of the New Forest using the 'Extract by mask' tool.

Terrestrial mapping and modelling

Carbon

The InVEST Carbon Storage and Sequestration: Climate Regulation (Natural Capital Project, 2015a) model was used to map carbon in the New Forest. The HBIC map was used as the landcover map, with a unique numerical land use code being added for each habitat type to the attribute table. Carbon pools data were extracted from Jiang et al. (2013) and Redhead et al. (2016), and median taken for each landcover type for above ground stored carbon in biomass (Mg ha-1), below ground stored carbon in biomass (Mg ha-1), carbon stored in soil and carbon stored in dead organic matter in Mg ha^{-1}. The values were especially relevant, as the data is for nearby county of Dorset, England, adjacent to the New Forest National Park (Table 4.3.1). Land use codes were added to a column in the carbon pool data to allow the model to cross-reference this with the landcover map. The output file was the same resolution as the input landcover map, thus a 25 m by 25 m cell size raster was generated. Raster calculator was used to normalise the map, dividing the cells by the highest value of 25.16.

Table 4.3.1 Pooled carbon pool data extracted from Jiang et al. (2013) for the adjacent county of Dorset. [1] Values are median values or each habitat type, [2] No value available, thus value for built-up areas and gardens used, [3]Value for Inland bare ground and rock used [4]Value for littoral sediment used [5]Values available for above and below ground carbon used from stream orders 5-7, and zero values assumed for soil and dead materials [6]Unidentified habitats and waters were given zero values.

Habitat	Carbon pools (Mg ha^{-1})[1]			
	Above ground	Below ground	Soil	Dead materials
Acid grassland	6	6	85.87	2
Arable and horticulture	3.01	2	50.7	0
Boundary and linear features[2]	0	0	25	0
Bracken	0	0	86.835	0
Broadleaved, mixed, and yew woodland	116	120	111.5	55
Built-up areas and gardens	0	0	25	0
Calcareous grassland	0	6	20	2
Coniferous woodland	94.75	110	95.5	50
Dwarf shrub heath	7	5.5	64.99	1.5
Fen, marsh and swamp	10	5	83.53	0
Improved grassland	3	2.5	21.5	0.5
Inland rock[3]	0	0	0	0
Inshore sublittoral sediment[4]	5.61	0	0	0
Littoral Rock	0	0	0	0
Littoral Sediment	5.61	0	0	0
Neutral grassland	3	4	70.3	1
Rivers and streams[5]	0	0	0	0
Standing open water and canals	4.64	0	115	0
Supralittoral Rock	0	0	0	0
Supralittoral Sediment	0	0	0	0
Unidentified habitat[6]	0	0	0	0
Unidentified water[6]	0	0	0	0

Recreation

Recreation was modelled using the InVEST Visitation: Recreation and Tourism model (Natural Capital Project, 2015c). Due to constraints within the model, the minimum cell size that could be set was 250m. The model provided various outputs including a shapefile with the average 'photo user days' per year. This data originates from the photo-sharing website 'flickr.com' (Yahoo, 2016), where most images have a specific longitude and latitude data attached. The InVEST Recreation model uses this data combined with the photographer's ID and the date of the image to calculate average photo user days for the period 2005 – 2014 for each cell (Natural Capital Project, 2015c) within the National Park boundary. Wood et al. (2013) used photographs from Flickr to estimate visitation rates globally across 836 sites, and found the Flickr derived estimates to be reliable when compared to empirical data from each site. As 40% of images on Flickr are European, with a large proportion being from the United Kingdom (Wood et al., 2013), this model provides a robust proxy for empirical visitation rates in the New Forest . The output from the model was a gridded shapefile of 250 m by 205 m cells, and was converted to raster and resampled using the 'Polygon to raster' tool.

Livestock productivity

The method utilised to analyse livestock productivity was adapted from Newton et al. (2012). There are two types of grassland in the UK; semi-natural and improved grassland, with the stocking rate of the latter being three times higher in English lowland grasslands (Bullock, 2011). This gave the initial index values of 1 for improved grassland and 0.33 for semi-natural grassland. Whilst grazing on semi-natural grassland still occurs, mostly for conservation management, the production is low due to low levels of phosphorous and nitrogen (Janssens et al., 1998). Semi-natural grassland is defined as encompassing five broad types: calcareous, neutral, acidic, fen meadows and Calaminarian (Crofts and Jefferson, 1999), providing direct matches with the HBIC BAP habitats of improved, neutral, acid, calcareous. Calaminarian grassland and fen meadows are not present within HBIC map, the latter only be included within 'fens, marsh and swamp' and hence are not included. Figures from (Crofts and Jefferson, 1999) were used to calculate more accurate figures for calcareous and acidic grassland using annual cattle stocking level guidelines. With a suggested rate of 0.5 cattle for 52 weeks on calcareous and neutral grassland, and 0.4 on acidic grassland, this 20% difference in figures was translated to productivity by reducing the index value of 0.33 by 20% giving a new value of 0.26 for acidic grassland (see Table 4.3.2). Only stocking levels for cattle were considered, as turning out sheep in the New Forest National park is uncommon due to lack of 'common of pasture for sheep' rights only belonging to a few properties, with only one commoner currently turning out sheep near Bramshaw (New Forest Park Authority, 2016). Figures for year round grazing were used, as commoner's rights allow this. Fen, marsh and swamp and dwarf shrub heath were not included, due to the uncertainty of the areas and extent of livestock grazing in these habitats. Values were joined to the attribute table of the HBIC map in ArcMap and the 'Look up' tool used to create a 25 m by 25 m raster.

Table 4.3.2: Livestock productivity values.

Habitat class	Livestock productivity index
Improved grassland	1.00
Neutral grassland	0.33
Acidic grassland	0.26
Calcareous grassland	0.33

Timber

The New Forest National Park has a long history of timber production beginning in the 1700's, (Forestry Commission, 2008). Instead of mapping the 8,500 ha of land that is currently designated as inclosures for timber production (Forestry Commission, 2008), broadleaved, mixed and coniferous woodland was mapped as being more indicative of timber ecosystem service value, as the inclosures are subject to change. The method utilised to analyse timber was adapted from Newton et al. (2012). To map the value of timber in the New Forest the mean price per m^{-3} of broadleaved timber (£9.50) and conifer timber (£11.50) (Newton et al., 2012) was used to create an index of 0.826 and 1.0 respectively, with all other habitat types getting a value of 0. Values were joined to the attribute table of the HBIC map in ArcMap and the 'Look up' tool used to create a 25 m by 25 m rasters.

Flood risk mitigation

The capacity of vegetation to mitigate flood risk was assessed utilising a scoring approach using a non-monetary valuing technique adapted from Newton et al. (2012). A comprehensive hydrological modelling approach would have provided the most comprehensive data on flood risk in Dorset but was unfortunately outside the scope of this study. Following Newton et al. (2012), the Environment Agency (2008) 'Sensitivity classification' values were allocated to LCM habitat classifications, with a score of 1 indicating very low likelihood of flooding, to a score of 4 indicating a high likelihood. The index is based on multiple factors including rain and slope variables that could cause rapid runoff (Newton et al., 2012). This method assumes that flood risk would be a direct consequence of land cover. As the index values represent capacity for run-off generation, they were inverted (stepping down an integer, so 1 become 0, etc) and normalised to create an index for water retention capacity (Table 4.3.3).

Table 4.3.3: Flood risk mitigation values. Habitat classifications were matched from LCM2000 (used by the Environment Agency (2008))as follows: [1]cereals and horticulture, [2]managed neutral grassland and set-aside grassland, [3]water (inland), [4]inland bare ground, [5]littoral sediment, [6] the lowest score was given and [7]Continous urban and suburban/ rural developed.

HBIC habitat	Land cover sensitivity class	Flood risk mitigation	Normalised Flood risk mitigation index
Bracken	1	3.00	1.00
Arable and horticulture[1]	4	0.00	0.00
Improved grassland	3	1.00	0.33
Neutral grassland[2]	1	3.00	1.00
Acid grassland	1	3.00	1.00
Standing open water and canals[3]	1	3.00	1.00
Fen, marsh and swamp	1	3.00	1.00
Rivers and streams[3]	1	3.00	1.00
Unidentified water[3]	1	3.00	1.00
Calcareous grassland	1	3.00	1.00
Littoral Sediment	1	3.00	1.00
Dwarf shrub heath	2	2.00	0.67
Inland rock[4]	1	3.00	1.00
Supralittoral Sediment	1	3.00	1.00
Inshore sublittoral sediment[5]	1	3.00	1.00
Littoral Rock	1	3.00	1.00
Broadleaved, mixed, and yew woodland	1	3.00	1.00
Coniferous woodland	1	3.00	1.00
Unidentified habitat[6]	1	3.00	1.00
Boundary and linear features[7]	1	3.00	1.00
Built-up areas and gardens[7]	1	3.00	1.00
Supralittoral Rock	1	3.00	1.00

Tranquillity

The Campaign to Protect Rural England (Campaign for the Protection of Rural England (CPRE), 2007, Jackson et al., 2008) National Tranquillity data (license no. 900018881) was used to as a direct indicator for the sense of tranquillity people feel at a given location. The data was in 500 m x 500 m raster resolution at a relative scale for the whole of the UK, with 33 positive and negative attributes (Table 4.3.4), with a composite raster of all attributes (with positive attributes weighted positively and negative negatively) for tranquillity. This composite layer ranged in values from -140.52 to 148.70. This raster was resampled to a 25 m x 25 m resolution (using the 'Resample' tool using the 'nearest' technique), clipped to the extent of the New Forest boundary using the 'Extract by mask' tool and normalised to values between 0 and 1.

Table 4.3.4: Positive and negative attributes that are in the Campaign for the Protection of Rural England (CPRE) (2007) National Tranquillity 2008 weighted composite data.

Positive attributes	**Negative attributes**
Seeing:	Seeing:
trees in the landscape	wind turbines
deciduous trees in the landscape	power lines
natural looking woodland	roads
streams and rivers	trains and railways
lakes	towns and cities
the sea	coniferous woodland
the stars at night	high altitude aircraft
wide open spaces	low flying aircraft
a natural landscape	overhead pollution (night time)
a wild landscape	lots of people
remote landscapes	urban development
	any signs of human impact
	anyone at all
Hearing:	Hearing:
running water	occasional noise from road traffic
lapping water	constant noise from road traffic
the sea	trains and railways
low noise areas	high altitude aircraft
	non-natural sounds

Biodiversity

Biodiversity was included in this analysis due to its key role as often underpinning other ecosystem services and being multi-layered in that it can be a regulator of ecosystem processes such as pollination, a final ecosystem services such as for pharmaceutical purposes or as a good such as acting as a flagship for conservation (i.e. large mammals) (Mace et al, 2012). NERC Section 41 listed species data within the New Forest National Park was acquired from Hampshire Biodiversity Information Centre (Supply Agreement 6335). There were 52,395 records supplied, including higher plants, mammals, reptiles and amphibians, mammals and birds at varying resolutions of 1 m^2, 10 m^2, 100 m^2, 1 km^2, 2 km^2 and 10 km^2. Records were filtered to only 2005-2015 (41,917 records) and non-presence records, where the status of the species was not-recorded, were removed by filtering out those species whereby quantity was recorded as zero (using the expression "NOT("Quantity" ='0')" in the 'Select by Attributes' tool). This left a total of 38,203 records. These records were then split into resolution categories with 46 records at 1m^2, 1,422 at 10 m^2, 16,297 at 100 m^2, 20,369 at 1 km^2, 8= at 2 km^2 and 61 at 10 km^2 by calculating geometry then extracting the data into new shapefiles. The coarser resolution 10 km^2 records were not included in this analysis, these were mainly for Surasian stone-curlew *Burhinus oedicnemus* and Roseate Tern and resolution was coarse to prevent their nests (*Sterna dougallii),* being discoverable.

At 10m^2 resolution, 1116 were amphibians and reptiles, 5 birds, 92 higher plants, 201 invertebrates (lepidoptera), 8 mammals (terrestrial non-bat). The resolution of the plants was not changed, as the distribution of the species will be fixed to the survey polygon, due to their fixed nature.

All records other records at 10m^2 were transformed to 100m^2 because of the mobile nature of the taxa by filtering out the plant species (leaving 1,330 records), then the 'Feature to point' tool used, to create points in the centre of the polygons. X and Y coordinates were calculated using the 'Calculate geometry' tool and the attribute data edited in excel so that coordinates were made coarser by grid. This was achieved by changing the values to 100m resolution by using the lower left hand corner of larger polygon, and re-centering the point. For example, if a points coordinates were x = 416935 and y = 113485, the lower left hand corner of the 100 x 100 grid would be x = 416900 and y = 113400, and to re-center 50 units would be added creating the final coordinates of = 416950 and y = 113450. These coordinates were displayed as X Y data in ArcMap and 50m round buffer polygons created around each point, before the 'Feature envelope to polygon' tool was used to create a square 100 m x 100 m polygon around the buffered points. This new 100 m resolution shapefile was then merged with the original 100 m resolution file using the ‘Append’ tool resulting in a shapefile with 17,627 records.

Records at $1m^2$ resolution were all lower plants (liverworts, hornworts and mosses), though the resolution was scaled up to $10m^2$ using the same method as above, apart from scaling to 10 m and not 100 m, as surveyors would have used hand held GPS units that can be inaccurate at fixing a location. This new 10 m^2 shape-file was then merged with the original 10 m^2 file using the ‘Append’ tool resulting in a shape-file with 138 records.

To count the number of different NERC section 41 species for a species richness indicator, as opposed to individuals, the ‘dissolve’ tool was used to count the number of species in each species polygon area, resulting in shape-files with overlapping records merged to give a value with the number of unique species in that polygon. The ArcMap additional ‘Count Overlapping Polygons’ tool was used to count the unique species in each overlapping polygon (Honeycutt, 2012). These shape-files were then converted in to raster shape-files individually with the ‘Polygon to raster’ tool using the total unique species value, with the cell size kept at the original resolution and the ‘Maximum combined area’ technique used. The resulting rasters for 10 m, 100 m and 1km were then made comparable and combinable, by dividing by the factor of the resolution (hence the 100 m x 100 m resolution raster unique species value was divided by 10,000) and the resolution changed to 25 m x 25 m using the ‘Resample’ tool using the ‘nearest ‘ resampling technique. The ‘Cell statistics’ tool was used to sum the 10 m, 100 m and 1km rasters, with the output raster having any remaining ‘no data’ values changed using the ‘Is null’ tool to identify these cells, and the ‘Con’ tool to change these pixels to a value of ‘0’.

Protected Areas analysis

As the common extent of all the ecosystem service modelled were the hydrological models, (dictated by watershed boundaries), all ecosystem service maps were clipped with a shape-file of the watershed boundaries, that had been clipped by the New Forest National Park boundary. All watersheds were aggregated using the ‘Dissolve’ tool. The impact of this can be seen as reducing the coastal habitats in the analysis.

All ecosystem service maps and biodiversity were clipped this new watershed guided New Forest boundary shape-file and normalised between 0 and 1 using a unity-based normalization using the following formula, Z = normalised value, and X = existing value:

$$Z = \frac{X - \min(X)}{\max(X) - \min(X)}$$

This was coded in python script in 'raster calculator' as:

("ES_RASTER" – [MIN_VALUE]) / ([MAX_VALUE – MIN_VALUE])

All normalised ecosystem service maps were summed together using the 'Cell statistics' tool to generate a map that showed the total provision of the mapped ecosystem services. The SSSI GIS boundary dataset was downloaded from Natural England (2016a) and clipped to the extent of the New Forest National Park and then the dissolved watershed shape-file and all polygons dissolved. The 'Erase' tool was then used to create a non-SSSI shape-file of the New Forest.

For point sampling, a 25 m grid was draped over the entirety of the SSSI shape-file and label points created using the 'Fishnet' tool and clipped using the 'Clip' tool to the extent of the SSSI and non-SSSI land to give two separate shape-files of 494,987 and 397,842 points respectively. This allowed SSSI and non-SSSI points to be extracted for each of the normalised ecosystem services, as well as the combined ecosystem service provision map, using the 'Sample' tool and exported into IBM SPSS Statistics v.22 to calculate scores by percentiles. This allowed further analysis to see what proportion of the higher scoring points were in the SSSI or non-SSSI areas. The Shapiro-Wilk test was not used due its requirements of small to medium sample sizes, as there were 892,829 samples in this design, Kolmogorov-Smirnov was used instead. As all ecosystem services (including the biodiversity and combined ES) are not normal (see Table 4.4.1), the Mann-Whitney U test was used as an alternative to using a t-test on the values inside the SSSI and outside.

Top 10% of ecosystem service provision

To see where the highest provision of ecosystem services were, the data was separated into categories. To split the data into percentiles, the data was first ranked using the 'Rank Cases' tool from smallest to largest. The 'Compute Variable' tool was then used to transform the ranks by using the numeric expression '((Ecosystem_services – 0.5) / Total_number_samples) * 100'. This provided a new column with exact percentile scores (as decimal numbers). The 'Select cases' tool was then used with 'If' expressions (i.e. 'Ecosystem_services >= 90' for the 90th percentile) to copy the cases to a new dataset.

4.4 Results

A total of ten ecosystem services were mapped and normalised (with nutrient retention split into nitrogen and phosphorous retention); covering cultural, supporting, provisioning and regulating services, with all ecosystem services found to show significant ($p < 0.001$) non-normal distributions (Table 4.4.1).

Table 4.4.1: Test for normality on all extracted values for ecosystem services in the New Forest National Park (df= 892,829, with Lilliefors significance correction). Kolmogorov Smirnov tests show either significance (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significance (n.s.) for all ecosystem services.

	Test statistic
Biodiversity	0.446***
Carbon storage	0.328***
Flood risk mitigation	0.392***
Livestock	0.447***
Nitrogen retention	0.319***
Phosphorous retention	0.332***
Recreation	0.400***
Timber	0.420***
Tranquillity	0.023***
Water yield	0.123***
Combined ES	0.205***

Hydrological ecosystem services

Water yield was output by watershed in the New Forest National Park (Figure 4.4.1). The highest water yield was found in a small pocket near the northern reach of Beaulieu River, near Mill Dam. The next highest water yield, were Bartley Water (0.66), followed closely by Ditchend Brook and Fletchwood Trib (both with a value of 0.65), Cadnam River (0.64) and Blackwater (0.63) (all river watersheds and in the northern parts of the National Park). A north-south gradient of high to low water yield was observed, with the lowest water yield belonging to the southern coastal watersheds. Nitrogen retention was visually similar to phosphorous retention (Figure 4.4.2 Figure 4.4.3). High values are visualised as blue, with mid values (visualised as yellow and orange) found to be adjacent to the highest values, with the lowest values (visualised as red) furthest from the high values. This pattern appears strongest within the SSSI boundary, within the central region of the New Forest National Park, with more mid values within the SSSI, and more low values outside. Phosphorous retention can be seen to follow the same pattern as nitrogen retention.

Other ecosystem services

Carbon storage was highest in woodland areas in both ‘coniferous’ and ‘broadleaved, mixed and yew’, with the latter having the highest values. Most of these high-value areas are within the SSSI (Figure 4.4.4). Recreation values were highest around the towns of Lyndhurst, Brockenhurst, Burley, Beaulieu and near Fritham (Figure 4.4.5). Other high areas were noted within the SSSI, for example near the Bolderwood Arboretum Drive. Livestock values are highest outside the SSSI areas, being linked to the grassland habitat types (Figure 4.4.6). As timber values were only linked to the woodland categories of ‘coniferous’ and ‘broadleaved, mixed and yew’, values are as high as expected in these areas, with the bulk of high values falling within the SSSI area (Figure 4.4.7).

Flood risk mitigation also shows highest values in the SSSI areas, with the mid-low values (visualised as orange) falling outside the SSSI (Figure 4.4.8). The normalised tranquillity map shows most of the low values falling outside of the SSSI area, apart from an area of high value near the River Beaulieu (Figure 4.4.9). The bulk of the high tranquillity areas can be seen to be within the northern and south-eastern part of the SSSI.

Biodiversity

The biodiversity map (Figure 4.4.10) shows the artefacts of the combined rasters of varying resolutions, and shows a low ‘background’ value, with pixels of high and medium value biodiversity. A visual assessment of the map shows that the bulk of these mid and high values are within the SSSI. A distinct deviation from this is the southern tip of the National Park, where there is no SSSI designation, but multiple high points of biodiversity value along the coast exist.

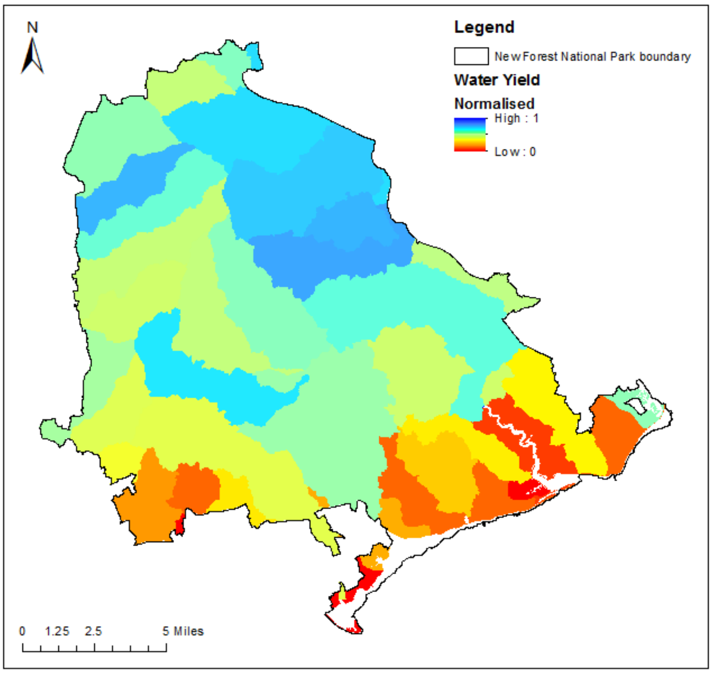

Figure 4.4.1: Normalised mean water yield per 25 m x 25 m pixel (mm) for catchments in the New Forest National Park.

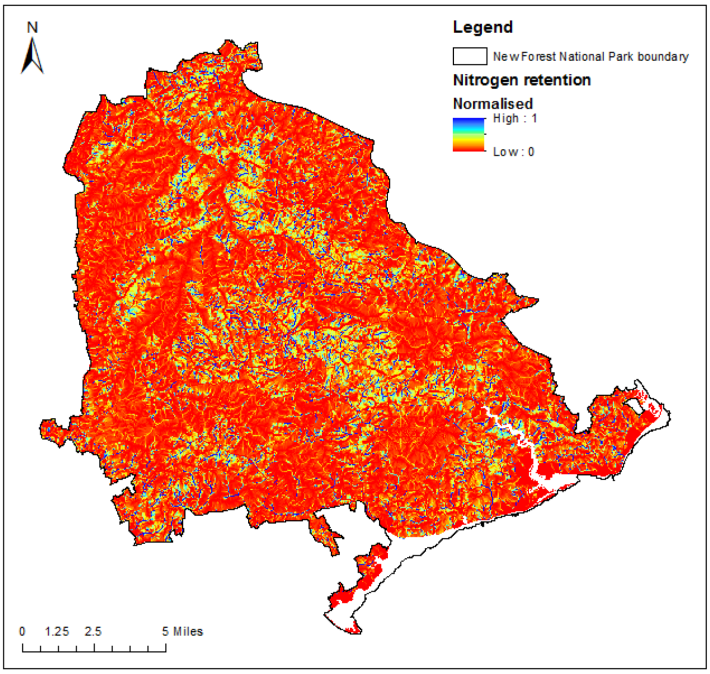

Figure 4.4.2: Nitrogen retention in the New Forest.

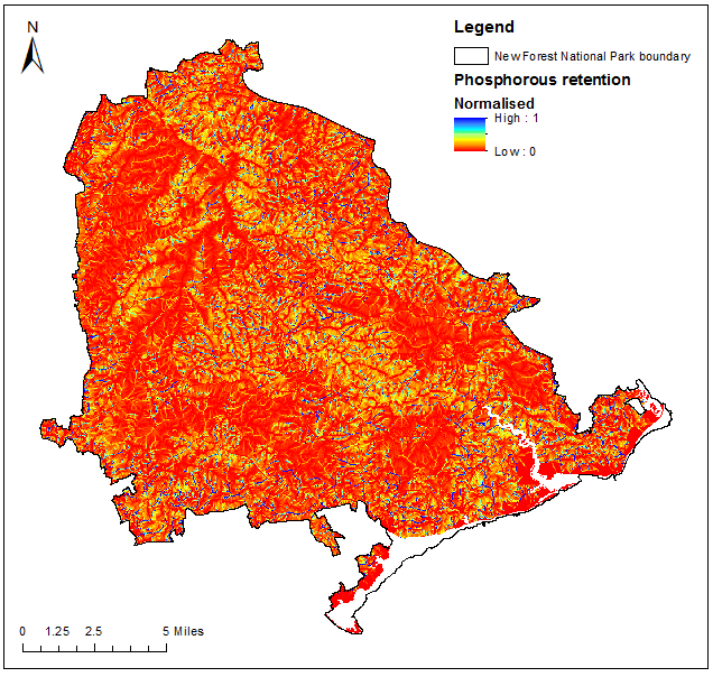

Figure 4.4.3: Phosphorous retention in the New Forest

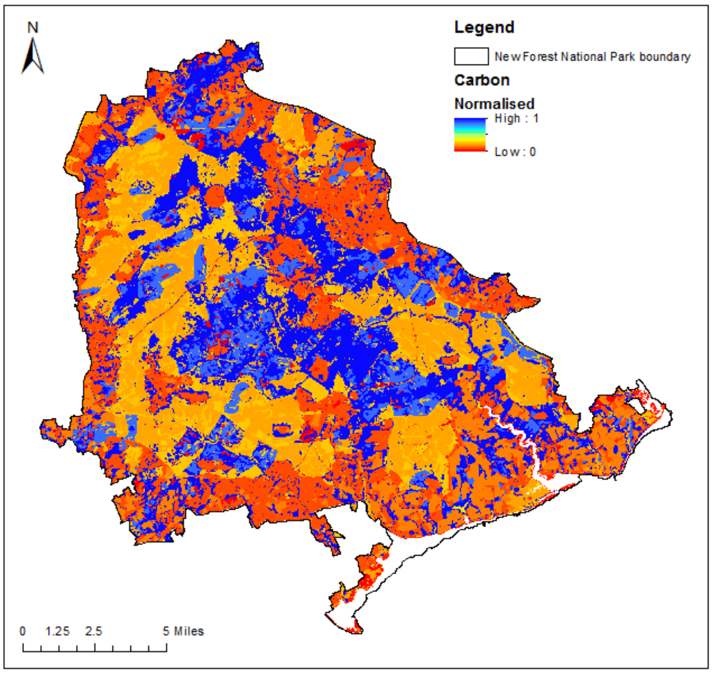

Figure 4.4.4: Normalised current carbon stocks in the New Forest.

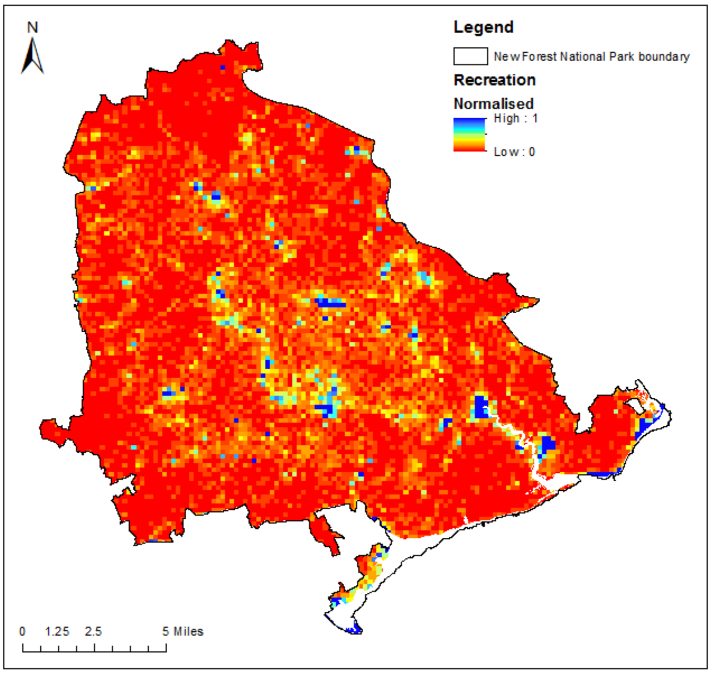

Figure 4.4.5: Average photo-user days per year in the New Forest, as a proxy for recreation and visitation, modelled using InVEST and normalised. Pixels are 25 m x 25 m, though the model only allowed a minimum resolution of 250 m, hence the data has been made finer for comparative purposes.

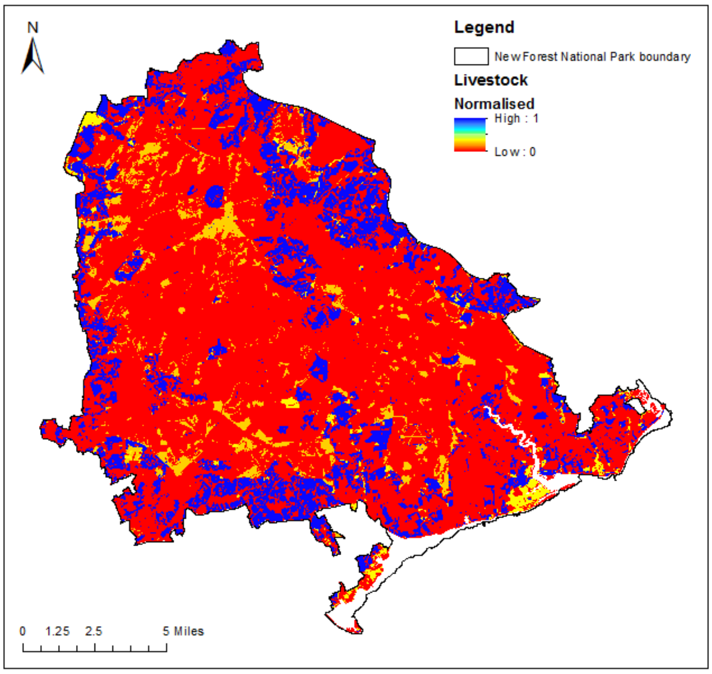

Figure 4.4.6: Normalised values of livestock in the New Forest.

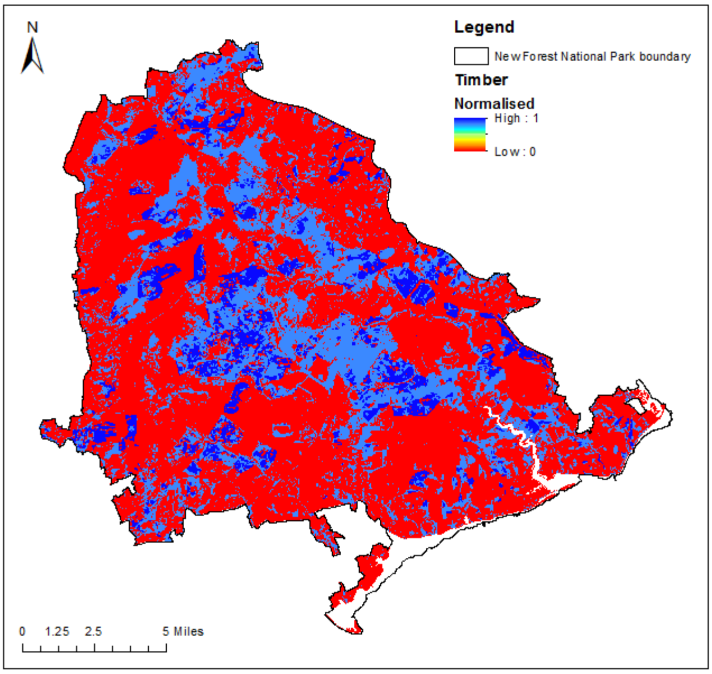

Figure 4.4.7: Normalised timber production in the New Forest.

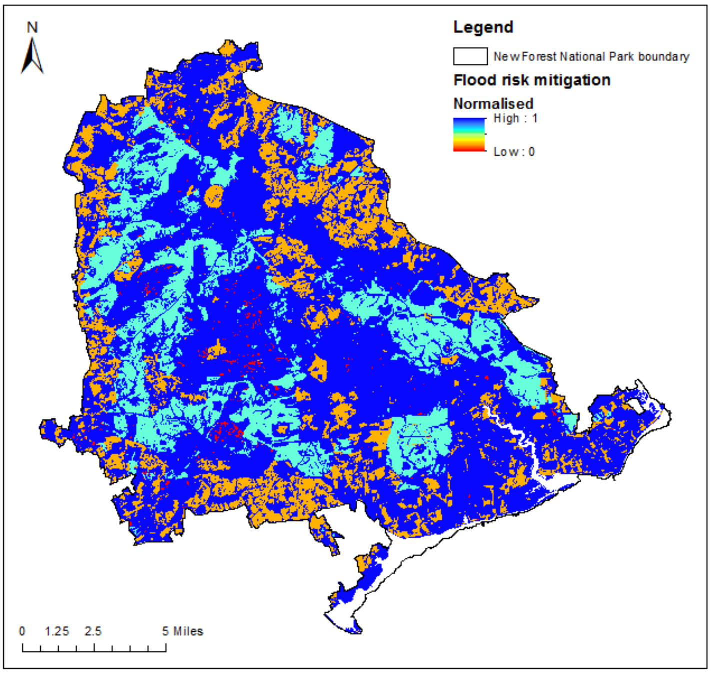

Figure 4.4.8: Normalised flood risk mitigation for the New Forest.

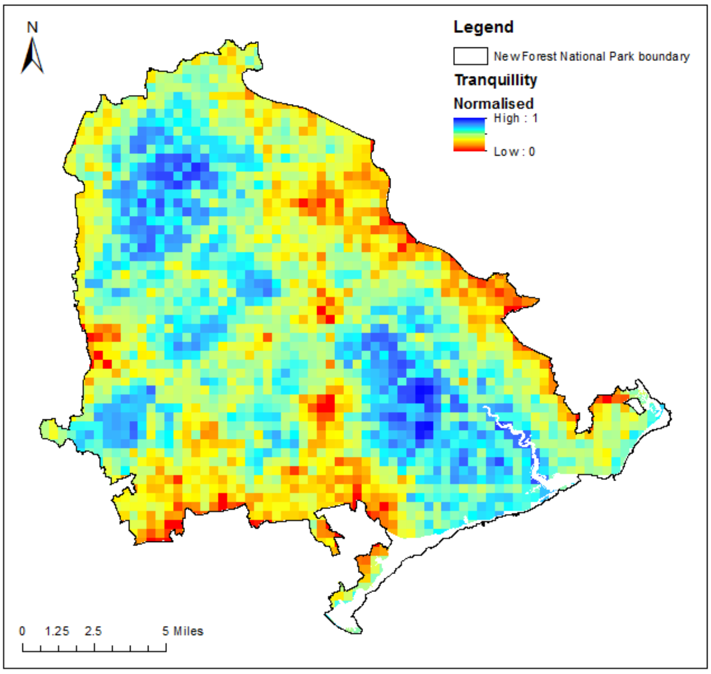

Figure 4.4.9: Normalised value of tranquillity in the New Forest.

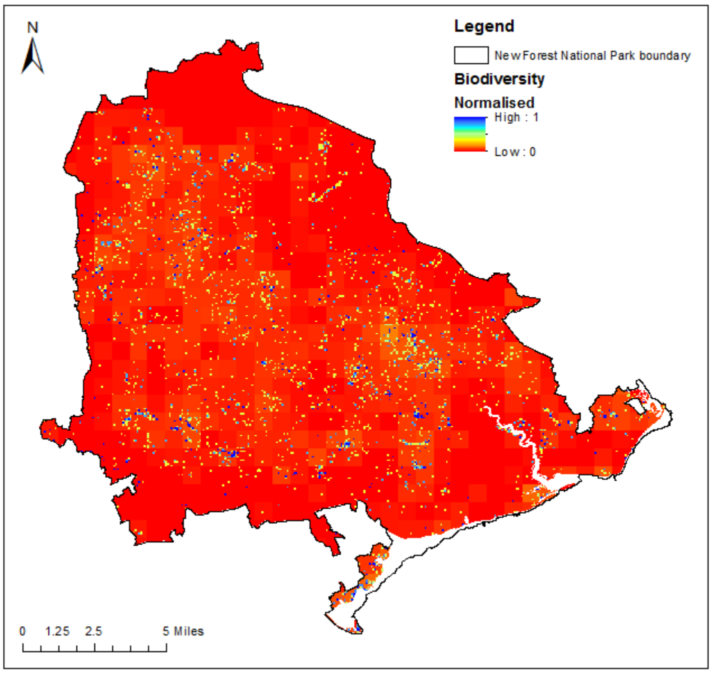

Figure 4.4.10: Normalised biodiversity values in the New Forest

Combined ecosystem service map

The combined map of ecosystem services (Figure 4.4.11 andFigure 4.4.12 with SSSI mask) show that the high value areas lie mostly within the SSSI boundary. There are noticeable exceptions with areas in the north and south-east of the park showing high values.

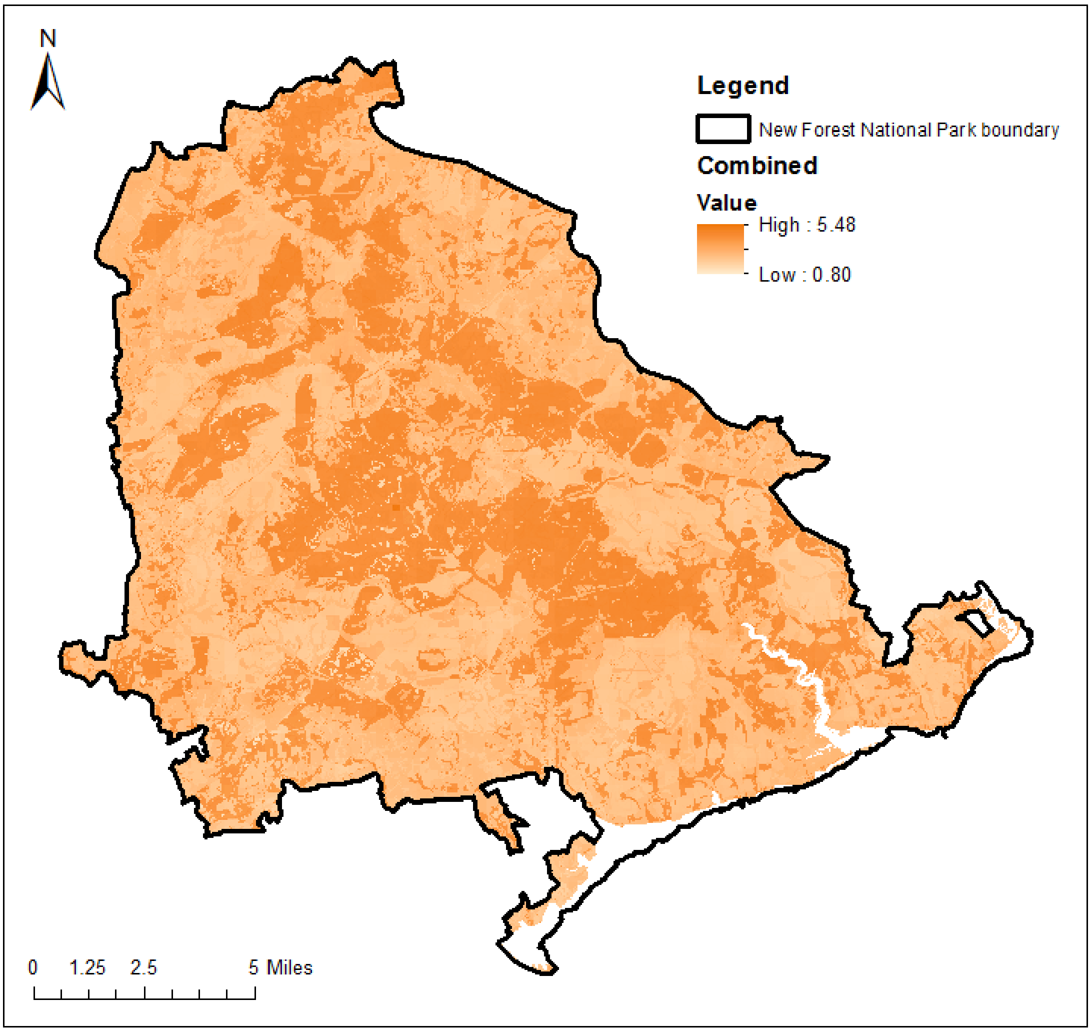

Figure 4.4.11: Combined ecosystem service map of the New Forest National Park. All ecosystem services mapped, with the addition of biodiversity were summed together at a 25 m x 25 m resolution. The white areas within the New Forest Park boundary are where no data has been calculated.

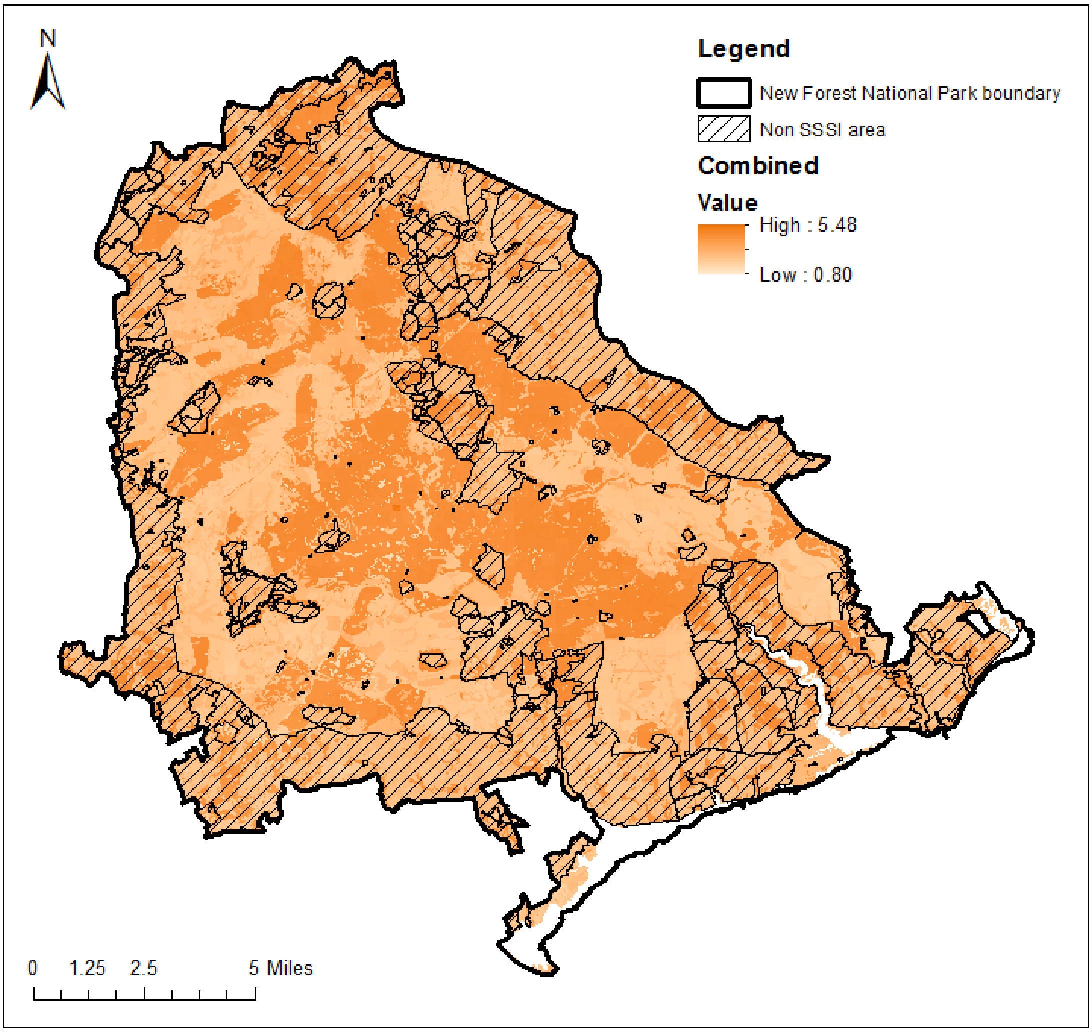

Figure 4.4.12: Combined ecosystem service map of the New Forest National Park with non-SSSI areas shaded. All ecosystem services mapped, with the addition of biodiversity were summed together at a 25 m x 25 m resolution. The white areas within the New Forest Park boundary are where no data has been calculated.

Ecosystem service correlation

Relationships between ecosystem services were explored using Spearman correlation coefficient tests (Table 4.4.2). Nitrogen retention and phosphorous retention showed a strong positive correlation. There was a strong negative relationship between flood risk mitigation and livestock, with carbon storage having a strong positive relationship with timber. Timber and carbon have positive moderate relationships with flood risk mitigation; both having a moderately negative relationship with livestock. The combined ecosystem service provision values correlate strongly with carbon storage and timber, whilst also moderately positive with flood risk mitigation - indicating these services are driving the combined values. The negative relationships infer trade-offs with the positive relationships inferring synergies.

Correlation matrix

Table 4.4.2: Correlation analysis was conducted for all ecosystem services in the New Forest National Park (df= 892,829, with 2-tailed test). Coefficient shown is from Spearman correlation coefficient tests (two-tailed) with either significance (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significance (n.s.).

	Biodiversity	Carbon storage	Flood risk mitigation	Livestock	Nitrogen retention	Phosphorous retention	Recreation	Timber	Tranquillity	Water yield
Biodiversity										
Carbon storage	0.222**									
Flood risk mitigation	0.072**	0.546**								
Livestock	-0.230**	-0.477**	-0.610**							
Nitrogen retention	-0.016**	0.305**	0.210**	-0.139**						
Phosphorous retention	-0.101**	0.186**	0.149**	-0.049**	0.939**					
Recreation	0.231**	0.034**	0.073**	-0.041**	-0.025**	-0.044**				
Timber	0.062**	0.793**	0.524**	-0.411**	0.302**	0.260**	0.021**			
Tranquillity	0.358**	0.274**	0.132**	-0.248**	0.058**	-0.028**	-0.012**	0.140**		
Water yield	-0.038**	0.069**	-0.046**	0.032**	0.034**	0.020**	0.058**	0.070**	-0.101**	
Combined ES	0.021**	0.700**	0.436**	-0.008**	0.344**	0.310**	0.033**	0.815**	0.281**	0.212**

ES provision in protected areas

All ecosystem services analysed show a significant difference inside the SSSI and outside the SSSI area (Table 4.4.3). Carbon storage was much higher inside the SSSI with a median score of 0.248, compared to 0.068 outside, showing a 365% increase. Nitrogen retention showed a 100% increase inside the SSSI with a median value of 0.002. Phosphorous retention is actually slightly higher outside the SSSI at 0.003 compared to a median of 0.002 inside. It is interesting to note that phosphorous and nitrogen retention were significantly and strongly positively correlated (Table 4.4.2). Further Spearman correlation tests show strong positive correlations inside (rho = 0.962, $P > 0.01$, n = 494,987) and outside the SSSI (rho = 0.946, $P > 0.01$, n = 397,842) for these two services. Median scores for recreation show an increase within the SSSI to 0.004 from < 0.001 outside. Tranquillity has the highest median score of 0.702 inside the SSSI, dropping to 0.605 outside. Water yield shows a marginal increase from 0.554 outside the SSSI to 0.561 inside the SSSI area.

Flood risk mitigation shows the same median scores of 1.000 for both inside and outside the SSSI, though mean scores show a higher value for SSSI at 0.881 compared to 0.734 outside. The median scores for agriculture, biodiversity, livestock and timber were too low to round up to a comparable number. Using the mean score for these services it was found that agriculture is much higher outside at 0.102 compared to 0.001 inside, showing over a 100% increase in mean value. Biodiversity is higher at 0.001 compared to 0.000. Livestock is considerably higher outside the SSSI at 0.394 compared to a 0.041 inside. Timber value is higher inside the SSSI at 0.371 compared to 0.211 outside.

Both SSSI and non-SSSI land were found to show significantly different-than-expected ecosystem service provision by percentile (Table 4.4.4). The percentiles were calculated by first ranking the combined ecosystem service provision before splitting the data for percentile. It can be seen that after the 50th percentile (values that are 50% or higher), increase continuously though to the 90th percentile (values that 90% or higher), with the 70th, 80th and 90th percentile being highly significantly different from expected. At the 90th percentile it was found that there was an increase of 23% of ecosystem service provision in SSSI areas.

Table 4.4.3: Mean ecosystem service provision across SSSI and non-SSSI land in the New Forest National Park. Difference between ecosystem service provision in and outside the SSSI area were tested (df = 892,828, with asymptotic significance (2-sided test). Independent-Samples Mann-Whitney U tests show either significance (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significance (n.s.) for all ecosystem service services.

	Mann-Whitney U	Median ecosystem service score		Mean ecosystem service score	
		SSSI	Non-SSSI	SSSI	Non-SSSI
Biodiversity	533.273 ***	0.000	0.000	0.001	0.000
Carbon storage	444.399 ***	0.248	0.068	0.525	0.306
Flood risk mitigation	175.011 ***	1.000	1.000	0.881	0.734
Livestock	-357.146 ***	0.000	0.000	0.041	0.394
Nitrogen retention	67.038 ***	0.004	0.002	0.015	0.013
Phosphorous retention	-74.927 ***	0.002	0.003	0.005	0.008
Recreation	164.611 ***	0.004	0.000	0.007	0.007
Timber	182.153 ***	0.000	0.000	0.371	0.211
Tranquillity	360.896 ***	0.702	0.605	0.704	0.608
Water yield	85.503 ***	0.561	0.554	0.553	0.531
Combined ES	104.785 ***	2.725	2.537	3.098	2.812

Table 4.4.4: Table illustrating the percentage of combined ecosystem service provision within varying percentile ranks for SSSI designated and non-SSSI land within the New Forest National Park. The percentile rank value indicates the percentage of scores in its frequency distribution under it, e.g. the 90th percentile rank indicates the ecosystem services that score at 90% or higher. Chi-squared tests of independence were performed to examine the relationship between observed and expected points by percentile, showing either significance (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significance (n.s.).

	SSSI designated areas (%)			Non SSSI areas (%)			
Percentile rank	**Observed**	**Expected**	**Difference**	**Observed**	**Expected**	**Difference**	**x^2**
10th	53.77	55.44	-1.67	46.23	44.56	1.67	0.05 (n.s)
20th	52.31	55.44	-3.13	47.69	44.56	3.13	0.28 (n.s)
30th	52.72	55.44	-2.72	47.28	44.56	2.72	0.20 (n.s)
40th	56.54	55.44	1.10	43.46	44.56	-1.10	0.02 (n.s)
50th	62.08	55.44	6.64	37.92	44.56	-6.64	1.53 (n.s)
60th	68.41	55.44	12.97	31.59	44.56	-12.97	6.29 *
70th	73.33	55.44	17.89	26.67	44.56	-17.89	12.24 **
80th	78.50	55.44	23.06	21.50	44.56	-23.06	20.60 ***
90th	82.36	55.44	26.92	17.64	44.56	-26.92	28.25 ***

4.5 Discussion

The most important result of this study, and most anticipated, was the provision of ecosystem services that the SSSI protected area provided. Results are not striking if only the average values of the ecosystem services are considered. However, once data was analysed at percentiles, with the highest scores of combined ecosystem services analysed, a clear pattern emerges. The higher the combined value of ecosystem service provision: the increased likelihood of it being within the SSSI.

There are various limitations to mapping ecosystem services and deciphering areas of high ecosystem service provision. This study has used proxies. This method has been criticised for being unsuitable for assessing performance of conservation strategies (Eigenbrod et al., 2009). These 'imperfect proxies' can lead to confusing spatial location with habitat type (Eastwood et al., 2016). This has been demonstrated in earlier research, whereby UK uplands (which are usually within protected areas), have high levels of carbon storage and are far from population centres, show low recreation usage, thus presenting a correlation between low recreation and high carbon storage (Reed et al., 2009, Eastwood et al., 2016). Another example is a study by Durán et al. (2013) which showed Chile's protected areas having a low representation of ecosystem services was explained due to the extent of ice and bare rock within the protected regions. Though using habitat proxies can be the best available solution when data is lacking. Data availability for ecosystem service mapping is still limited, this is not the case for all ecosystem services, e.g. data on carbon is abundantly available (Eastwood et al., 2016). Cultural ecosystem services are difficult to value, due to spatial and temporal heterogeneity (Martín-López et al., 2009), hence use of New Forest specific data for the recreation mapping makes this study more robust. Values for timber production were not specific to the region, originating from data for the county of Dorset, located adjacent to the New Forest National Park. Future refinement of this research would need elicitation of expert knowledge, though data on ecosystem service at local scales is still rare (de Groot et al., 2010). The resolution of the data used can also have a direct impact on the spatial covariance of ecosystem services, with Anderson et al. (2009) demonstrating that using a 100 km x 100 km resolution, compared to 2 km x 2 km resolution, showed the relationship between carbon and biodiversity was significantly less negative at the courser resolution.

My findings suggest that the higher values of ecosystem service provision exist within in the boundaries of the SSSI. There is a 16.49% increase in the top 30% of combined ecosystem service provision values in the SSSI. The authors are not aware of other studies analysing the top proportion of ecosystem service delivery in protected areas. Similar studies include Castro et al. (2015) whom found that protected areas in Andalucía supply 59% of regulating ecosystem services, whilst only occupying 48% of land and Eastwood et al. (2016) whom found ecosystem service delivery higher in protected areas. Our findings reinforce this argument, showing that the regulating services analysed have a higher median value in the SSSI; with nitrogen retention having double the value, carbon storage being 365% higher within the SSSI, flood risk mitigation having the same median but higher mean value. The only regulating service with a higher median value outside the SSSI area was phosphorous retention. Castro et al. (2015) argues that protected networks (namely RENPA and Natura-2000 in Spain) are important to preserve intermediate services, with final services occurring outside the protected network. It is important to understand the provisioning of ES in protected areas, as Protected Areas are not always designed optimally for the protected of biodiversity or ES (Xu et al. 2017). Though it has been discussed by Bridgewater and Babin (2017) that United Nations Educational, Scientific and Cultural Organization (UNESCO) Biosphere Reserves are designed to protect biodiversity and ecosystem services (including cultural services).

Results show that the mean for biodiversity in the SSSI is higher than in the non-SSSI areas (with the medians being the same). This follows Castro et al. (2015) whom found that biodiversity for threatened species was greater in the protected areas. Whereas my values are much lower, Eigenbrod et al. (2009) found that protected areas overall in England had 3.3 times the expected biodiversity. Carbon storage was found to be much greater in the SSSI, showing similar results to Castro et al. (2015) who found that protected areas were significantly relevant for preserving carbon stocks, and Anderson et al. (2009) who found carbon storage to be 1.8 times that expected in protected areas.

Recreation was found to have a higher median value inside the SSSI area, though the same means for both inside and outside the SSSI in the New Forest. This is in slight contrast to Eigenbrod et al. (2009) who found that recreation was under-represented in protected areas with a ratio of 0.88. My findings are similar to Chan et al. (2006) who found that found that biodiversity driven conservation strategies provided good recreation delivery. Interestingly, Eigenbrod et al. (2009) used results from the 'England Leisure Visits Survey 2005', whereas Chan et al. (2006) estimated recreational value based on natural habitat area, access and distance to population centres. This study used the InVEST Recreation and Visitation model that used actual data from geo-referenced photos from the photo-sharing website Flickr.com. Due to this data being based on actual photos taken by people at specific geographic points, we can be sure people visited these areas, and thus is more robust than Chan et al. (2006).

By modelling and mapping multiple ecosystems, I was able to demonstrate synergies and trade-offs between the various ecosystem services, with a slight contrast to Nelson et al. (2009) whom found little trade-offs with scenarios of multiple high ecosystem services and biodiversity.

Timber and carbon storage showed the strongest synergy. If timber value is assumed to be related to tree species composition, then the synergy can be explained. Studies have shown that an increase in tree species richness has a positive relationship with multiple ecosystem service provision, including soil carbon storage, as well as tree biomass and berry production (Gamfeldt et al., 2013). Biodiversity was found to a have a weak correlation with carbon storage, a result that contrasts Anderson et al. (2009) who showed regionally the 'SU' and 'SZ' National Grid squares that encompass the New Forest National Park were positively significant, even though across Britain important carbon storage areas tended to have low biodiversity.

Nelson et al. (2009) investigated alternative land-cover scenarios in the Willamette Basin, Oregon, USA. They found that those scenarios with high values for multiple ecosystem services had high biodiversity values, inferring little trade-offs. Our research showed very weak to weak correlations between biodiversity and ecosystems services, inferring a lack of a relationship with the strongest of these correlations with tranquillity. This is explained partially by Anderson et al. (2009) who investigated biodiversity and ecosystems service covariance, surmising that one can reach very different results, in different regions of Britain with a congruency, when analysed with Britain as a whole.

The link between tranquillity and biodiversity, especially in protected areas, has been little explored in scientific literature. The CPRE dataset used within this study used indicators that are reflective of biodiversity, i.e. seeing wild and natural landscapes, thus is not unexpected. Though a strong positive correlation with the NERC Section 41 species that were used to map biodiversity demonstrates the innate link between the two. This is similar to results from Cimon-Morin et al. (2013) whom found a greater spatial correlation of regulating, supporting and cultural services compared to provisioning services to biodiversity.

Anderson et al. (2009) suggest that regional approaches are key, rather than an establishment of general patterns of covariance between biodiversity and ecosystem services, demonstrating that strong regional variations between biodiversity and ecosystem services exist. This study is strengthened by the fact it focuses on one area, the New Forest National Park, rather than attempting to generalise ecosystem service provision.

The New Forest Park Authority (2015) lists several goals in their 2015-2020 Partnership Plan, which were linked to ecosystem service provision. Part of the Partnership Plan states having 60% of the SSSI's in the New Forest listed as 'favourable condition' by 2020 (New Forest Park Authority, 2015). Though the ecosystem service provision assessment undertaken here does not measure the condition of the areas, it is worthwhile to note that the combined ecosystem service provision within the SSSI was higher (with a median of 2.725), compared to non-SSSI land (with a median of 2.537). As ecosystem functions can underpin ecosystem services, this may suggest that the habitat condition is better in the SSSI, than outside it. Tranquillity was actively managed for, with the New Forest Park Authority (2015) aiming to maintain and enhance the tranquil areas of the National Park. Results show that the mean score within the SSSI area was 0.702, and 0.605 outside, showing that a higher level of tranquillity is being maintained within the SSSI.

An aim to promote an integrated approach to river catchment management led the (New Forest Park Authority, 2015) to monitor and improve water quality. My results suggest the nitrogen retention is twice as higher in the SSSI (with a median of 0.004), than the non-SSSI area (with a median of 0.002). Though phosphorous retention was found to be lower in the SSSI (with a median of 0.002), compared to the non-SSSI area (with a median of 0.003). Water yield was similar across SSSI and non-SSSI areas.

In line with the Government's Biodiversity 2020 targets, the New Forest Park Authority (2015) is looking to actively monitor wildlife and habitats, with the addition of maintain the carbon sink in the New Forest. Results show that biodiversity values were higher in SSSI (if the means are considered), with carbon storage being substantially higher in the SSSI. Under the promotion of a climate adaption plan, the New Forest Park Authority (2015) a plan for flooding and management to reduce flooding are currently being prepared. This work highlights that the flood mitigation in the SSSI and non-SSSI areas are both similar if the median values are considered, though a closer look at the mean scores suggests the value is higher in the SSSI areas.

There are eight management priorities listed for recreational use of the New Forest National Park in the Partnership Plan (New Forest Park Authority, 2015), with livestock being mentioned as appealing for visitors. Results show that recreation was marginally higher in the SSSI, though livestock values are much higher if the mean scores are considered, in the non-SSSI areas, as would be expected. Lastly ‘economic well-being’ is actively managed, with private sector investment in forestry encouraged. The timber score was found to be higher in the SSSI, as would be expected.

Managing for multiple ecosystem services in protected areas can be challenging, though numerous studies have shown that ‘bundles’ of services can often be identified. Lin et al. (2018) found that in the Three Parallel Rivers Region, China, most ecosystem services investigated showed medium/high interactions, though four distinct ES bundles could be identified. The multi-functionality of landscapes is better reflected with the use of the bundles concept, though the association between landscape heterogeneity and multiple ecosystem services is not final (Crouzat et al., 2015, Mastrangelo et al., 2014).

The authors have attempted to expand the range of ecosystem services analysed, though data availability on ecosystem services is still a constraint (Naidoo et al., 2008, Eigenbrod et al., 2009). Inclusion of different ecosystem services would alter the overall provision values, hence any study investigating overall provision will be biased by the ecosystem services chosen. Eastwood et al. (2016) refers to this as ‘effectively a sub-sample’, with a distorted picture of ecosystem service delivery being presented. This study has shown that there are relatively few strong correlations between the thirteen ecosystem services; hence use of a large number of ecosystems services is required when investigating overall ecosystem provision. These results mirror those of Eastwood et al. (2016) who found little correlation between 24 ecosystem services investigated in protected areas. Though one aspect is clear; combined ecosystem service provision for the higher combined values is found at increased levels within the SSSI within the New Forest, suggesting that the protected areas within the New Forest provide protection beyond their scope of preserving biodiversity. There is a strong suggestion that the ‘priority goals’ of the New Forest Park Authority (2015) are currently being met more in the SSSI areas, though there are still improvements that could be made, potentially by focussing on the non-SSSI areas more.

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Chapter 5

The impact of changing climate on ecosystem services in Protected Areas

5.1. Abstract

The realities of a changing climate have been documented extensively, with a growing field of literature on the effects of climate change on ecosystem services, though information on the implications within protected areas is limited and cultural ecosystem services are under-represented in the literature. This study was conducted within the Special Area of Conservation (SAC) within the New Forest National Park in the south of England. A myriad of ecosystem services (including cultural services, which are of importance in this area) were investigated and mapped, before a multi-criteria analysis was conducted, comparing service delivery across low, medium and high emissions scenarios for the year 2116. The results of this study were clear; climate change will lead to a decrease in ecosystem service provision, for all climate change scenarios. The New Forest National Park will change, from its species composition to the vast array of ecosystem services it delivers with future management decisions needing to consider the decrease in ecosystem service provision. Since ecosystem service provision is regionally variable, it is important that management decisions that impact land and protected areas acknowledge the impact of climate change, including its uncertain nature.

5.2. Introduction

Greenhouse gases in the atmosphere, coupled with extensive global land use changes, are having an impact on climate (Pielke, 2005). Climate in turn has been seen to have an impact on nature, including initial species range shifts in response to climate change already being detected (Heikkinen et al., 2006, Brown et al., 2016, Hickling et al., 2006). It has also been suggested there is a link between climate change in the past and societal crises, with the 'golden' and 'dark' ages in Europe being causally related to climate change (Zhang et al., 2011).

It has been argued that ecosystems may be susceptible to 'tipping points'. Hastings and Wysham (2010) suggest that 'regime shifts' can occur with no forewarning, with Lenton (2011) stating that human induced climate change could push several 'tipping-elements' (i.e. dieback of the Amazon rainforest) past a point of tipping. These points, whereby thresholds are crossed, have been shown to shift to irreversible states in localised ecological systems (Barnosky et al., 2012). This has prompted international agreements to reduce the extent of climate change through the reduction in greenhouse gas emissions from the previous target of 2 °C above pre-industrialisation (Elkin et al., 2013), and to 1.5 °C since the Paris climate conference (COP21) in late 2015 (Rhodes, 2016).

With further predicted rises in global temperatures, ecosystem services will be significantly and extensively impacted (Millennium Ecosystem Assessment, 2005). Essential ecosystems services are already under stress, with further climate change likely to increase the burden in the future (Mooney et al., 2009). The Intergovernmental Panel on Climate Change (IPCC) (2014) clearly stated that the impact of a changing climate can change over time and is uncertain in nature. Large changes in land use or climate reflect in ecosystem service supply, with most being affected negatively (Schröter et al., 2005). A way to understand the changes to ecosystem services is through the use of comparing spatial or temporal trends. Various methods for assessing trends in ecosystem services including mapping, trade-off curves, star diagrams, correlation coefficients, PCA/ MCA and overlap analysis (Mouchet et al., 2014), these are discussed in more depth in Chapter 1.

Despite the uncertainty behind a changing climate, especially at a regional scale, it is important that future climate be considered in the management decision planning process where the aim is to protect ecosystem services. This was demonstrated by Runting et al. (2016b) who found that the ecosystem services in coastal wetlands could decline if sea level rise was ignored during any planning stages of management. A review of over 1,500 studies published on climate change and ecosystem services found that the majority reported a negative impact (Runting et al., 2016a).

Much research has focused on ecosystem service provision and co-location of services under the current climate, for example use of conservation networks for biodiversity and ecosystem service flow protection (Chan et al., 2006) and models of ecosystem services under varying land use scenarios (Nelson et al., 2009). Other studies have investigated bundles of ecosystem services and correlations between them, for example in Southern Africa (Bohensky et al., 2006), Quebec, Canada (Raudsepp-Hearne et al., 2010), the Grote Nete basin, Belgium (Van der Biest et al., 2014) and the Yangtze River Delta, China (Yang et al., 2015). Other studies have examined the impact of climate change on ecosystem services. Lamarque et al. (2014) investigated the trade-offs incurred by ecosystem services under a changing climate using a functional trait-based framework for mountain grasslands in the French Alps. They found that ecosystem services were less sensitive to induced land management change than climate change. Elkin et al. (2013) demonstrated that certain ecosystem services (gravitation hazard protection such as from avalanches and timber production) would be sensitive to a 2 °C rise in the Saas and Disma valleys in the Alps, Switzerland.

Climate adaption services are the social ability benefits that are received from ecosystems from their ability to adapt to climate change (Lavorel et al., 2015). With increased risks to provisioning and cultural services, maintenance of regulating and services is often recommended (Abson and Termansen, 2011). It has been predicted that ecosystems will undergo unexpected changes due to climate change, due to warming globally of >+3 °C (Schellnhuber et al., 2012). The adaptation services concept allows an understanding of key ecological mechanisms that support an ecosystem's ability to adapt to a changing climate. An ecosystem in good condition is necessary for its ability to resist to direct and indirect impacts of climate change, or conversely to allow it to transform into a state that supports social adaption (Lavorel et al., 2015)

Globally, 15.4% of land is under protected area status (Juffe-Bignoli et al., 2014) with the Convention on Biological Diversity establishing a goal of 17% of terrestrial land being designated under protected area status by 2020 (under the Aichi Targets) (Convention on Biological Diversity (CBD), 2011). It has been reported that indigenous lands and other protected areas encompass approximately 312 billion tons of carbon, with the potential for increased carbon storage if protected lands were increased in size and existing ones offered stronger protection (Campbell et al., 2008), with forest protected areas being the most important for climate-change mitigation (Ricketts et al., 2010). The Reducing Emissions from Deforestation and Forest Degradation (REDD) policies explicitly demonstrate this link.

Several studies have investigated the impact of changing climate in protected areas, with the focus often on biodiversity (i.e. Hannah et al. (2007) on expanding protected areas to encompass range shifts for conservation) or on their role in mitigation (i.e. Soares-Filho et al. (2010) on forest carbon in the Brazilian Amazon). Thomas and Gillingham (2015) have demonstrated the value of protected areas over the last 40 years of anthropogenic climatic warming, protecting species through range shifts and acting as 'stepping-stones' as breeding sites, in addition to species remaining protected within the protected area. However, research on protected areas for ecosystem services is limited.

The impact of climate change on the provision of ecosystem services within protected areas has not been investigated. With the importance of protected areas for conservation, and research showing the added advantage of protecting ecosystem services (Castro et al., 2015, Eastwood et al., 2016), as discussed in Chapter 4, it is vitally important to understand how ecosystem provision will be impacted in these areas. Past studies that have investigated protecting processes, for example, Anderson and Ferree (2010) investigated geologic factors controlling total species diversity to identify and protect drivers of total species diversity under current and future climate scenarios. The study included the entirety of the North-eastern US and Maritime Canada and found that protecting geological settings would be more robust method than using species-level predictions.

For effective management practices, there needs to be an understanding of trade-offs among ecosystem services, though trade-offs can differ temporally and spatially (Deng et al., 2016). Correlation and cluster analysis can be used with ecosystem service indicators to map these trade-offs. Examples include studies undertaken on the trade-offs between biodiversity and ecosystem services in Europe by habitat condition, finding that habitats in favourable condition provided higher biodiversity and higher potential to supply cultural and regulating services, inferring no trade-offs between them (Maes et al., 2012). Trade-offs between carbon storage and primary production in Britain have been investigated by Maskell et al. (2013), who found that ecosystem services were constrained by a 'fundamental productivity gradient'. Another study focused on Dorset, UK at the landscape scale of trade-offs between agricultural production, carbon storage biodiversity (Jiang et al., 2013), finding that between 1930 and 2000, the distribution of carbon storage and provisioning services had become more unequally distributed, with losses in biodiversity and increases in agriculture. Chapter 4 is an example of investigating trade-offs and synergies of ecosystem services within the SSSI protected area within the New Forest National Park. I found that timber and carbon were strongly correlated, though carbon and agriculture had a negative correlation and biodiversity had few trade-offs with other ecosystem services.

The New Forest National Park was used as a case study area in this chapter due to the protected area (Special Area for Conservation) within the New Forest National Park boundaries. The New Forest National Park includes a myriad of habitats, including beech and oak dominated Ancient and Ornamental woodlands (Grant and Edwards, 2008). The New Forest Committee (2003) have suggested that climate change is a potential problem in the New Forest National Park. Grant and Edwards (2008) suggest that the existing Ancient and Ornamental woodlands in the New Forest will be increasingly difficult to preserve in their current form, due to stress on Beech trees in the future under expected climate change. Various studies have shown that climate change will affect tree species distribution, with losses in diversity in areas not being offset by gains in other parts of Europe (Thuiller et al., 2011). Hanewinkel et al. (2013) suggests that the economic value will drop with a decline in those species that are economically valuable with a loss of 14 - 50% by 2100.

A wide spectrum of ecosystem services were investigated, with cultural ecosystem services (recreation value, aesthetic value and heritage value) included due to the under-representation in the body of literature investigating the impact of climate change on ecosystem services (Runting et al., 2016a). Contrasting counter-factual scenarios have been used in other assessments of ecosystem assessment (e.g. Balmford et al. (2011) with conservation scenarios of losing wild nature and Birch et al. (2010) on forest restoration in dryland areas in Latin America). Alternative scenarios were used in terms of the low, medium, high and control emissions scenarios from the UK Climate Projections (UKCP) with a 90% probability to aid towards the limitations of uncertainty. This will be achieved mainly though calculation of values for habitat types and attaching them to maps of climate change of the New Forest National Park. Hence the objective of this chapter will be to examine the impact of climate change within the protected area (the Special Area of

Conservation) within the New Forest National Park through the impact of climate on habitat.

5.3. Materials and methods

Study area

The New Forest National Park, in the south of England, UK, was designated as a National Park in 2005 and is a mostly unenclosed tract of mixed habitats including forest, heath and grassland with a commoning system of grazing rights (Grant and Edwards, 2008). As demonstrated in Chapter 4, the Special Site of Scientific Interest (SSSI) of the New Forest National Park was found to have a higher provision of ecosystem services. The New Forest is also designated as a Special Area of Conservation (SAC), Special Protection Area (SPA) and includes RAMSAR sites. The SAC, with an extent of over 29,000 ha, is designated for eleven Annex I habitats included as primary reasons for site selection, including North Atlantic wet heaths, old acidophilous oak woods and Atlantic acidophilous beech forests (JNCC, 2010).

Future climate map of the New Forest National Park

Future climate of the New Forest was modelled by Evans (2017) using the forest dynamic model 'LANDIS-II' (Scheller et al., 2007). LANDIS-II works by tracking the spatial distribution of tree and shrub species over large spatial and temporal spectrums (Scheller et al., 2007). Evans (2017) used precipitation and temperature data from the UK Climate Projections or UKCP09 (probabilistic projections of climate change) (Murphy et al., 2009) as inputs for low, medium and high emissions scenarios, at a 90% probability, meaning that the climate change values were very unlikely to be above the projected levels. Only woodland and woody shrub species were modelled, and only within the Special Area of Conservation of the New Forest National Park. The output from the model received from Evans (2017) were maps of 800 by 800 pixels, each pixel representing 50 m x 50 m, as pre-classified maps of dominant vegetation, defined by biomass (Scheller et al., 2007). Two classification types were used, National Vegetation Classification (NVC) categories (W1, W2, W8, W9, W10, W12, W12.1, W13, W14, W15, W16, W18, W21, W22 and W2 – each given a unique value for raster calculations between 1 and 14) and additionally as a 'non-native or heath' classification raster, as 'img' raster maps for each low, medium and high, emission scenarios, modelled at every 10 years between 2016 – 2116, with three replicates for each (Figure 5.3.1). The 'non-native or heath' reclassification raster was defined by Evans (2017) and was designed to cover the dominant non-native species that the NVC does not cover, and to a lesser extent heathland. As the NVC only deals with 'true woodland' communities, it was necessary to use the extra 'non-native woodland and heath' reclassification raster to add further communities of homogeneous non-native species and heath from Evans (2017). Any remaining value as ‘0’ was assumed to be grassland (no distinction on the type can be made), as these are ‘active’ areas within the model, they just lack colonisation from tree and woody shrub species. No other habitat types (for example bog) could be classified due to the limited species LANDIS-II models. An additional ‘eco-regions’ raster of both active and non-active pixels within the SAC (the latter being sea shore habitats, urban-built-areas, quarries and bare land, and arable horticulture) was provided by Evans (2017).

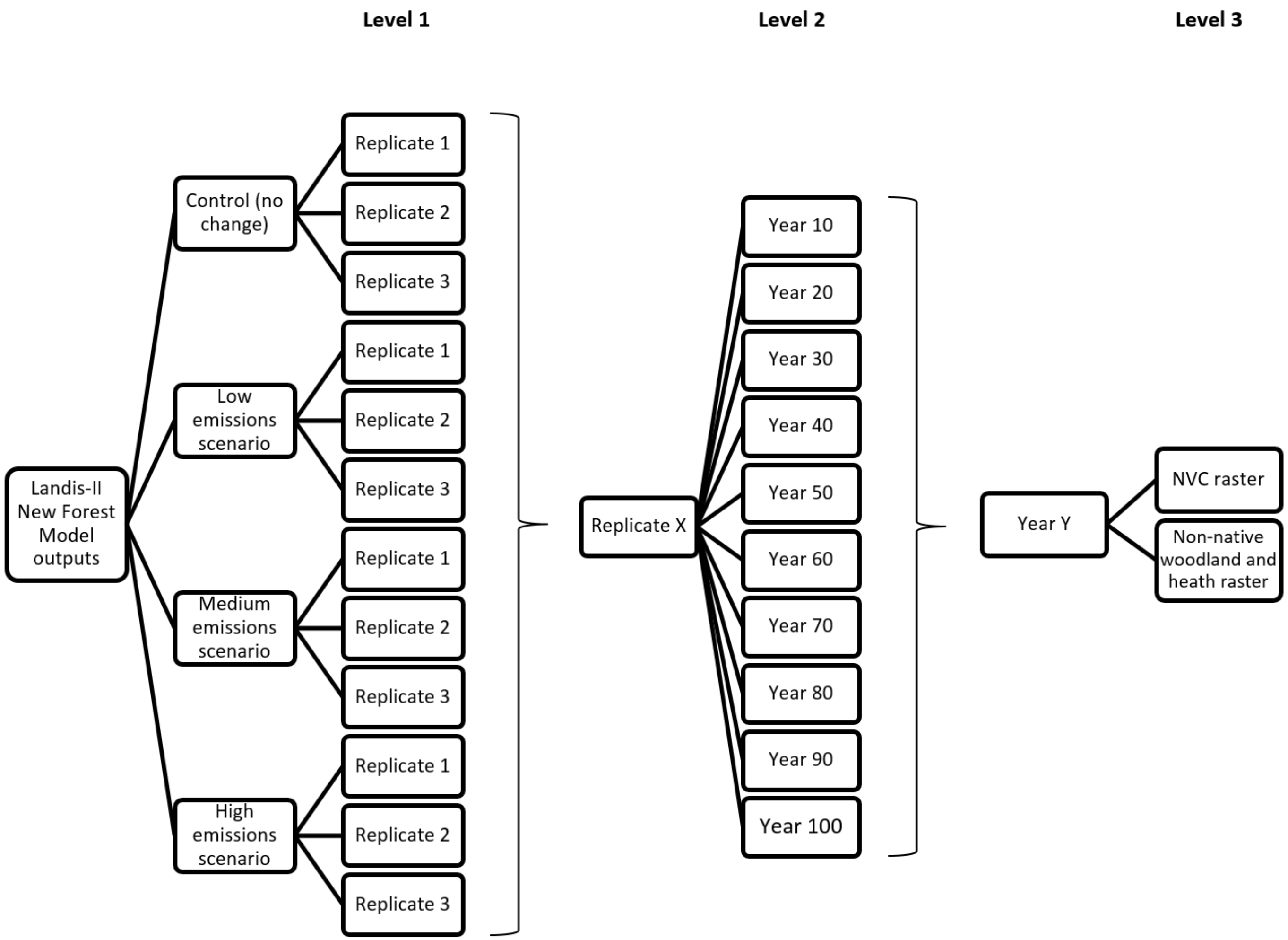

Figure 5.3.1: Data diagram of outputs obtained from Landis-II New Forest Model from Evans (2017), totalling 120 NVC and 120 Non-native woodland and heath rasters used to produce the climate change maps. 'Replicate X' refers to the replicate from level 1, and 'Year Y' refers to the year timestep from level 2.

The same process was followed for every timestep in every replication, for each emission scenario. The 'non-native and heath' rasters were first reclassified to three BAP broad habitat types (dwarf shrub, coniferous woodland and broadleaved woodland) to match the HBIC map using the 'Reclassify' tool. Dwarf Shrub Heath included *Calluna vulgaris* or *Ulex europaeus* dominated areas, Coniferous woodland included Larch (*Larix decidua*), Grand Fir (*Abies grandis*) and Douglas-fir (*Pseudotsuga menziesii*), Sitka spruce (*Picea sitchensis*), Norway spruce (*Picea abies*), Ponderosa Pine (*Pinus ponderosa*), Black pine (*Pinus nigra*), Scots pine (*Pinus sylvestris*), Coast redwood (*Sequoia sempervirens*), Western Hemlock (*Tsuga heterophylla*) and Lawson cypress (*Chamaecyparis lawsoniana*)) dominated areas and 'Broadleaved, mixed and yew woodland' included Sweet chestnut (*Castanea sativa*), Hornbeam (*Carpinus betulus*) or Sycamore (*Acer pseudoplatanus*) dominated areas. Unique values were given to each habitat type (Dwarf Shrub Heath being 100, coniferous woodland being 200 and broadleaved, mixed and yew being 300, these values were arbitrary and used for ease of raster calculation further in the analysis). This raster was then summed with the NVC raster using the 'Raster calculator' tool.

A new raster was then created by using the 'reclassify tool'. The NVC was used as the priority layer, as it shows native woodlands, and the 'non-native and health' raster used to inform areas that were blank. As the NVC raster was coded using integer values between 1 and 14 (these values are arbitrary, and correspond to the 14 NVC communities discussed earlier), to preserve the priority order of NVC, the general rule applied was any values between 1 and 14, 101 – 114, 201 - 214, 301 -314 were classified as dwarf shrub heath, coniferous or broadleaved, mixed and yew woodland on the NVC classification. The exceptions were for NVC categories W21 *Crataegus monogyna – Hedera* helix scrub and W22 Prunus spinosa – *Pteridium aquilinum scrub*, whereby the 'non-native woodland and heath' raster classification was used to guide the final classification, as these two categories as dominant are not conclusively indicative of coniferous or broadleaved habitats, but are more aligned with our category of dwarf shrub heath. NVC W21 and W22 are scrub communities dominated by hawthorn and blackthorn respectively (Hall et al., 2004), where other classifications are dominant for heath, conifer or broadleaved species, they were classified as such. Where NVC W21 and W22 were present only by themselves, with no information from the non-native woodland and heath raster, they were reclassified as being dwarf shrub heath as these are 'shrubby' communities according to the NVC. Full details of classifying the difference combination of NVC and non-native and heath rasters can be found in Appendix III.

The maps were re-classified with arbitrary numbers for ease of raster calculation, with grassland (all '0' value regions within the SAC) given a value of 1, dwarf shrub heath a value of 10, coniferous woodland a value of 100 and broadleaved, mixed and yew woodland a value of 1000. To get a final map of each time step, for each emissions scenario, all three rasters from each replicate were summed together using the 'Raster calculator' tool, and common areas identified as values with 3 for grassland, 30 for dwarf shrub heath, 300 for coniferous woodland and 3000 for broadleaved, mixed and yew woodland. Any other values (e.g., 21, 201 or 1111) showed non-commonality between rasters, these varied between 1,474 km^2 and 2,884 km^2, with greater values the further into the future the model progressed. These non-common areas were treated as grassland, as any non-classified area is assumed to be grassland, as coniferous, broadleaved, and heathland species are not dominating Evans (2017). New rasters were created using the 'reclassify' tool, extracting the four habitat types common across all three replicates. For each scenario, areas that had the same predicted habitat under each of the three replicates were chosen to create final maps.

The LANDIS-II outputs lacked a co-ordinate or projection system. Hence the final maps were moved and rescaled using the 'Shift' and 'Rescale' tools using the lower left-hand corner coordinates of -409995.5 and 85804.5 (Evans, 2017) with a rescale factor of 50 and the 'Define projection' tool used to add British National Grid projection added. Pixel counts for each habitat type, including non-common pixels, were extracted and km^2 of each habitat per timestep calculated (by multiplying each pixel by 50) (see Results, Table 5.4.1).

The 'Resample' tool was used to create 25 m by 25 m rasters from the 50m x 50 m rasters, using the 'nearest' method. This allowed the final maps to be directly comparable to other ecosystem services already modelled in Chapter 4 at resolution of 25 m by 25 m. The 'eco-regions' raster containing the active cells that were modelled was reclassified so that only active pixels were given a value of 1, and all other pixels (non-SAC, or not active pixels) given a value of 0. The final rasters were clipped using the 'extract by mask' tool, leaving final maps of habitat for further analysis.

Ecosystem service values

Ecosystem services values were modelled using a benefits transfer approach, project the current value of the ecosystem service into the future, with only the quantity of landcover being considered. The limitation in using this technique is the lack of consideration of other factors that will affect the provision of ecosystem services, including the underpinning ecological processes, temperature, and supply and demand amongst others.

Recreation values

Recreation values were available from Chapter 2 as fixed sum survey questions and participatory GIS (pGIS) mapping values and Chapter 3 as GPS tracked values. The recreation data from Chapter 4's InVEST: Visitation: Recreation and Tourism model was not used, as the pGIS and GPS included many more sample points and the model predicted tourism and recreation based on photographs (hence may be confounded by aesthetic values).

Firstly, the normalised values were extracted from Chapter 2 for the pGIS maps and fixed-sum values for each HBIC habitat classification and Chapter 3 for the normalised GPS values for each HBIC habitat type. As grassland habitat is general and not specific in the climate change maps in this chapter, all grassland types present in the SAC areas were averaged. The HBIC raster map was clipped using the 'extract by mask' tool in ArcMap and the presence of grassland habitats checked, with all being present apart from calcareous grassland. Values for broadleaved, mixed and yew, coniferous and dwarf shrub heath were already available as independent categories (Table 5.3.1). Values from Chapter 3 were extracted from a normalised score from how long people spent in a habitat compared to its availability. The 5000m buffer was chosen as this encompassed all the GPS tracks recorded in the Chapter 2 study (Table 5.3.2). Both Chapter 2 and 3 recreation normalised values were averaged to produce a final recreation value for each habitat type in this chapter (Table 5.3.3).

Table 5.3.1: Recreation values from fixed sum scoring questions for habitat types (from Chapter 2).

Habitat type	Recreation value	Normalised score
Acid grassland	9	0.45
Arable cereals	2.5	0.125
Arable horticulture	4	0.2
Broadleaved / mixed woodland	20	1
Coniferous woodland	10	0.5
Dwarf shrub heath	15	0.75
Fen/ marsh/ swamp	10	0.5
Improved grassland	5	0.25
Neutral grassland	10	0.5
Suburban / rural developed	2	0.1
Urban	0	0

Table 5.3.2: Normalised values of recreation from difference in time spent in habitat (%) compared to the habitat available (%) from Chapter 3 GPS data.

BAP Broad habitat	Difference (%)	Normalised value
Acid grassland	207.33	0.67
Boundary and linear features	354.46	1.00
Bracken	-75.94	0.04
Broadleaved, mixed, and yew woodland	-28.1	0.15
Built-up areas and gardens	72.47	0.37
Coniferous woodland	-11.83	0.18
Dwarf shrub heath	63.29	0.35
Fen, marsh and swamp	-21.52	0.16
Improved grassland	-94.18	0.00
Neutral grassland	-36.71	0.13
Unidentified habitat	-57.75	0.08
Water (River and streams, standing open water and canals, unidentified water)	-59.9	0.08

Table 5.3.3: Normalised recreation values for the New Forest National Park. [1]Grassland values are averages for acid, neutral and improved grassland.

Broad habitat type	Fixed sum value	pGIS value	GPS value	Mean value
Grassland[1]	0.400	0.191	0.267	0.286
Dwarf shrub heath	0.750	0.106	0.350	0.402
Coniferous woodland	0.500	0.128	0.180	0.269
Broadleaved, mixed and yew woodland	1.000	0.115	0.150	0.422

Aesthetic value

Aesthetic values were available from Chapter 2 as fixed sum survey questions and participatory GIS (pGIS) mapping values. The aesthetic data from Chapter 3 was not available for all habitat types (limited to woodland) so not used. The normalised values were extracted from Chapter 2 for the pGIS maps and fixed-sum values, with the latter being normalised (Table 5.3.4) for each HBIC habitat classification and the final aesthetic values averaged (Table 5.3.5)

Table 5.3.4: Aesthetic values from fixed sum questions by habitat type (from Chapter 2).

Habitat type	Aesthetic value	Normalised score
Acid grassland	5.5	0.275
Arable cereals	0	0
Arable horticulture	0	0
Broadleaved / mixed woodland	20	1
Coniferous woodland	15	0.75
Dwarf shrub heath	10	0.5
Fen/ marsh/ swamp	5	0.25
Improved grassland	10	0.5
Neutral grassland	10	0.5
Suburban / rural developed	4	0.2
Urban	0	0

Table 5.3.5: Normalised aesthetic values for the New Forest National Park. [1]Grassland values are averages for acid, neutral and improved grassland.

Broad habitat type	Fixed sum value	pGIS value	Mean value
Grassland[1]	0.425	0.247	0.336
Dwarf shrub heath	0.500	0.185	0.342
Coniferous woodland	0.750	0.218	0.484
Broadleaved, mixed and yew woodland	1.000	0.210	0.605

Biodiversity values

The property value, biodiversity, tranquillity, water yield, nitrogen retention and phosphorous retention maps from Chapter 4, covering the full extent of the New Forest National Park were used. They were first normalised by using raster calculator using a unity-based normalisation. Points at 25 m intervals were created using the 'Fishnet' tool, and then clipped to the extent of the New Forest National park boundary, totalling 911,318 points. The 'Extract values to Points' tools was used to extract and append the habitat type from the HBIC raster to the points in a new shapefile.

The 'Sample' (spatial analyst) tool was used to extract the values to point, using the 'nearest' resampling method and all point imported into SPSS. The mean value for every habitat type was calculated in SPSS for every habitat calculated (Table 5.3.6) and the mean value calculated for grassland (Table 5.3.7).

Table 5.3.6: Biodiversity values for broad habitats in the New Forest National Park.

Broad habitat type	Biodiversity
Bracken	0.161
Arable and horticulture	0.000
Improved grassland	0.057
Neutral grassland	0.287
Acid grassland	0.243
Standing open water and canals	0.357
Fen, marsh and swamp	0.243
Rivers and streams	0.148
Unidentified water	0.131
Calcareous grassland	0.653
Littoral Sediment	0.376
Dwarf shrub heath	0.277
Inland rock	0.158
Supralittoral Sediment	0.544
Inshore sublittoral sediment	0.899
Littoral Rock	1.000
Broadleaved, mixed, and yew woodland	0.153
Coniferous woodland	0.183
Unidentified habitat	0.211
Boundary and linear features	0.414
Built-up areas and gardens	0.317
Supralittoral Rock	0.443

Table 5.3.7: Biodiversity values for the New Forest National Park from Chapter 4. [1]Grassland values are averages for acid, neutral and improved grassland.

Broad habitat type	Biodiversity
Grassland[1]	0.195
Dwarf shrub heath	0.277
Coniferous woodland	0.183
Broadleaved, mixed and yew woodland	0.153

Flood risk mitigation, pollinator habitat quality, livestock and timber values

Flood risk mitigation, pollinator habitat quality, livestock and timber values were calculated for HBIC broad habitat types in Chapter 4, with grassland values being averaged as a mean value (Table 5.3.8).

Pollinator habitat quality was taken from mean values for nectar productivity, species nectar diversity and functional diversity (Chapter 4).

Table 5.3.8: Flood risk mitigation, pollinator habitat quality, livestock and timber values from Chapter 4.

Broad habitat type	Flood risk mitigation	Pollinator habitat quality	Livestock	Timber values
Grassland[1]	0.777	0.560	0.530	0.000
Dwarf shrub heath	0.670	0.600	0.000	0.000
Coniferous woodland	1.000	0.540	0.000	1.000
Broadleaved, mixed and yew woodland	1.000	0.730	0.000	0.826

Ecosystem service provision graphs

All ES values are the mean from rasters at a 25 m x 25 m resolution, to be able to compute the provision per habitat for the climate change maps, the ES values were multiplied by 4 to match the climate change resolution of 50 m by 50 m before being graphed. The number of pixels per habitat were multiplied by the ES value for each of the four habitat types, and summed together for each time step in each scenario. For each ecosystem service, each emissions scenario was tested for normality using the Shapiro-Wilks test (Table 5.3.9). Only pollinator habitat quality (medium), tranquillity (control), biodiversity (medium, high and control) and nitrogen retention (low, medium and high) had data that deviated from a normal distribution.

Table 5.3.9: Normality tests for all emissions scenarios and ecosystem services, tests show either significance (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significance (n.s.) using Shapiro-Wilk.

Ecosystem service/ emission scenario	Low	Medium	High	Control
Flood risk mitigation	0.916 (n.s.)	0.954 (n.s.)	0.907 (n.s.)	0.867 (n.s.)
Pollinator habitat quality	0.872 (n.s.)	0.855*	0.883 (n.s.)	0.876 (n.s.)
Livestock	0.939 (n.s.)	0.931 (n.s.)	0.931 (n.s.)	0.862 (n.s.)
Timber values	0.951 (n.s.)	0.961 (n.s.)	0.939 (n.s.)	0.946 (n.s.)
Biodiversity	0.906 (n.s.)	0.837*	0.815*	0.851*
Recreation	0.936 (n.s.)	0.929 (n.s.)	0.931 (n.s.)	0.859 (n.s.)
Aesthetic	0.950 (n.s.)	0.956 (n.s.)	0.923 (n.s.)	0.860 (n.s.)

Spatial Multi-Criteria Analysis

Spatial Multi-Criteria Analysis is a tool that is often used in decision making to allow different scenarios to be compared using multiple criteria (this analysis uses ecosystem services as spatial factors or criteria), without any ‘costs’ in this analysis, the process is similar to normalising, weighting and combining the various ecosystem services. The Spatial Multi-Criteria Analysis was conducted using ILWIS v 3.8 (North 52 GmbH, Germany) for 2116(the limit of the model) for low, medium, high and the control scenarios. All ecosystem service values for grassland, dwarf shrub heath, coniferous woodland and broadleaved, mixed and yew woodland were saved in a comma separated file (CSV) and ‘joined’ in

ArcMap to the 2116 low, medium, high and control rasters. The ‘Look-up’ tool was used to create new rasters from each ecosystem service.

Files were exported into ASCII format using the ‘Raster to ASCII’ tool in ArcMap. The ASCII files were then imported in ILWIS and all georeferenced against the same file (the control 100 year aesthetic value map), as all maps were the exact same size in terms of resolution, columns and rows and projected to British National Grid, any file could have been used. A new ‘Decision tree’ for a ‘Decision making’ situation created, as this allowed several alternative scenarios to be explored, despite looking at climate change scenarios and not management ‘decisions’. Four alternatives were added to the decision tree to correspond to alternative climate change scenarios; control, low, medium and high. Spatial factors were added, in this case all the ecosystem services, and standardised using an interval method (thus all services between 0 and 1). A direct method for the spatial multi-criteria analysis (or Spatial Multi-Criteria Evaluation SMCE) was chosen, allowing an equal weighting assigned to each ecosystem service. Finally, the resultant maps were exported into ArcMap for further visualisation.

The limitations of modelling the ES using proxy data and the ‘benefit-transfer method’ for future scenarios should not be ignored or taken lightly, as false spatial correlations have been found when applying ES values from smaller areas to larger scales. Though more advanced modelling for each ES was outside of the scale of this study, and this approach has been used to provide an indication of ES change (Eigenbrod et al., 2009, Eigenbrod et al., 2010). It should be noted that factors such as recreation preference change for habitats with a changing climate, or a change in value for timber in the future scenarios, and individual factors that would affect ES values have not been taken into account in this study either. The approach used here simply projects the currect values for the New Forest and applies them for the habitats present in the New Forest in the modelled projections. It is hoped this work will be the intial starting point for future research in the area.

5.4. Results

Results show that there was variability in the non-common area for all emissions scenarios (Table 5.4.1). The general trend infers the greater the year, the larger the non-common areas present in the timestep, reflective of the uncertainty of projecting further into the future. In the final 2116 map the low, medium and high emissions scenarios show similar non-common land of 2884 km^2, 2706 km^2, 2637 km^2 respectively, with approximately 1000 km^2 less in the control with 1646 km^2. Interestingly, a spike at year 2036 can be seen across all scenarios.

The low emissions scenario shows an increase in grassland area of 30% between 2016 and 2116. This increase rises for the medium and high emissions scenarios to 50% and 48%. All other habitat types; dwarf shrub heath, coniferous woodland and broadleaved, mixed and yew woodland, show reductions of area for the low, medium and high emissions scenarios. Dwarf shrub heath shows a reduction of 14% for the low emissions scenario, with a much higher reduction of 76% and 73% for medium and high emissions scenarios respectively. Coniferous woodland shows high reductions for low, medium and high emissions scenarios of 63%, 79% and 65%. Broadleaved, mixed and yew woodland have a 33% reduction in the low emissions scenario, with a higher 37% reduction for the medium and high scenarios.

The control scenario using historical climate data was the only scenario showing a decrease in grassland of 42%. Conversely all other habitat types show increases in land area, with dwarf shrub heath increasing 150%, coniferous woodland 45% and broadleaved, mixed and yew woodland increasing just 1%.

Table 5.4.1: Habitat area (km^2) by emission scenario and timestep in the New Forest SAC. Timestep 0 is modelled using actual historical data and was used to allow it to be comparable to the other timesteps. Non-common areas were added to the grassland classification

Low emissions scenario						
		Area (km^2)				
Timestep	Year	Non-common to all three replicates	Grassland	Dwarf shrub heath	Coniferous woodland	Broadleaved, mixed and yew woodland
0	**2016**	0.0	2884.2	725.8	220	1921.1
10	**2026**	1637.3	2972.1	671.4	184	1923.7
20	**2036**	2044.7	2382.2	1339.5	187.2	1842.3
30	**2046**	1473.5	1631.6	2198.4	182.4	1738.8
40	**2056**	1196.4	1300.8	2623.9	164.4	1662.1
50	**2066**	1253.1	1362.0	2657.0	144.5	1587.7
60	**2076**	1666.8	1808.9	2280.7	120.1	1541.5
70	**2086**	2315.2	2565.9	1596.8	99.0	1489.5
80	**2096**	2714.8	3180.0	1047.4	105.5	1418.3
90	**2106**	2795.1	3490.5	784.8	100.5	1375.3
100	**2116**	2884.1	3751.5	624.4	82.6	1292.7

Medium emissions scenario						
		Area (km^2)				
Timestep	Year	Non-common to all three replicates	Grassland	Dwarf shrub heath	Coniferous woodland	Broadleaved, mixed and yew woodland
0	**2016**	0.0	2884.2	725.8	220.0	1921.1
10	**2026**	1721.6	3110.4	665.4	173.9	1801.5
20	**2036**	2221.9	2616.3	1209.3	180.4	1745.2
30	**2046**	1781.0	1985.9	1966.6	172.6	1626.2
40	**2056**	1695.1	1875.2	2226.6	157.9	1491.4
50	**2066**	1936.5	2160.9	1997.2	156.3	1436.8
60	**2076**	2608.3	2999.4	1222.3	137.6	1391.9
70	**2086**	2755.3	3739.4	558.7	116.0	1337.1
80	**2096**	2802.4	3955.9	401.5	90.4	1303.4
90	**2106**	2794.7	4181.9	235.5	70.8	1263.0
100	**2116**	2705.8	4316.9	180.5	46.3	1207.5

High emissions scenario

		Area (km²)				
Timestep	**Year**	**Non-common to all three replicates**	**Grassland**	**Dwarf shrub heath**	**Coniferous woodland**	**Broadleaved, mixed and yew woodland**
0	**2016**	0.0	2884.2	725.8	220.0	1921.1
10	**2026**	1611.8	2953.0	674.3	174.5	1949.4
20	**2036**	2071.8	2455.9	1258.0	178.2	1859.1
30	**2046**	1678.6	1911.2	1866.6	164.2	1809.3
40	**2056**	1701.3	1948.7	1964.2	160.2	1678.1
50	**2066**	2072.5	2409.8	1647.1	144.3	1550.1
60	**2076**	2622.6	3254.6	873.3	132.1	1491.2
70	**2086**	2637.2	3741.7	473.8	107.1	1428.6
80	**2096**	2675.3	3940.8	342.0	118.2	1350.3
90	**2106**	2663.6	4128.2	253.9	92.8	1276.3
100	**2116**	2637.1	4278.8	189.0	76.3	1207.1

Control (using current climate data)

		Area (km²)				
Timestep	**Year**	**Non-common to all three replicates**	**Grassland**	**Dwarf shrub heath**	**Coniferous woodland**	**Broadleaved, mixed and yew woodland**
0	**2016**	0.0	2884.2	725.8	220.0	1921.1
10	**2026**	1626.1	2726.2	677.5	185.0	2162.4
20	**2036**	2226.8	2876.1	663.3	204.35	2007.4
30	**2046**	1987.6	2384.1	1143.1	233.0	1990.9
40	**2056**	1907.6	2129.3	1429.2	207.1	1985.6
50	**2066**	1724.0	1869.4	1696.1	199.0	1986.7
60	**2076**	1954.2	2053.6	1496.8	190.4	2010.4
70	**2086**	1694.3	1755.9	1790.9	190.1	2014.3
80	**2096**	1818.9	1861.0	1631.6	253.1	2005.5
90	**2106**	1666.8	1698.9	1790.0	273.9	1988.5
100	**2116**	1654.5	1681.7	1816.8	318.8	1933.9

Many ecosystem services show interesting predicted dynamics over time (Figures 5.4.1-5.4.7). Flood risk mitigation, livestock, heritage value, property value, nitrogen retention and phosphorous retention all show the same trend for the emissions scenarios of first declining after 2026 until approximately 2056-66, before rising again. Interestingly, the low emissions scenario consistently has the lowest value at 2116 for these named services compared to the medium and high scenarios. Apart from flood risk mitigation, the control scenario for these ecosystem services is actually lower in 2116 than all three emissions scenarios. Tranquillity, biodiversity, recreation and water yield are predicted to increase in value in the short term until approximately 2056-66 before falling in value by 2116. Pollinator habitat quality follows this trend, though not as strongly, with value decreasing after 2046-56, apart from the control scenario increasing over time. Apart from tranquillity, the control scenario for these ecosystem services has a higher value in 2116 than for all three emissions scenarios. Timber and aesthetic value does not show either of the trends of the two ecosystem service groups described, though rather shows a steady decline until 2116 for all three emissions scenarios, with the control scenario rising in 2026 before plateauing.

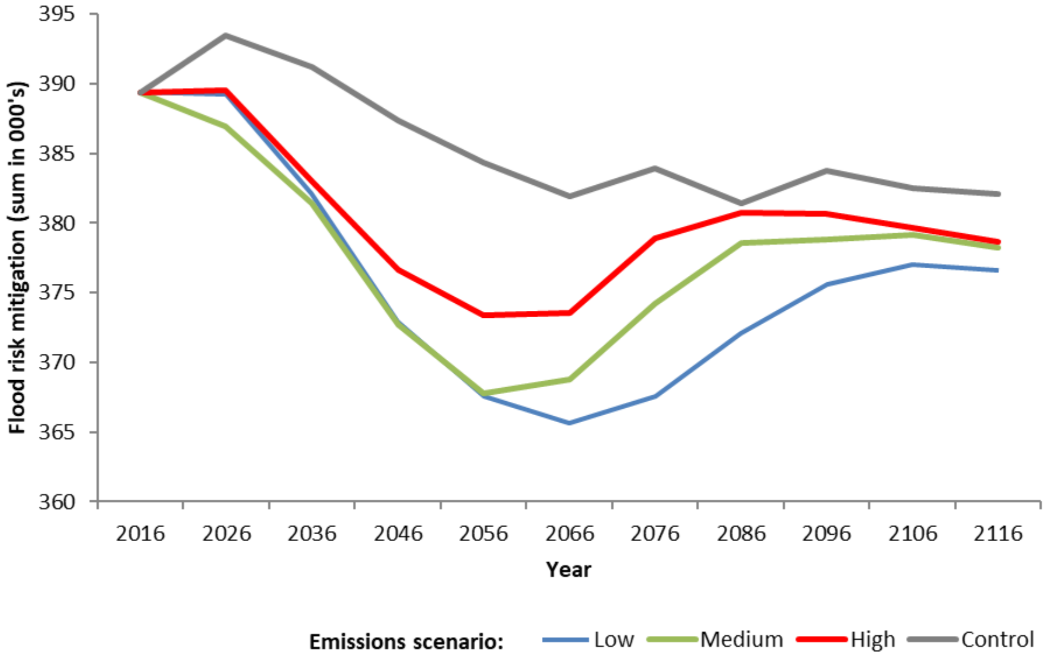

Figure 5.4.1: Graph illustrating the provision of flood risk mitigation for low, medium and high emissions scenarios within the SAC of the New Forest National Park. The control uses current temperature and precipation to model future land classification.

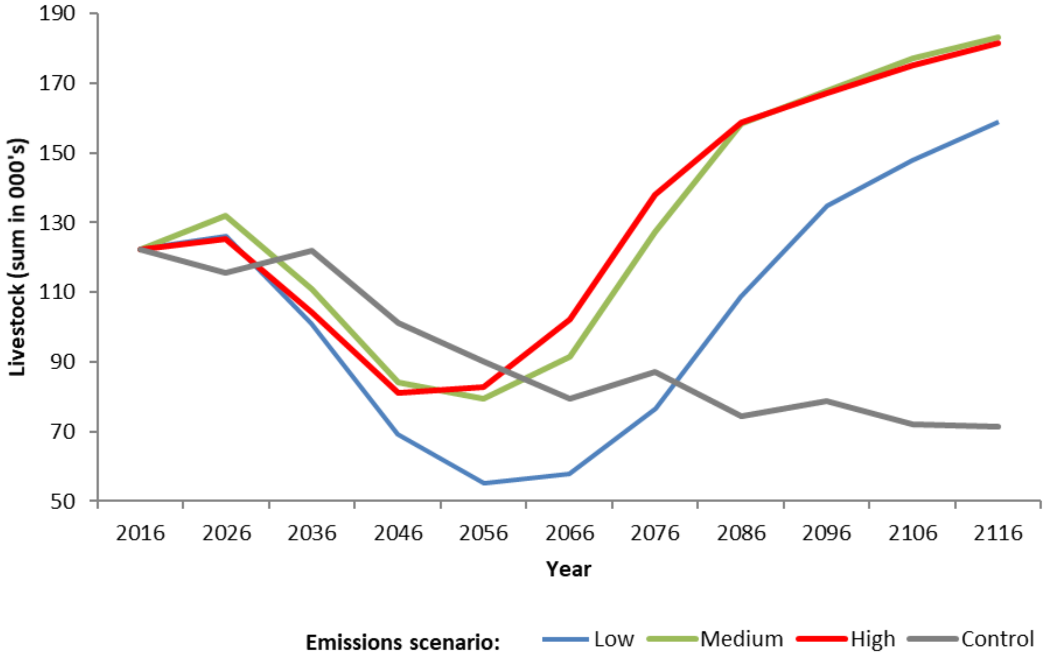

Figure 5.4.2: Graph illustrating the provision of livestock for low, medium and high emissions scenarios within the SAC of the New Forest National Park. The control uses current temperature and precipitation to model future land classification.

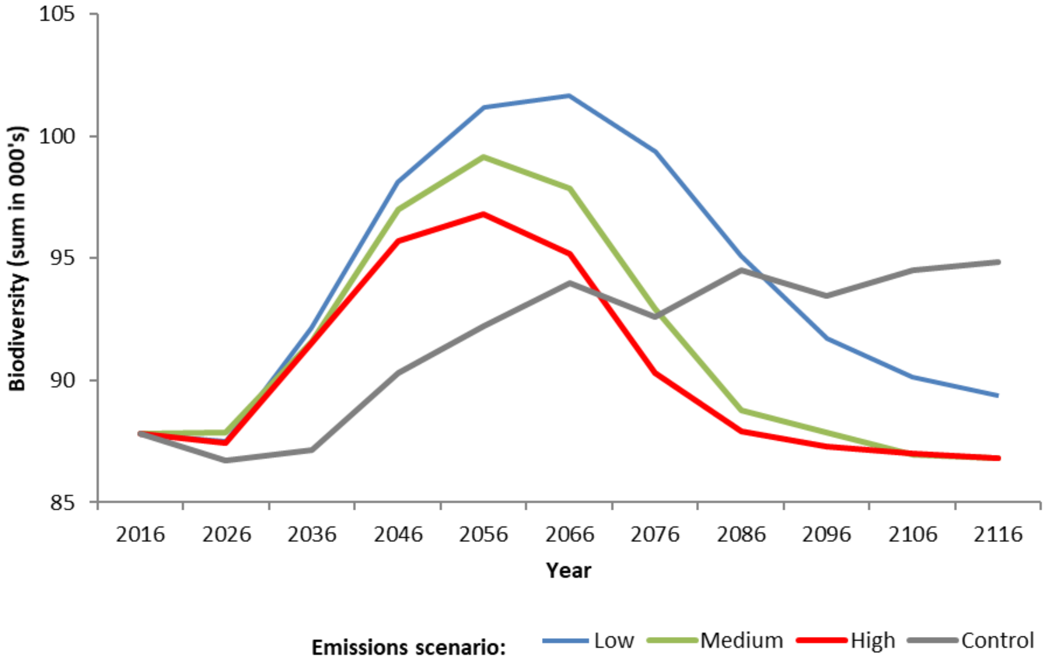

Figure 5.4.3: Graph illustrating the provision of biodiversity for low, medium and high emissions scenarios within the SAC of the New Forest National Park. The control uses current temperature and precipitation to model future land classification.

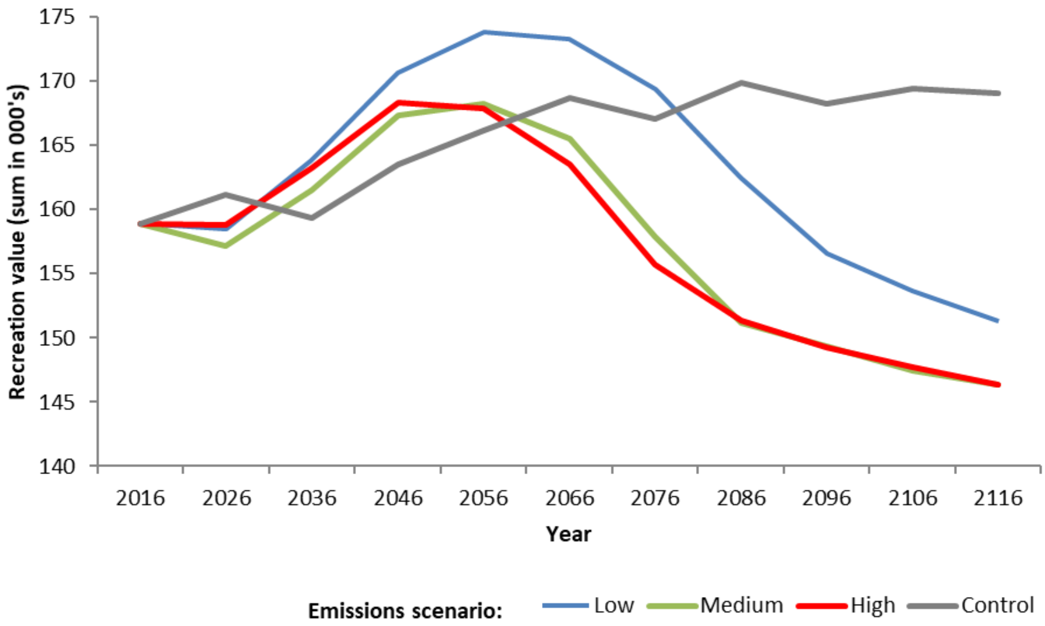

Figure 5.4.4: Graph illustrating the recreation provision for low, medium and high emissions scenarios within the SAC of the New Forest National Park. The control uses current temperature and precipitation to model future land classification.

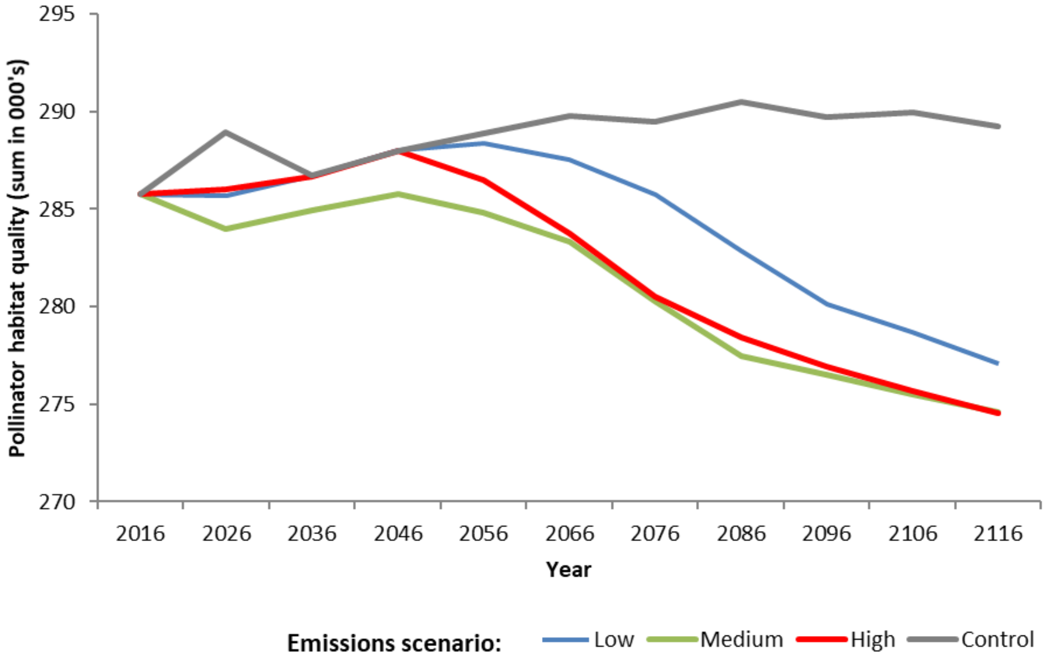

Figure 5.4.5: Graph illustrating the provision of pollinator habitat quality for low, medium and high emissions scenarios within the SAC of the New Forest National Park. The control uses current temperature and precipitation to model future land classification.

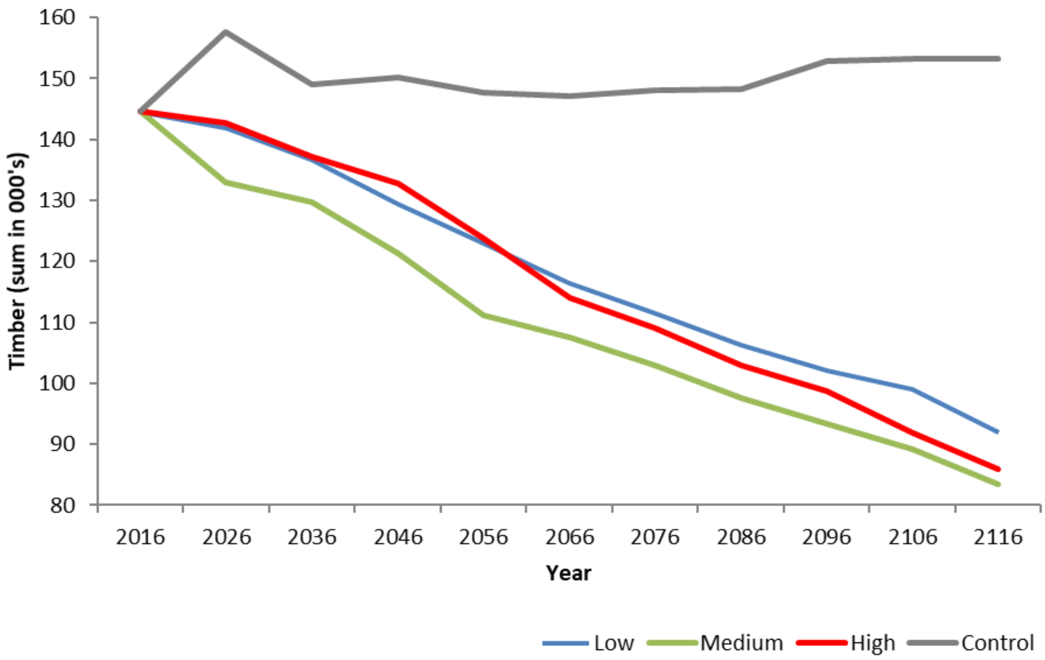

Figure 5.4.6: Graph illustrating the provision of timber for low, medium and high emissions scenarios within the SAC of the New Forest National Park. The control uses current temperature and precipitation to model future land classification.

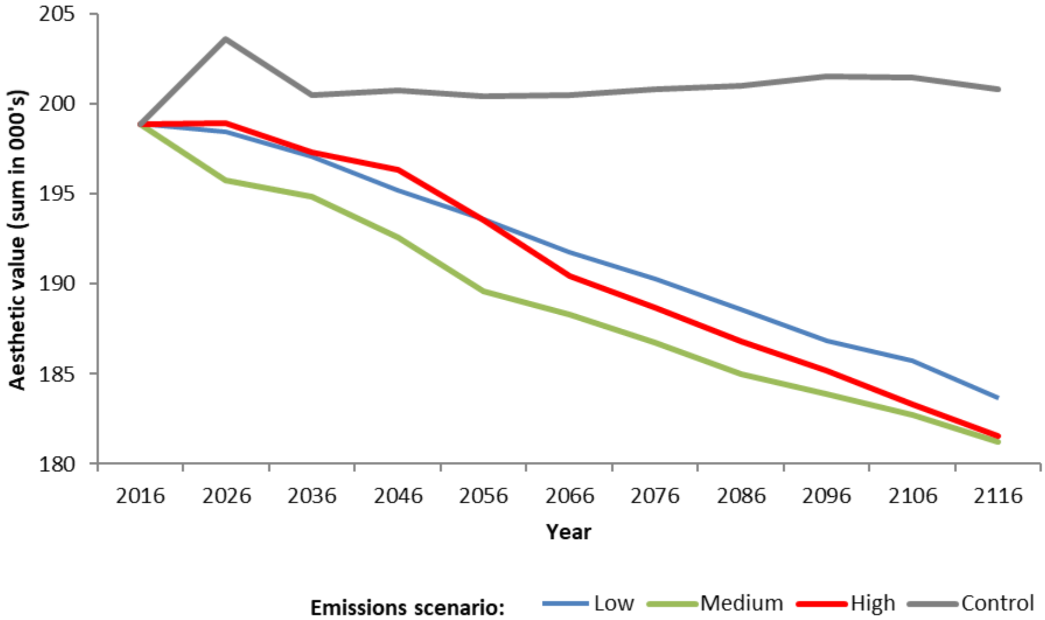

Figure 5.4.7: Graph illustrating the aesthetic provision for low, medium and high emissions scenarios within the SAC of the New Forest National Park. The control uses current temperature and precipitation to model future land classification.

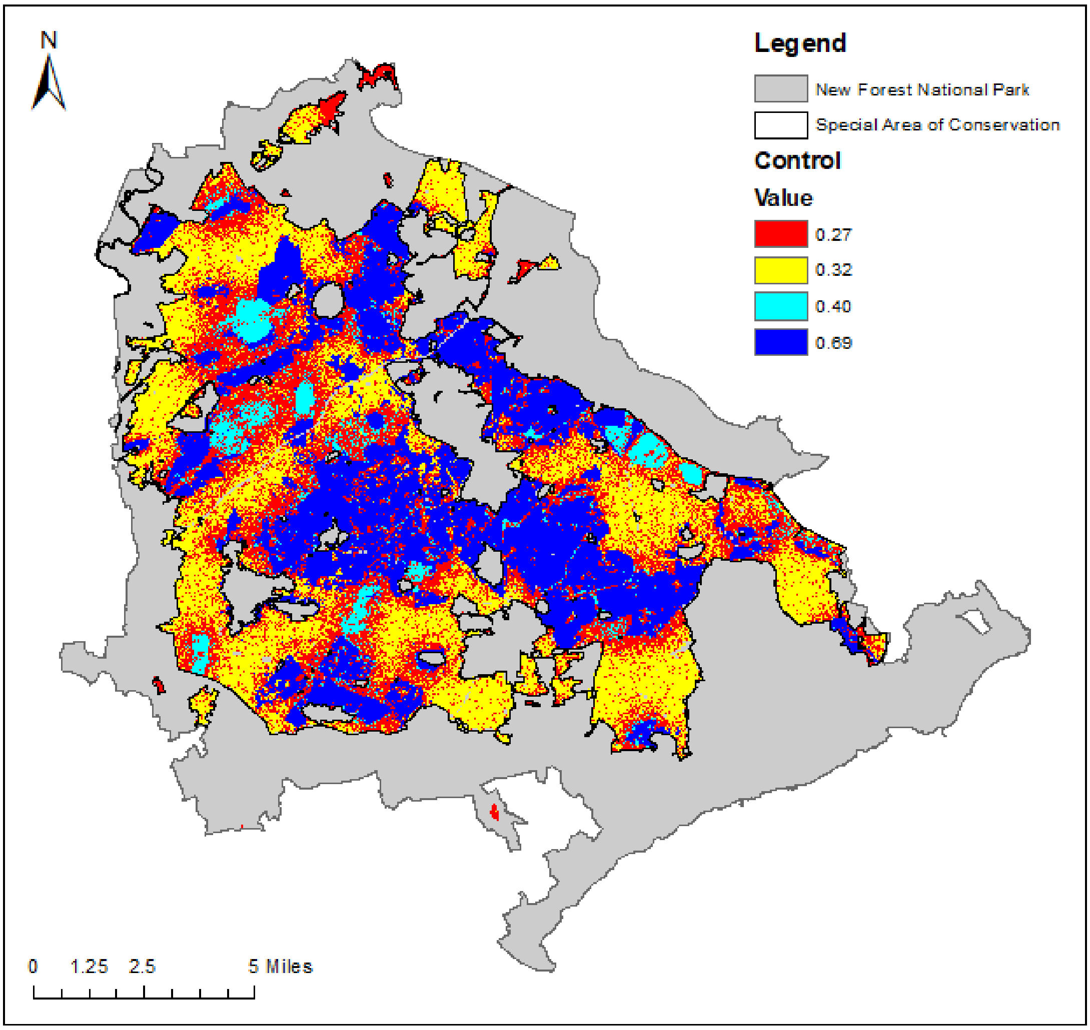

Figure 5.4.8: Equally weighted spatial multi-criteria analysis modelled using historical temperatures (the control scenario) for 2116.

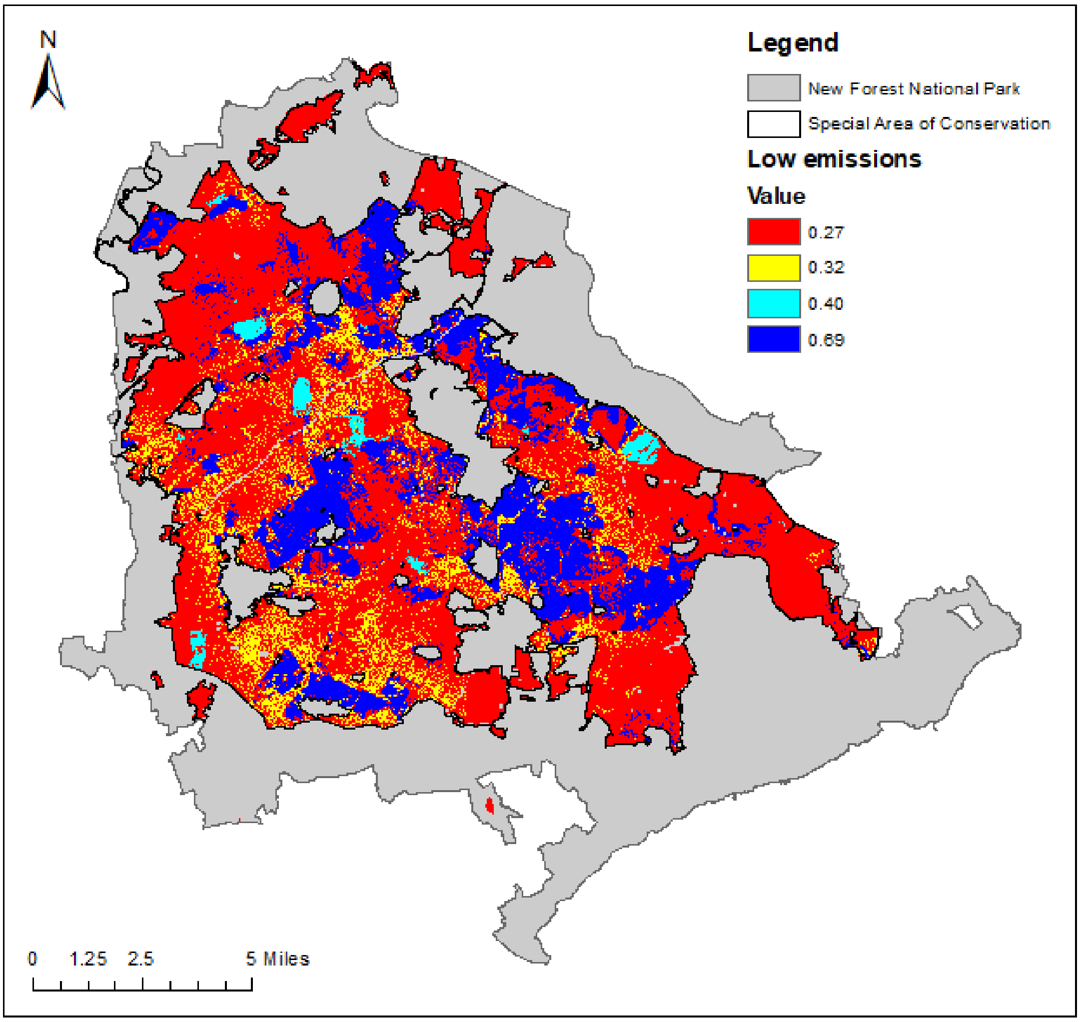

Figure 5.4.9: Equally weighted spatial multi-criteria analysis modelled using the low emissions scenario for 2116.

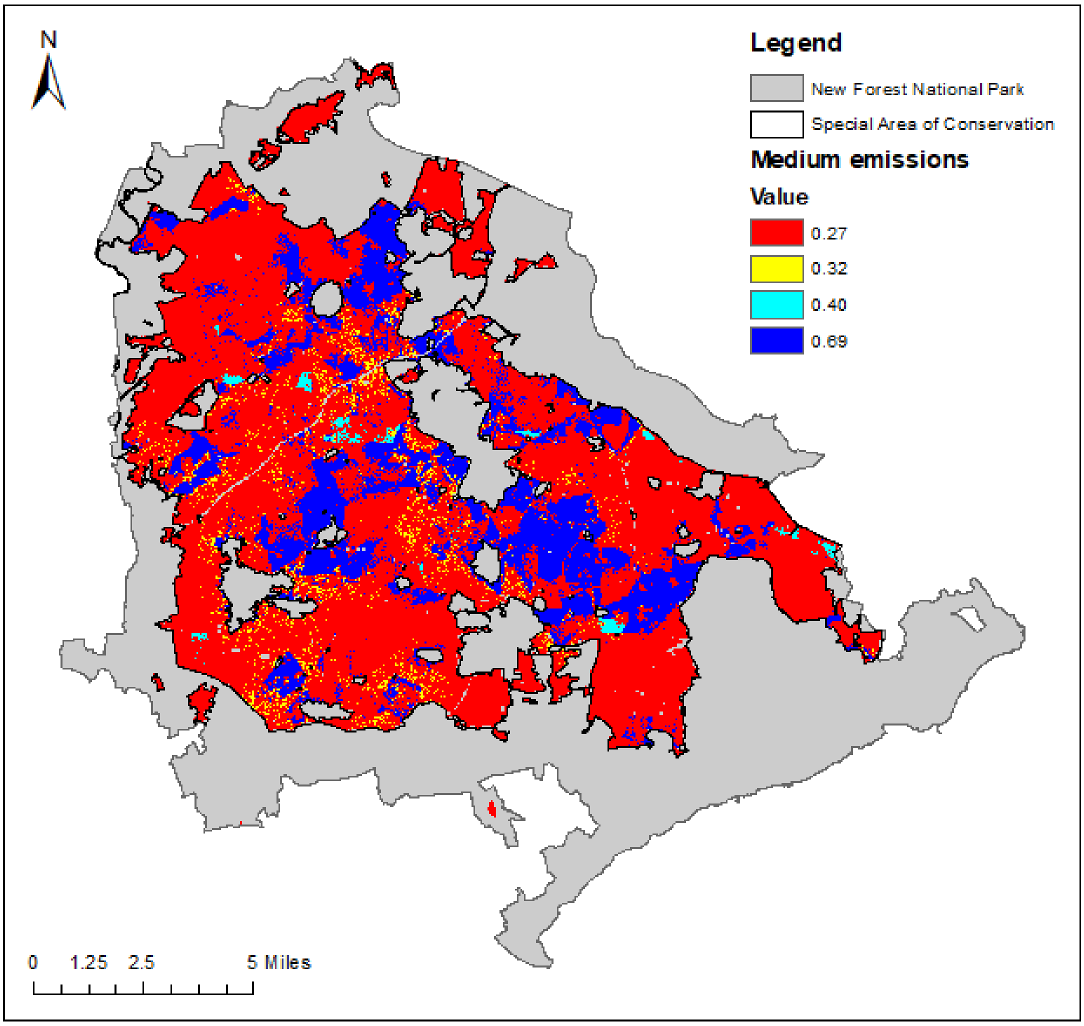

Figure 5.4.10: Equally weighted spatial multi-criteria analysis modelled using the medium emissions scenario for 2116.

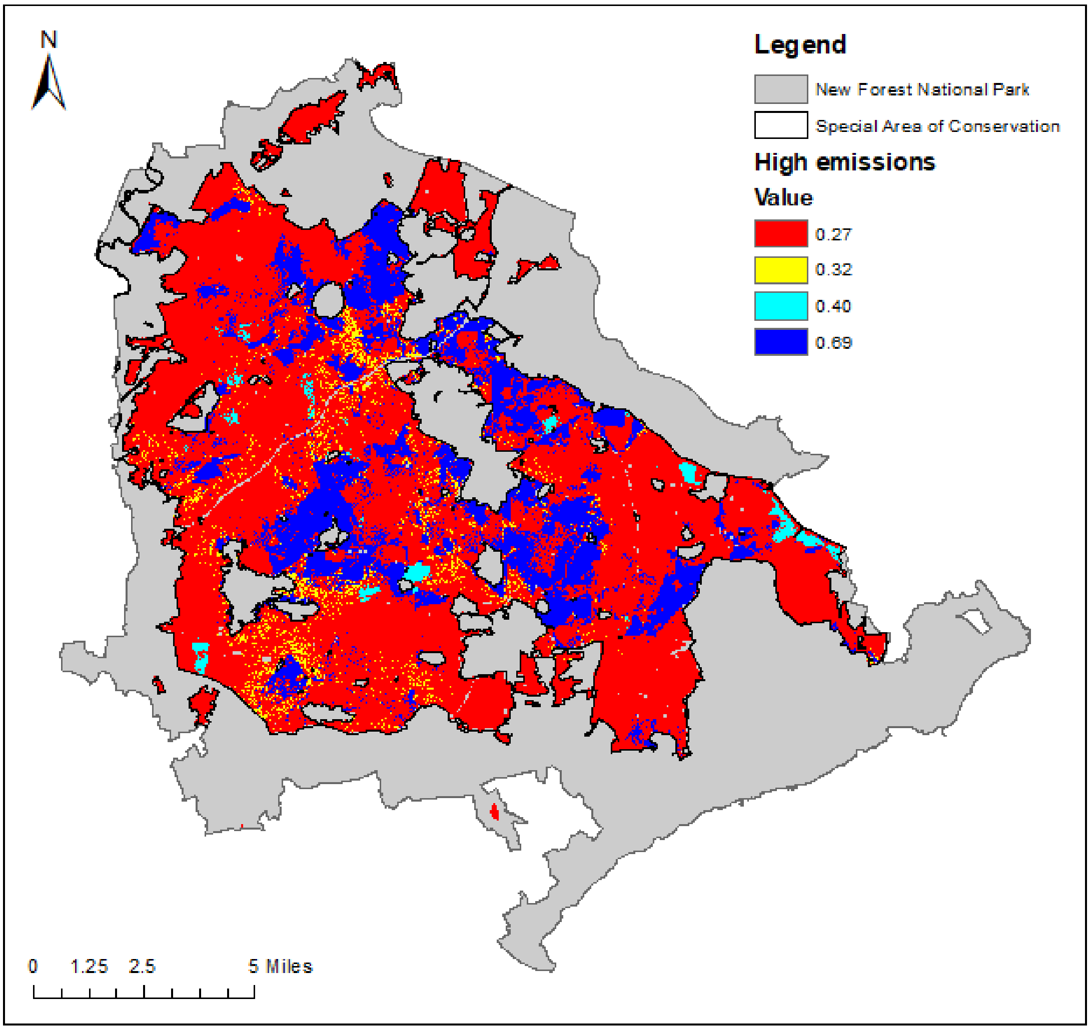

Figure 5.4.11: Equally weighted spatial multi-criteria analysis modelled using the high emissions scenario for 2116.

As all ecosystem services were spatially mapped by habitat type (Figure 5.4.8 to Figure 5.4.11), the control map shows that the higher values of ecosystem services are distributed across the New Forest SAC in clusters, most predominately in the central and northern regions of the SAC. The low, medium and high emissions scenarios show a reduction in the high value areas. The final spatial multi-criteria analysis maps consist of four unique values corresponding to the four habitats; grassland 0.34, dwarf shrub heath 0.36, coniferous woodland 0.44 and broadleaved, mixed and yew woodland with 0.64, see Table 5.4.2 for extracted values from Figure 5.4.8 to Figure 5.4.11). The highest values of ecosystem service are dependent on broadleaved woodland and the area of this shows an 11.15%

decrease in the low emissions scenarios compared to the control scenario, with a further approx 1.5% decrease in ES provision between the low and medium/ high scenarios. Coniferous woodland has the second highest value with 0.44, with the control scenario having 5.54% land area, with the emissions scenarios all showing large decreases of over 4% to 0.80 and 1.44. Grassland and Dwarf shrub heath have a similar value of 0.34 and 0.36 respectivley, though grassland areas increase extensivley, more than doubling from 29.24% in the control scenario to 65.23%, 75.06% and 74.40% for the low, medium and high emissions scenarios respectivley. It must be noted that it may be an aretefact of designating uncertain areas as grassland. Dwarf shrub heath presents a decrease of 31.59% in the control scenario to 10.86% in the low emissions scenario, and further decreasing to 3.14% and 3.29% in the medium and high emissions scenarios.

Table 5.4.2: Percentages area of ecosystem service provision values for 2116 extracted from the spatial multi-criteria analysis maps.

	Grassland	**Dwarf shrub heath**	**Coniferous woodland**	**Broadleaved, mixed and yew**
Value	0.27	0.32	0.40	0.69
% per scenario				
Control	29.24	31.59	5.54	33.63
Low	65.23	10.86	1.44	22.48
Medium	75.06	3.14	0.80	21.00
High	74.40	3.29	1.33	20.99

5.5. Discussion

Climate change seems likely to have a negative impact on the total provision of ecosystem services in the New Forest SAC. The spatial multi-criteria maps demonstrate that whether a low, medium or high emissions scenario becomes a reality, a reduction in ecosystem services will be seen. Although explicit studies regarding the impact of climate change on ecosystem service provision within protected areas are not available, parallels to other studies exist. The Millennium Ecosystem Assessment (2005) suggested there is a likely negative impact of climate change on ecosystem services with a review by Runting et al. (2016a) showing that 59% of studies showed a negative impact of climate change on most types of services. This suggests that ecosystem service provision within the SAC will respond in largely similar ways to their provision in general, so the existence of the SAC will not mitigate the effects of climate change without some sort of intervention for the majority of ecosystem services, though not all will be negatively impacted.

Grassland is the only habitat type to increase with all emissions scenarios, apart from the control. This may be explained through the non-common areas being included as grassland, under the assumption that if it was non-classified regularly as a different habitat type it would be grassland. This limitation could be mitigated through further replicates of the LANDIS-II model. Low, medium and high emissions scenarios all showed reductions in area for dwarf shrub heath, coniferous woodland and broadleaved, mixed and yew woodland, with increases in grassland. Only the control scenario, using historical climate data, showed increases in all but the grassland habitats investigated. The 'lag' that can be seen in the first few decades of each ecosystem service value is indicative of the insensitivity of the climate system to emissions due to factors such as inertia (Murphy et al., 2009)

It has been shown that timber value would be negatively affected by a 2 °C global increase in temperature (Elkin et al., 2013), with the results of my study suggesting that the value of timber would be halved under any of the emissions scenarios. Other studies have shown climate change has a negative impact on biodiversity, with 'alarming' consequences being reported by Bellard et al. (2012), with worst case scenarios leading to the mass extinctions (Runting et al., 2016a, Pacifici et al., 2015). Carbon storage was not included in this analysis, though it has been suggested by Elkin et al. (2013) that carbon dynamics and runoff would be relatively insensitive to a 2 °C increase in temperature. This has been suggested to be due to an insensitivity to vegetation dependent on forest cover (Fahey et al., 2010, Elkin et al., 2013). This is similar to other studies that have demonstrated that carbon storage is strongly dependent on forest cover, suggesting carbon storage would decrease as forest cover decreased. Wolf et al. (2012) demonstrated that in multiple catchments in Switzerland carbon storage only increased under a reforestation scenario during the re-growth phase. It has also been suggested that carbon uptake rates are not driven by climate, though rather by forest age and demographics (Hlásny et al., 2011).

Reviews Runting et al. (2016a) and Martinez-Harms et al. (2015) suggest that most studies focus on the 'biophysical supply' of ecosystem services. Bagstad et al. (2013) has suggested that measuring the benefit flows from ecosystems to people is more accurate for valuation than ecosystem service provision alone. There has also been criticism that studies investigating the supply of ecosystem services alone are flawed without attempts to understand demand (Deng et al., 2016). Castro et al. (2014) reported that using a method whereby ecosystem services were mapped with social demand were useful for identifying spatial mismatches.

The method of 'benefit transfer', whereby values from other studies are applied to an area or service, was used for some of the ecosystem services in this chapter. For example, values for timber came from the Frome area, Dorset. This method has been criticised, as ecosystems services have been found to vary, even by nearby regions, and thus values are often not transferable (Plummer, 2009). However, as Dorset and the New Forest are adjacent to each other, the values should be similar. Though using ecosystem services values from today, and projecting them into the future can be very unreliable, as other environmental factors, and their influence on ecosystem services is not considered. This study used the same values across the different climate change scenarios to allow direct comparison between them. Multi-criteria analysis is usually utilised for management scenarios, where spatial distributions of vegetation can be controlled. For this chapter, this technique was used for climate change scenarios, as though the scenarios were not 'chosen' they were used in the modelling of the vegetation for the New Forest, and the resultant maps were of ‘final’ vegetation, and thus similar in end product to those that would have been outputted from a management scenario. The multi-criteria analysis allowed direct comparisons between scenarios, whether management informed or climate modelled.

A major limitation with any climate change study is the uncertainty around climate change. The UKCP 90% probability projections were used to generate the habitat maps that informed the ecosystem service provision. The uncertainty in the modelled climate maps is apparent though the larger areas of non-common land amongst the three replicates. Refinement of this work would see the climate maps for the New Forest modelled at different probabilities and compared. There is regionalised variation in drivers of climate change (van Vuuren et al., 2007). Baró et al. (2014) have found that urban forests can help towards climate change mitigation strategies through abating pollution to an extent. Due to variability in climatic variables (i.e. temperature and precipitation) which often show complex temporal and spatial patterns; ecosystem service provisions are often reflective of this variability (Dobrowski et al., 2013, Runting et al., 2016a). Murphy et al. (2009) suggests that there are three causes of climate change uncertainty; uncertainty regarding future greenhouse gas emissions, incomplete understanding of Earth system processes and variability in the natural climate. It is important to understand the condition of an ecosystem, as this impacts its ability to resist impacts of climate change (Lavorel et al. 2015). Big changes in climate will typically lead to big changes in ES supply (Schröter et al. 2005). To this end, if the modelled New Forest climate change maps had more than 3 repetitions we would be able to see the variability in the modelling approach clearer.

The impacts of climate change in my study are indirect and through impacts on vegetation, and don’t include direct impacts such as evapotranspiration (Middelkoop et al., 2001), the effect of insect herbivores with rising temperature (Bale et al., 2002), ecological changes in phenology (Parmesan, 2006, Ashmore, 2005), or changes in the hydrologic and thermal regimes of rivers (van Vliet et al., 2013). A further dimension is the impact on people’s behaviours, with cultural ecosystems services having the potential to being impacted by recreation and tourism, and factors such as attractiveness of certain weather , for example sunnier days, extreme weather (Hamilton et al., 2007), psychological adaptation to climate change (Grothmann and Patt, 2005) and behaviour guided by the effects of climate change on health (Patz et al., 2005). Despite limitations of this study, the results are still useful in helping to guide future management decisions, and indeed forewarn of a drop in ecosystem service provision in the New Forest National Park.

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Chapter 6

Discussion

Results show that when presented with a variety of ecosystem services, people ranked aesthetic, heritage and recreation as the most important (Chapter 2). This highly important result is reflected in the work of Quétier et al. (2010) whom found in mountain grasslands in the French Alps found people mostly spoke in terms of cultural ecosystems services. Similarly, Palomo et al. (2011) found most users in Doñana, Spain, used cultural ecosystems services the most. Cultural ecosystem services have been suggested as being important for physical and mental health, cultural diversity and spiritual values (English Nature, 1994, De Groot et al., 2002). The high importance of cultural ecosystem services indicate they need to be included in land management planning processes. Studies have suggested that focusing on cultural ecosystem services can lead to productive land and water resources management (Tielbörger et al., 2010). Yet cultural ecosystems services are rarely included in assessment, with Benayas et al. (2009) finding that out of 89 studies reviews, none used spatially explicit cultural ecosystem service indicators.

Practical cultural ecosystem service indicators are difficult to investigate, with a review of literature by Hernández-Morcillo et al. (2013) showing that only 23% used spatially explicit information. These services are often not included in ecosystem assessments due to methodological challenges (Plieninger et al., 2013, Daniel et al., 2012, Seppelt et al., 2011, Feld et al., 2009). Hernández-Morcillo et al. (2013) recommends participatory mapping tools and involvement of relevant stakeholders to investigate cultural ecosystem services to aid avoidance of misrepresentation. Spatially explicit approaches allow identification of habitat types that have high values for ecosystems services.

Broadleaved, mixed and yew woodland was found to have the highest aesthetic value used fixed-sum survey questions (Chapter 2). The participatory GIS (pGIS) results showed some clustering around towns, though not as much as heritage and recreational value (Chapter 2). The pGIS method gave broadleaved, mixed and yew woodland a mid (0.5) normalised score, compared to the top score in the fixed sum question, with grassland having the highest aesthetic score in pGIS (Chapter 2). The aesthetics of woodland, this time factoring in stages of decline, was investigated in Chapter 3 using two methods; the first, conducting face-to-face surveys with visitors, and the second, GPS tracking of the surveyed individuals yielded striking results. When shown images of beech woodland in various stages of dieback, visitors indicated a preference for wooded landscape (with 100% woodland scoring the highest aesthetic value), though behaviourally much less time was spent in woodland than expected.

Similar to aesthetic value, broadleaved, mixed and yew woodland was found to have the highest recreational value from the fixed-sum question, with clustering being seen to greater extent around towns than aesthetic value (Chapter 2). Broadleaved, mixed and yew woodland scored lower with the pGIS method, with a normalised score of 0.4, with grassland having the highest recreation score with pGIS (Chapter 2). Chapter 3 shows a positive relationship between woodland cover and recreation, with a partial 75% of woodland cover having the highest preference scores for recreation. The spatially explicit modelling tool 'InVEST Visitation: Recreation and Tourism' was used to value recreation in Chapter 4, with results indicating that recreational value was higher within the protected Sites of Special Scientific Interest (SSSI) area, than in the rest of the New Forest National Park. Heritage had the lowest pGIS value across the New Forest (Chapter 2), with strong clustering around towns. Visually, the hotspots for heritage value were more distinctive than aesthetic and recreational value.

Chapter 2 showed the differences that were seen between the fixed sum and pGIS methods for valuation of cultural services. Woodland areas scored higher with fixed sum methods compared to pGIS. This result was mirrored in Chapter 3, where face-to-face survey responses indicated a recreational and aesthetic preference for woodland, even though GPS tracking showed that time in woodland was much lower than expected. This suggests that visitor's aesthetic preferences are not translated into actual behaviour. This can either be explained by a difference in visitors stated and behaviour-aesthetic preferences, or that visitors are more guided by recreational appeal than visual aesthetic appeal. Either way, the implications are clear. To understand where people go, you have to 'see' where they go, using techniques that monitor behaviour. Asking people what they prefer can elicit responses that do not directly translate to behaviour.

Recreation and aesthetic value both had synergies with high scores for broadleaved, mixed and yew woodland (Chapter 2). This indicates that woodland is needed to preserve these cultural services. Urban habitats scored low for recreational and aesthetic value using fixed sum scoring (Chapter 2). Though the pGIS exercise showed strong clustering of heritage points around towns and villages (Chapter 2). Interestingly, clustering of recreational and aesthetic value points were seen along the coastal areas within the study area (Chapter 2). Few strong correlations among ecosystem services (Chapter 4). Biodiversity showed weak trade-offs with livestock, phosphorous retention and weak synergies with carbon storage, pollinator habitat quality and a moderate synergy with tranquillity. Despite the biodiversity score being low overall, mean scores were higher within the SSSI. The SSSI was found to have significantly higher high values of combined ecosystem service provision, with an increase of 23% over expected, for the 90th percentile (Chapter 4). This highlights the need to protect and expand appropriate areas as highlighted in the Making Space for Review (Lawton et al., 2010), with future threats, such as climate change only adding to this need.

Climate change will have a negative impact on the ecosystem service provision within the New Forest SAC (Chapter 5). Any management of policy measure or initiative could be rendered invalid in its approach if future climate uncertainty is not considered. Climate change adaptation decisions are difficult due to the ‘uncertainty cascade’ surrounding climate change. This includes assumption on greenhouse gas increasing, incomplete climate models, downscaling of climate projections, physical, data and parameter uncertainty amongst an array of other (Refsgaard et al., 2013). The uncertainty of the exact nature of future climate change is the reason why efforts should be made to predict changes in natural environments, so that the range of protected areas and the protection to ecosystem service provision and biodiversity they provide, is understood and managed appropriately.

The temporal and spatial variability of ecosystem services make the implications the study site very relevant. The New Forest’s National Park Management Plan was updated in 2015 and renamed the Partnership Plan for the New Forest National Park (NFPA, 2015). The plan explicitly states that planning for climate change is a priority, with concerns over the spread of plant diseases and pests, and change in habitats (including the drying of wetlands) and changes to the coastline. The results of this thesis, including the fact that the SSSI areas protect the higher levels of ecosystem service provision are directly related to Goal CC4, that aims to 'identify likely future climate-related impacts'.

This thesis is well timed, with the Partnership Plan for the New Forest National Park (NFPA, 2015) clearly identifying that the development of an 'ecosystem services approach' is key to moving forwards under the Biodiversity 2020 National Strategy for England (DEFRA, 2011). Lawton et al. (2010), suggests that the current protected areas network does not provide a resilient ecological network to capably respond to future climate change, though such a network to protect ecosystem services and biodiversity could be created. The Biodiversity 2020 National Strategy for England responded with several large scale initiatives including creating twelve Nature Improvement Areas and increasing the number of SSSIs. The Partnership Plan for the New Forest National Plan goes further, with Goal LM3 stating '[To] promote the use of ecosystem service maps of the National Park… to show the value of the natural environment for social well-being and the economy'. This thesis demonstrates that synergies can exist between ecosystem service provision with biodiversity, with win-win scenarios for multiple ecosystem services only identifiable through the application of a spatially explicit approach. This is indicative of the nature of biodiversity as an underpinning device of ecosystem good and services (Hooper et al., 2005). As Mace et al. (2012) explains, it is not biodiversity alone has impacts on ecosystem service flows, though rather the complexity of interactions between abiotic and biotic elements. Though the synergies that are seen in the New Forest between ecosystem services and biodiversity can also be indicative of stabilising property of biodiversity on ecosystem service supply (Cardinale et al., 2012).

The Wild Purbeck area extends west to Poole Harbour, less than 16 kilometres from the New Forest National Park and is subject to similar issues, owing to similar habitats and tourist pressure. The Wild Purbeck Improvement Areas was a project that ran for three years, culminating in 2015 (Dorset Area of Natural Beauty (AONB), 2015). A 'community gateway' approach was used to increase dialogue between urban and rural communities to understand ‘natural infrastructure’, and its management needs. Projects included heathland restoration in Special Areas of Conservation (SAC), Special Protection Areas (SPA) or Sites of Special Scientific Interest (SSSI). This suggests that using a ‘community gateway’ approach, and using methods including participatory GIS could be combined with improvement of unfavourable or declining protected (SSSI) areas within the New Forest to increase the ecosystem service and biodiversity provision. Several projects of the Wild Purbeck Improvement Areas project could have been more informed through the understanding of ecosystem service trade-offs and synergies and factoring in changing climate, such as the heathland restoration project.

The most recent New Forest National Park Recreation Management Strategy (NFPA, 2010) states that recreation will need to adapt to climate change to help lessen the Park’s carbon footprint. One of the key ways for managing recreational impacts was stated as focusing recreational activities around ‘gateway’ sites and linear routes. This thesis directly helps this goal through the identification of the habitats and areas that visitors to the New Forest prefer (specifically Chapters 2 and 3). An example is a visitor’s preference for partially open woodland, in line with the Recreation Management Strategy. Recreational activities could be focused in already popular areas, such as Bolderwood.

The New Forest National Park Recreation Management Strategy’s vision aims to have a strong sense of heritage (NFPA, 2010), though my results indicate that of all the cultural services investigated: aesthetic, recreational and cultural heritage value, that the latter is least valued (Chapter 2). Malpas (2008) suggests that a loss of sense of place is a feature of modern life, tied closely with heritage and a sense of culture. This suggests that visitors lacked an understanding, or knowledge, of the cultural heritage significance of the New Forest. The New Forest has a rich history, with human activity reaching back over 750,000 years ago. Colourful traditions, customs and historic sites that can be explored by people to develop a sense of place and associated feeling of heritage, need to be explored further by management if it is to meet its vision.

Limitations and further research

Limitations for individual data chapters have been discussed in further detail within each chapter. General limitations of the thesis will be discussed in this chapter. One of the largest issues in mapping ecosystem services is that of proxies to map distribution of ecosystem services. This has been explored by Eigenbrod et al. (2010). With so much data for ecosystem services in the world being unavailable, proxies are often used; usually land-cover based or modelled surfaces from non-primary data. The cultural services assessed in this thesis (aesthetic, recreational, and heritage value) have used multiple approaches, collecting primary data from within the study area, avoiding using proxies (specifically Chapters 2 and 3). Chapters 4 and 5, in an effort to broaden the ecosystem services assessed, used several habitat based proxies. This included timber, livestock, pollinator habitat quality and flood risk mitigation. The InVEST Water Yield, Nutrient Retention, and Carbon Storage and Retention models had data inputs that were specific to the area due to the spatial nature of the input, including precipitation, evapo-transpiration and land-cover, though data for biophysical factors and carbon pool data was either national or from the neighbouring county of Dorset. Chapter 5 used multiple proxies by calculating the value of an ecosystem service by habitat type, and projecting it directly into a future climate change emissions scenario, informed by New Forest land cover maps. It has been suggested that using proxies for identifying hotspots is unreliable (Eigenbrod et al., 2010), hence the results seen in Chapter 5 should be considered to pose the most risk.

Proxies have been found to be a poor fit to primary data for biodiversity, recreation and carbon storage (Eigenbrod et al., 2010). Apart from investigating the impact of climate change on ecosystem service provision in Chapter 5, this thesis has not used proxies for any of these services, though it could be argued that the carbon pool parameters used in the InVEST Carbon Storage model pose a risk from being based on figures from Dorset. Biodiversity was mapped in Chapter 4 from data supplied by the Hampshire Biodiversity Information Centre on locations of protected species. The protected species included were taken from Section 41 of the Natural Environment and Rural Communities (NERC) Act, indicating species that are England’s rarest and threatened (Natural England, 2013). Recreation values were mapped from multiple primary data methods.

Cultural services have been found to often be under-represented in ecosystem service assessments, as discussed earlier in this thesis, thus have been the primary objective for data collection as part of this research. Limitations with proxies cannot be overlooked; the ideal situation would be to substitute proxies with primary data for the later chapters investigating ecosystem service provision within protected areas. In regard to climate change, the modelling of each ecosystem service using projected parameters would be the next step. This would be the ideal, and would make my approach more robust, though is equally resource intensive and, unfortunately, beyond the scope of this thesis.

Other changes to this thesis would include broadening the variety of cultural services that were assessed, including valuing heritage by more than one method (Chapter 2). Other studies have investigated an increased number of cultural ecosystem services, incorporating the whole range of cultural ecosystem services that the Millennium Ecosystem Assessment (2005) established; including Plieninger et al. (2013) whom investigated spiritual services, education values, inspiration and social relations as well as aesthetic, recreation and heritage values, in Saxony, Germany. Plieninger et al. (2013) found that cultural services followed patterns, often following landscape features. This thesis has only focused on benefits, or ecosystems services. To provide a more balanced and robust approach, it is recommended that future research include ‘ecosystem disservices’, the costs of the ecosystem. (Plieninger et al. (2013)) found that cultural ecosystem disservices (unpleasantness, scariness and noisiness) gave an indication of recreation overuse. Additionally, both the New Forest National Park, and SSSI or SAC areas within the Park are protected under varying levels of legal protection. To make this thesis more robust, I would have expanded my analysis to include the surrounding area, in order to make comparisons with non-protected areas.

The objectives set out at the start of this thesis have been achieved. In Chapter 2 recreation and aesthetic values were found to differ between pGIS and fixed-sum scoring methods, with Chapter 3 further showing that face-to-face surveys and GPS tracking yield differing results for these cultural ecosystem services. This first half of the thesis demonstrated the importance of cultural ecosystem services, and how stated and behavioural responses yielded differing values. Chapter 4 reaffirmed the importance of management goals to protect ecosystem services, with the highest areas of combined ecosystem service provision being demonstrated to exist within the boundaries of the New Forest SSSI, compared to the rest of the National Park. And finally, Chapter 5 looked forwards and attempted to provide an indication of the future ecosystem service provision amongst multiple uncertainties, suggesting a drop in provision. This thesis has been an exercise in utilising an ecosystem services approach to assess the varied benefits we garner from nature, how areas we designate 'protected' protect these benefits and how these could change in the future.

By looking at the New Forest at a local scale, the nature of spatial patterns being scale dependent should be considered (Grafius et al., 2016). By using a 25m resolution, rather than courser resolutions such as 1 km, I have been able to relate to the environmental element being studies (in this instance mostly vegetation cover) and being able to capture spatial heterogeneity in the landscape. This should also lead to ecosystem values being more accurate, as studies have shown up to 200% in values from using finer, than courser resolutions (Konarska et al., 2002), and the importance of using robust landcover maps, as was here (Hampshire Biodiversity Information Centre maps) has been demonstrated through the negative impact on valuation with the use of low quality misclassified maps (Foody, 2015). Using the CICES classification (Haines-Young and Potschin, 2011) to choose ecosystem services allows comparisons of data collected here to be compared with future studies.

There are two main areas that this thesis expands knowledge; provisioning of ES in Protected Areas and the differences in methods for assessing cultural ecosystem services, namely aesthetic and recreational value. To a lesser extent, the dramatic ability of climate change on provision of ES has also been demonstrated, though a finer and more robust approach is needed to model ES for climate change due to the important nature of the task.

Despite the contrast seen in literature regarding Protected Areas not being able to conserve biodiversity or ecosystem services (Xu et al. 2017), my thesis has contributed towards the opposite view of Bridgewater and Babin (2017) and as demonstrated by Eigenbrod et al. (2009) for protected areas in England. Biodiversity was seen to be greater in the SSSI area of the New Forest National Park. The multi-layered relationship of ES with biodiversity infers that higher values of biodiversity within the SSSI will also strengthen other ecosystem services through its underpinning role and ability to increase temporal stability of ecosystem system functioning (Mace et al., 2012 & Cardinale et al., 2012). Using a spatially explicit approach can lead to more efficient land management decisions. It is often difficult for management actions and interventions to be assessed, and the use of mapping ES can allow a visually strong representation for decision makers.

CES are often acknowledged as being important but often neglected in assessments due to methodological challenges (Plieninger et al., 2013, Daniel et al., 2012, Seppelt et al., 2011, Feld et al., 2009) causing representation bias (Hernández-Morcillo et al. 2013). There is a lack of robust approaches to assessment which may contribute to this (Milcu et al. 2013, Hirons et al. 2016, Hernández-Morcillo et al. 2013). This thesis examined recreation and aesthetic services as examples of CES, at the landscape scale which has been lacking in other studies (Hirons et al. 2016, Milcu et al. 2013). Here a comparison was made between a stated (survey) method and behavioural (GPS tracking) method. The implication here was that stated responses methodologies were best utilised for aesthetic assessment and behavioural monitoring methodologies for recreational assessment of these cultural services. This thesis has explored provision of ES in the New Forest National Park though has demonstrated the application of local methodologies to assess ES is important for accurate values, especially for CES, which are more complex to dis-entangle and measure.

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Appendix I: Supplementary material for Chapter 2

I.1. Website design

The website consisted of an introductory welcome page, a demographic questions page, an ecosystem scoring page, a page explaining the ecosystem services, three pages scoring habitats for aesthetic, conservation and recreation value respectively, a page with habitat photographs, a mapping introduction page, a mapping page, and a thank you page, with a further page acknowledging email entry into the prize draw. Figure I.1.1 – Figure I.1.11 show screenshots of these pages with descriptions. MySQL databases were used to store the data from the online survey (Figure I.1.12).

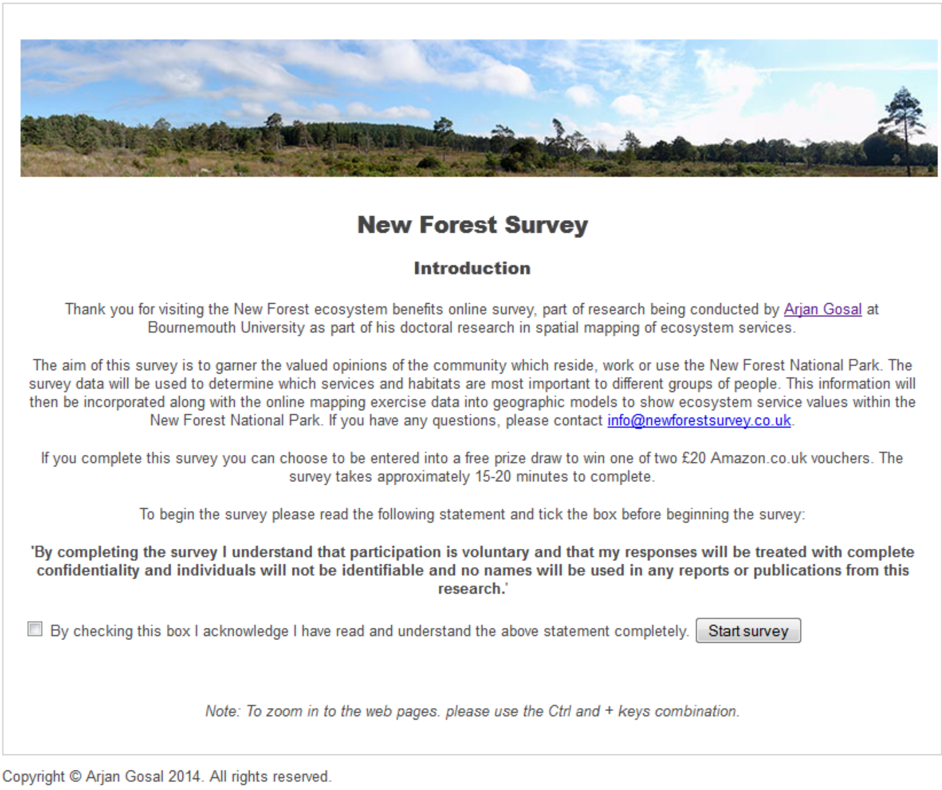
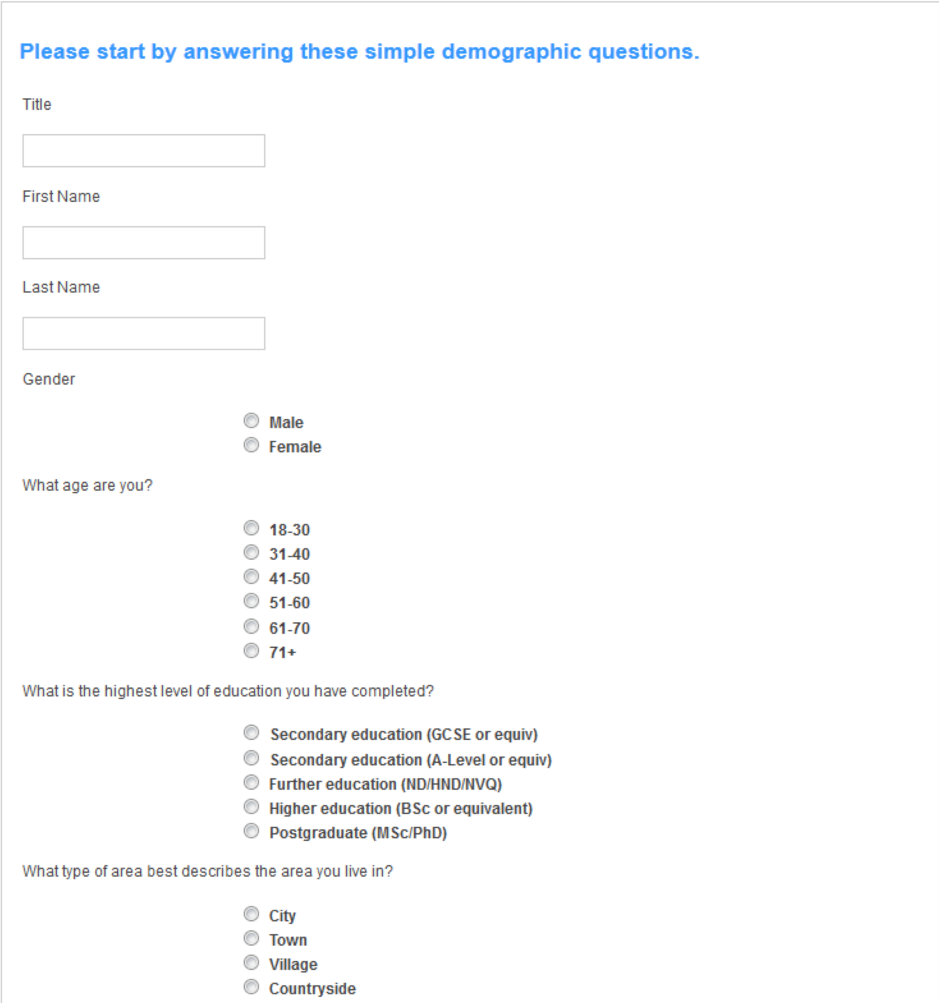


Figure I.1.1: New Forest Survey introduction page: the survey and how the results would be used was explained with a consent box that the participant had to check being before able to proceed to the next page. If a participant selected 'Start survey' without checking the consent box a warning pop-up that they had to read the provided information and check the box to be able to proceed.



Please start by answering these simple demographic questions.

Title

First Name

Last Name

Gender

- Male
- Female

What age are you?

- 18-30
- 31-40
- 41-50
- 51-60
- 61-70
- 71+

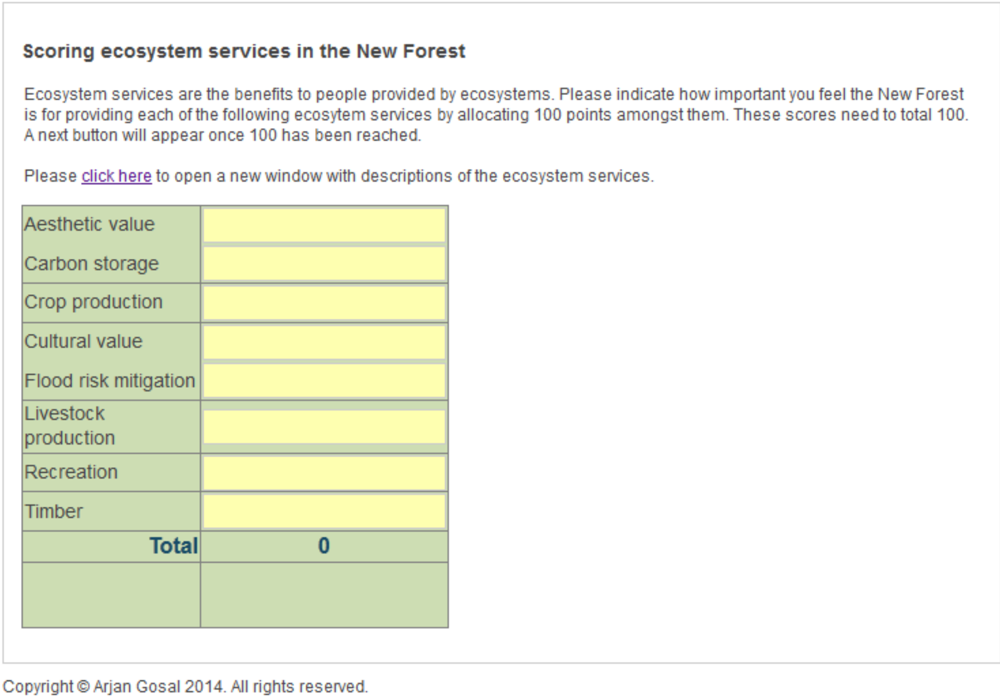
What is the highest level of education you have completed?

- Secondary education (GCSE or equiv)
- Secondary education (A-Level or equiv)
- Further education (ND/HND/NVQ)
- Higher education (BSc or equivalent)
- Postgraduate (MSc/PhD)

What type of area best describes the area you live in?

- City
- Town
- Village
- Countryside

Figure I.1.2: Demographic questions page. The participant was asked for title, first name, last name, gender (male or female), age group (18-30, 31-40, 41-50, 51-60, 61-70 and 71+), highest level of education completed (secondary education - GCSE or equivalent, secondary education - A-Level or equivalent, further education -ND/HND/NVQ, Higher education - BSc or equivalent and postgraduate -MSc/PhD), type area of residence (city, town, village and countryside), proximity to the New Forest National Park (within the National Park boundaries, within 10 km of the National Park boundaries, within 30 km of the National Park boundaries and more than 30 km away from the National Park boundaries), interest groups and organisation affiliation (local government, nature conservation management, agricultural production, forestry, water production, fisheries, game management, residential and tourism), land ownership in the New Forest (own private property, own commercial property, manage private property, manage commercial property, do not own or manage any land, have Commoners Rights within the New Forest) and agreement with the statement 'I have a good understanding of conservation and ecosystem management issues within the New Forest' (strongly agree, neither agree or disagree, disagree and strongly disagree).



Scoring ecosystem services in the New Forest

Ecosystem services are the benefits to people provided by ecosystems. Please indicate how important you feel the New Forest is for providing each of the following ecosytem services by allocating 100 points amongst them. These scores need to total 100. A next button will appear once 100 has been reached.

Please click here to open a new window with descriptions of the ecosystem services.

Aesthetic value	
Carbon storage	
Crop production	
Cultural value	
Flood risk mitigation	
Livestock production	
Recreation	
Timber	
Total	**0**

Figure I.1.3: Scoring Ecosystem services in the new forest page: participants were asked to indicate how important they felt the New Forest was for providing each of the following ecosystem services by allocating 100 points amongst them; aesthetic value, carbon storage, crop production, cultural value, flood risk mitigation, livestock production, recreation and timber. Descriptions of each ecosystem service were provided in the form of a new web-page popping up when the participant clicked the link.

Close Window

Description of ecosystem services

Click the links below to view a description for each ecosystem service.

Aesthetic value

Carbon storage

Crop production

Cultural value

Flood risk mitigation

Livestock production

Recreation

Timber

Aesthetic value

Characteristics of the landscape and natural world can be of aesthetic value to people (a beautiful landscape), may inspire a sense of spiritual well-being and can act as an inspiration to the arts past and present.

Back to top

Carbon storage

The characteristics of different plant species determine how much carbon dioxide is taken up from the atmosphere and how much is released into it. Certain habitats (e.g. bogs and old forests) are also important stores for carbon captured in the course of time. Therefore, the destruction of these habitats would result in releasing high amounts of carbon dioxide to the atmosphere, contributing negatively to climate change.

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Figure I.1.4: Ecosystem service descriptions page: the page displayed when the participant clicked the link from the previous page for more details on ecosystem services. See Appendix I.2 for a full list of descriptions.

Scoring New Forest habitat types for aesthetic value

The **aesthetic value** of a habitat can be described as the value you place on a habitat for its beauty or visual appearance.

Please indicate how important you feel the following New Forest habitat types are for **aesthetic value** in your opinion, by allocating 100 points amongst them. These scores need to total 100. A next button will appear after once 100 has been reached.

Please click here to open a new window with photos of the habitat types.

Acid grassland	
Arable cereals	
Arable horticulture	
Broad-leaved / mixed woodland	
Coniferous woodland	
Dwarf shrub heath	
Fen, marsh, swamp	
Improved grassland	
Neutral grassland	
Suburban / rural developed	
Urban	
Total	**0**

Figure I.1.5: Scoring New Forest habitat types for aesthetic value page: Participants were asked to indicate how important they felt different New Forest habitats were for aesthetic value by allocating 100 points amongst them; acid grassland, arable cereals, arable horticulture, broad-leaved / mixed woodland, coniferous woodland, dwarf shrub heath, fen/ marsh/ swamp, improved grassland, neutral grassland, suburban / rural developed and urban. Photographs were provided for each habitat type in the form of a new web-page popping up when the participant clicked the link.

Scoring New Forest habitat types for conservation value

The **conservation value** is a measure of how important a habitat is for wildlife. It reflects how strongly you feel a habitat should be protected, or conserved, to prevent its loss.

Please indicate how important you feel the following New Forest habitat types are for **conservation value** in your opinion, by allocating 100 points amongst them. These scores need to total 100. A next button will appear after once 100 has been reached.

Please click here to open a new window with photos of the habitat types.

Acid grassland	
Arable cereals	
Arable horticulture	
Broad-leaved / mixed woodland	
Coniferous woodland	
Dwarf shrub heath	
Fen, marsh, swamp	
Improved grassland	
Neutral grassland	
Suburban / rural developed	
Urban	
Total	**0**

Figure I.1.6: Scoring New Forest habitat types for conservation value page: Participants were asked to indicate how important they felt different New Forest habitats were for conservation value by allocating 100 points amongst them; acid grassland, arable cereals, arable horticulture, broad-leaved / mixed woodland, coniferous woodland, dwarf shrub heath, fen/ marsh/ swamp, improved grassland, neutral grassland, suburban / rural developed and urban. Photographs were provided for each habitat type in the form of a new web-page popping up when the participant clicked the link.

Scoring New Forest habitat types for recreation value

The **recreation value** of a habitat can be descibed how important you feel it is for recreation including hiking, dog-walking, horse-riding, cycling and picnicking.

Please indicate how important you feel the following New Forest habitat types are for **recreation value** in your opinion, by allocating 100 points amongst them. These scores need to total 100. A next button will appear after once 100 has been reached.

Please click here to open a new window with photos of the habitat types.

Acid grassland	
Arable cereals	
Arable horticulture	
Broad-leaved / mixed woodland	
Coniferous woodland	
Dwarf shrub heath	
Fen, marsh, swamp	
Improved grassland	
Neutral grassland	
Suburban / rural developed	
Urban	
Total	**0**

Figure I.1.7: Scoring New Forest habitat types for recreation value page: Participants were asked to indicate how important they felt different New Forest habitats were for recreation value by allocating 100 points amongst them; acid grassland, arable cereals, arable horticulture, broad-leaved / mixed woodland, coniferous woodland, dwarf shrub heath, fen/ marsh/ swamp, improved grassland, neutral grassland, suburban / rural developed and urban. Photographs were provided for each habitat type in the form of a new web-page popping up when the participant clicked the link.

Figure I.1.8: Habitat photographs page: displayed when the participant clicked the link from the previous page for more details on habitat types. See Appendix I.2 for a full list of descriptions.

Mapping task

Thank you for your participation so far. The final part of this survey involves a mapping task. Please read the instructions below before continuing.

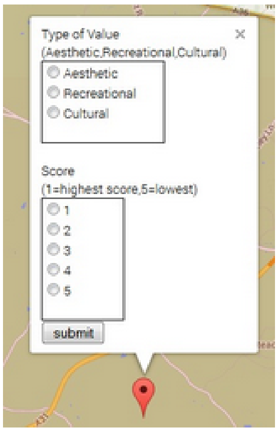
The purpose of this task is to identify locations in the New Forest that are important to you. To do this, you are required to place markers on the map, which indicate locations that you believe are particularly important. The decision of where you place the markers is entirely up to you. For each marker you must assign a 'value' and a 'score'. There are three different 'values':

Aesthetic value - How visually pleasing you find the location.
Recreational value - The importance of the location to you for your recreational activities.
Cultural value - How important you feel that the location is culturally.

The scores are ranked 1 to 5, with 1 being the most important, and 5 the least important. When you position your markers, you should select which area is the most important, and assign it a score of 1. The next most important, you should select a score of 2, and so on.

Please place up to 5 markers for each value type. You are invited to do this for each of three different types of value - aesthetic, recreational and cultural. ***Please do not assign the same score to multiple markers of the same type (e.g. if you placed 5 markers for recreation values, each would need to have a different score between 1 and 5, with no two markers having the same score.)***

To place marker, position the cursor on the location that you wish to select. You can then select the value and score by left clicking, which will produce a pop-up menu. This will allow you to select the value and score. Please make sure you click 'submit' every time you place a new marker, to ensure that it will be saved automatically. Clicking an existing marker will enable you to edit its value or score, or delete it entirely.



Click here to continue

Figure I.1.9: Mapping task introduction page: the objective of the task was explained, asking the participant to identify locations in the New Forest that were important to them. The ranking and selecting of the type of ecosystem service was explained.

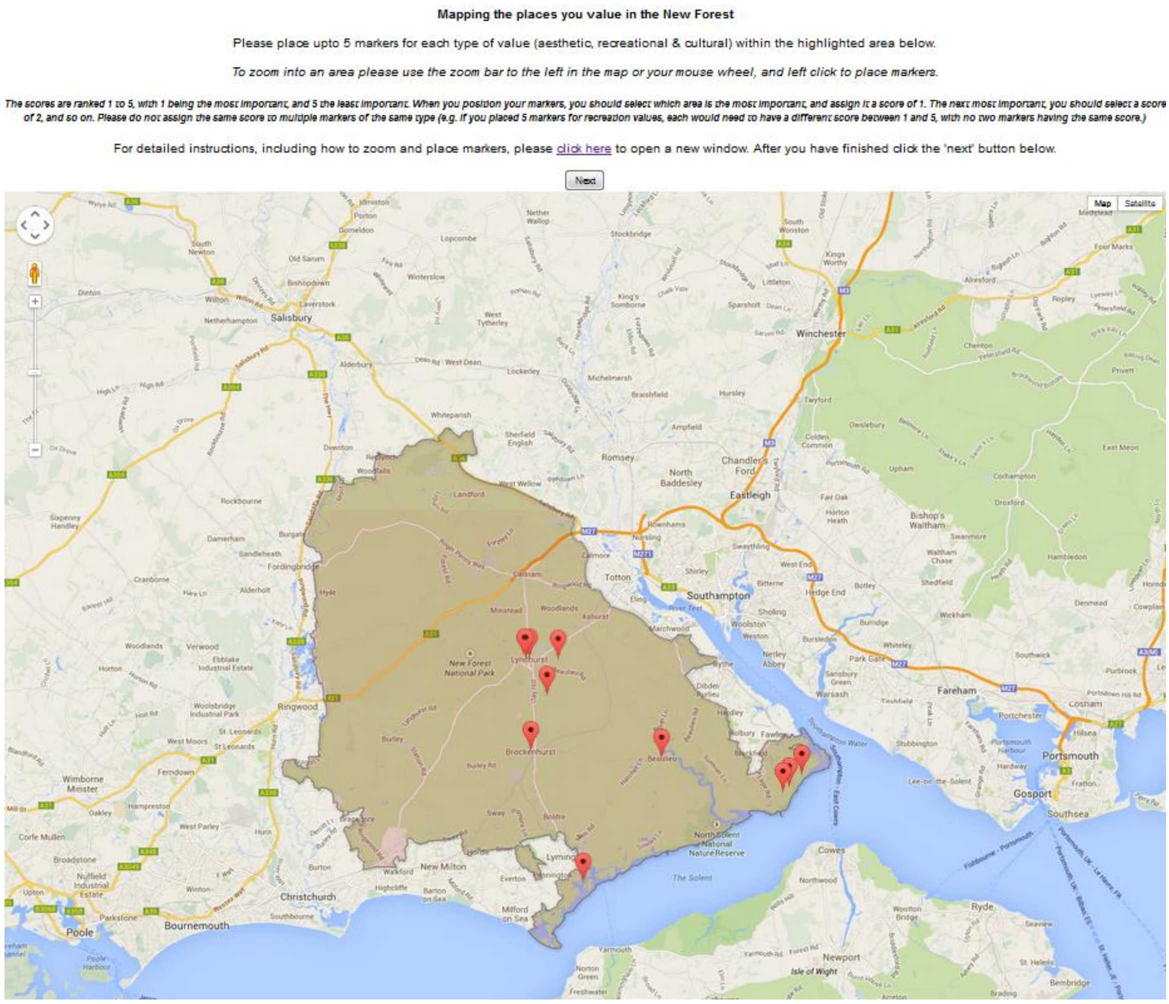


Figure I.1.10: Mapping the places you value in the New Forest page: An interactive google map with the boundary of the New Forest was presented to the participant with instructions on how to place pins on the map. The participant could zoom in and out, though could only place pins within the defined boundary of the New Forest.

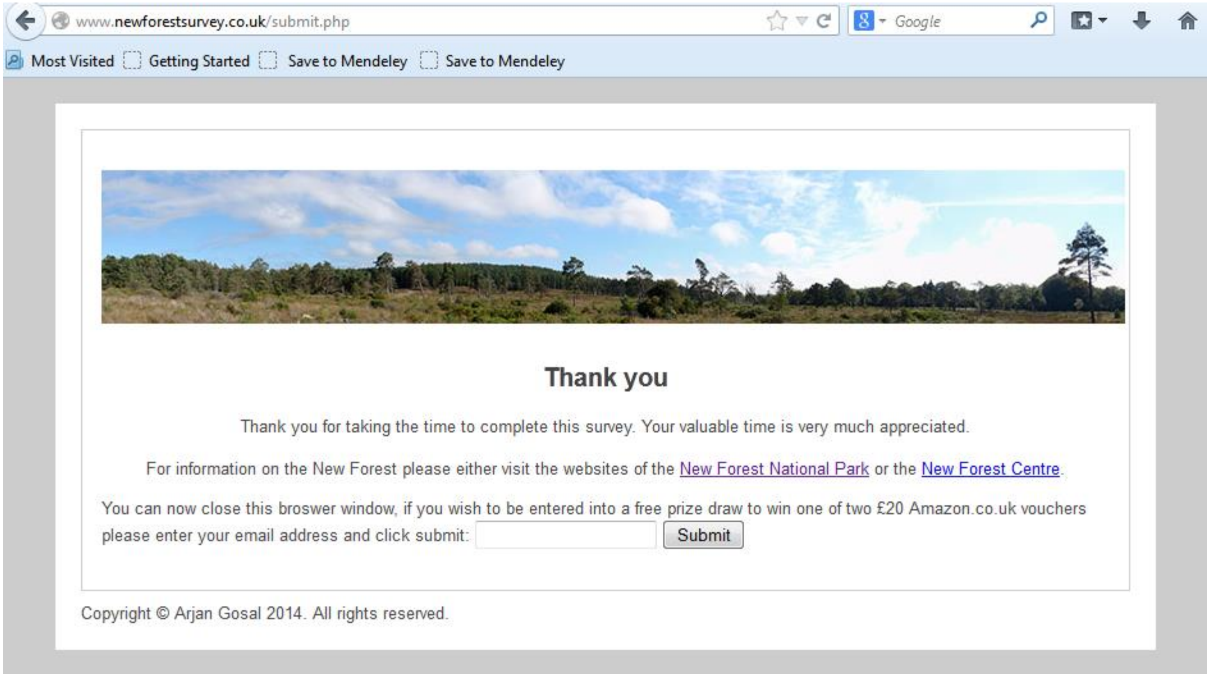


Figure I.1.11: Thank you page: the participant was thanked and links to further information about the New Forest provided. If the participant chose to enter the free prize draw, they entered their email address and clicked submit, ending the survey.

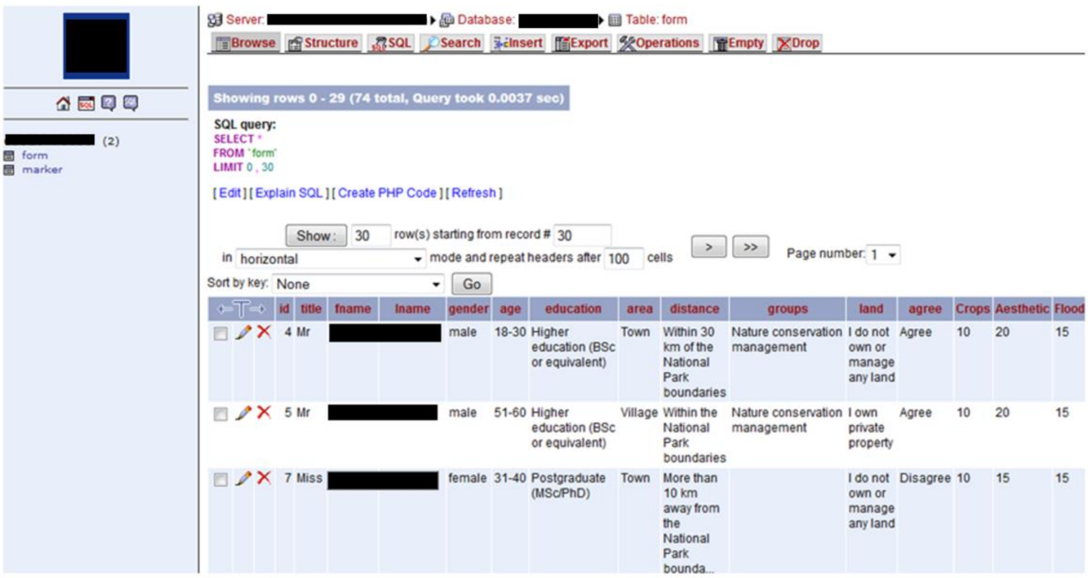


Figure I.1.12: MySQL database page accessible through the administrator's webserver account that was used to extract survey and location marker data. Note: personal details are blacked out.

I.2. Descriptions of ecosystem services

Aesthetic value

Characteristics of the landscape and natural world can be of aesthetic value to people (a beautiful landscape), may inspire a sense of spiritual well-being and can act as an inspiration to the arts past and present.

Carbon storage

The characteristics of different plant species determine how much carbon dioxide is taken up from the atmosphere and how much is released into it. Certain habitats (e.g. bogs and old forests) are also important stores for carbon captured in the course of time. Therefore, the destruction of these habitats would result in releasing high amounts of carbon dioxide to the atmosphere, contributing negatively to climate change.

Crop production

This includes production of agricultural and horticultural crops, including fruits and vegetables but not timber.

Cultural value

Certain landscapes have sites of historical and cultural importance, including historical monuments and memorials, locations featured in artistic works or literature, and sites of historical events.

Flood risk mitigation

Different habitats can play an important role in flood protection by holding water back and providing space for water, both within the floodplain and in upper reaches of a river. For example where habitats can provide an important water store, helping regulate river flows downstream. The creation and restoration of certain habitats within floodplains, such as flood meadows and reed beds can act as important water stores and can encourage the reconnection of rivers with their floodplain.

Livestock production

This includes ponies, cattle, pigs, donkeys and such.

Recreation

There are many opportunities for nature and bird-watching in the New Forest, attracting thousands of visitors a year. The provision of permissive access routes (footpaths, bridleways, etc.) and open access to certain habitats such as areas of heathland, woodland and grassland provides plenty of opportunities for hiking, dog-walking, horse-riding, cycling and picnicking in the New Forest.

Timber

The provision of timber used in construction, furnishings, paper and card etc.

I.3. Photos of New Forest habitat types

(a)

(b)

(c)

(d)

(e)

(f)

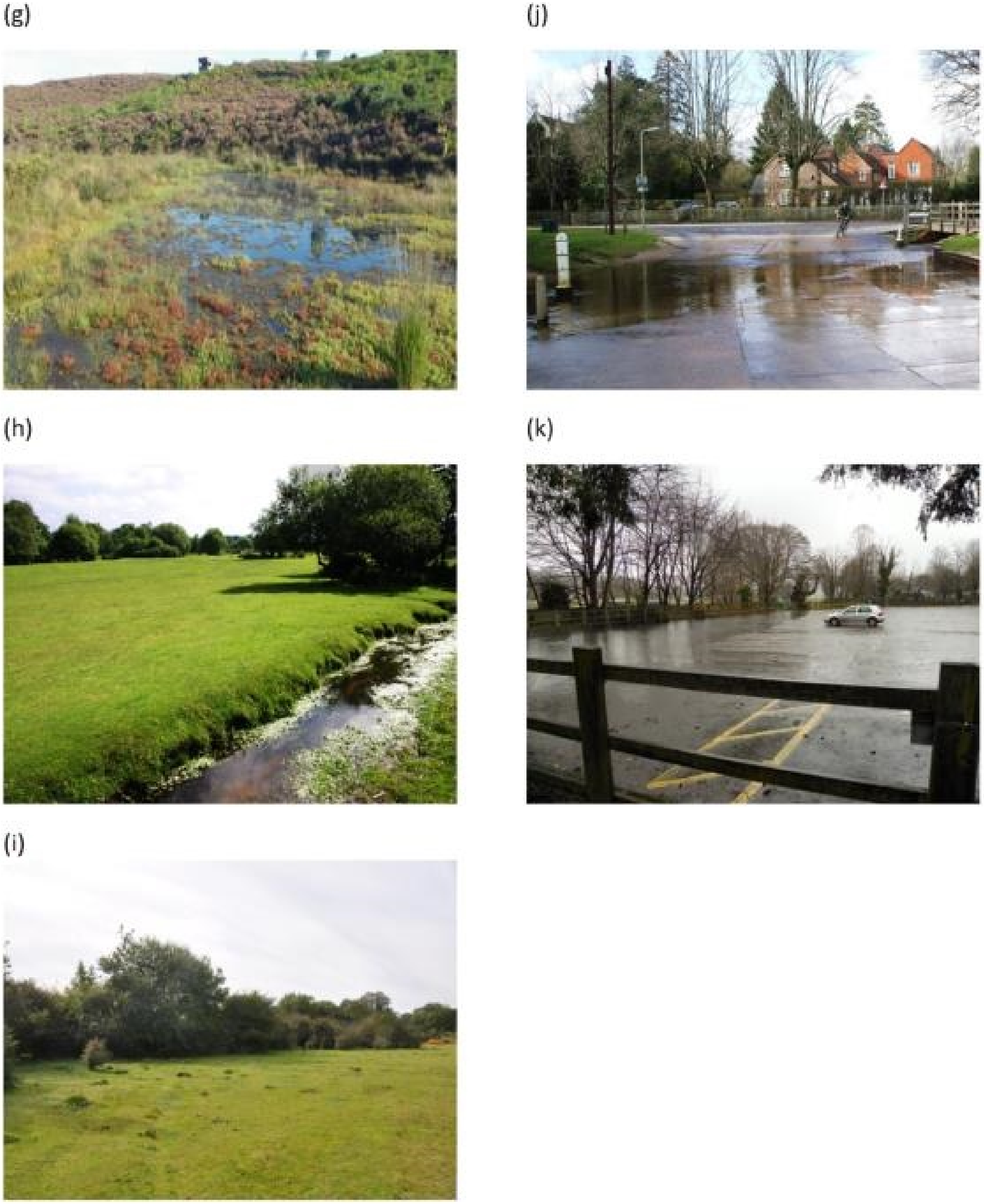


Figure I.3.1: Habitat photographs shown to survey participants (a) acid grassland (unimproved grassland on acid soils) (Perrin, 2006), (b) arable cereals (Woodland Trust, 2014), (c) arable horticulture (Lyburn Farm, 2014), (d) broad-leaved / mixed woodland (Cantarello, 2014), (e) coniferous woodland (Smith, 2007), (f) dwarf shrub heath (Moody, 2014), (g) fen, marsh, swamp (Lovegrove, 2014), (h) improved grassland (i.e. fertilised) (Danks, 2005), (i) neutral grassland (unimproved grassland on non-acidic, more calcareous soils) (Newton, 2014), (j) suburban / rural developed (Thomas, 2007) and (k) urban (Dixon, 2012).

I.4. Online survey publicity

Forest survey

SIR, — As part of my doctoral studies at Bournemouth University looking at spatial mapping of ecosystem services I have developed an online survey that is open to all stakeholders in the New Forest national park and surrounding areas.

This includes local residents, tourists, farmers, land owners, nature conservation organisations, local government, game managers and many others.

This research aims to gather valued opinions of the community who live, work or use the national park. The survey data will be used to determine the services and habitats most important to stakeholder groups.

This information will be incorporated with the online mapping exercise data into geographic models to show ecosystem service values within the national park. The survey takes approximately 15-20 minutes to complete.

If any reader would like to contribute to this important study please complete our short survey online at: www.newforestsurvey.co.uk For further information or for any questions please contact me info@newforest survey.co.uk

Arjan Gosal

Address supplied.

Figure I.4.1: Letter to the Editor of the Lymington Times, requesting participants to complete the online survey, published in February 2014 to encourage the public in the local New Forest area to complete the survey.

I.5. Normality testing

Table I.5.1: Kolmogorov-Smirnov test for normality (df = 132) for eight ecosystem services. Results show significant (* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$) or non-significant (n.s.) relationships.

Ecosystem service	Kolmogorov-Smirnov
Aesthetic value	0.246 ***
Carbon storage	0.182 ***
Crop production	0.216 ***
Cultural value	0.186 ***
Flood risk mitigation	0.150 ***
Livestock production	0.213 ***
Recreation	0.203 ***
Timber	0.237 ***

Table I.5.2: All habitat types across aesthetic, recreation and conservation value demonstrated significant distributions deviating from normal, verified using the Kolmogorov-Smirnov test. Results show significant (* P < 0.05; ** P < 0.01; *** P < 0.001) or non-significant (n.s.) relationships.

	Kolmogorov-Smirnov test statistic		
Habitat type	*Aesthetic value*	*Recreation value*	*Conservation value*
Acid grassland	0.167 ***	0.164***	0.190 ***
Arable cereals	0.250 ***	0.424 ***	0.297 ***
Arable horticulture	0.210 ***	0.418 ***	0.243 ***
Broadleaved / mixed woodland	0.230 ***	0.172 ***	0.191 ***
Coniferous woodland	0.190 ***	0.174 ***	0.169 ***
Dwarf shrub heath	0.182 ***	0.147***	0.178 ***
Fen/ marsh/ swamp	0.202 ***	0.186 ***	0.160 ***
Improved grassland	0.189 ***	0.232 ***	0.253 ***
Neutral grassland	0.202 ***	0.215 ***	0.192 ***
Suburban / rural developed	0.258 ***	0.305 ***	0.337 ***
Urban	0.378 ***	0.313 ***	0.436 ***

Table I.5.3: Kolmogorov-Smirnov test for normality (df = 132) for cultural ecosystem services. Results show significant (* P < 0.05; ** P < 0.01; *** P < 0.001) or non-significant (n.s.) relationships.

Ecosystem Service	Kolmogorov-Smirnov
Aesthetic	0.278***
Recreation	0.171 (n.s.)
Cultural	0.232**

I.6. References

Cantarello, E., 2013. Broadleaved woodland image. In Gosal, A. (Ed.) (Email.).

Danks, K., 2005. *Mill Lawn Brook* [online]. Available from: http://www.geograph.org.uk/photo/19600 [Accessed 04/11/2013].

Dixon, D., 2012. *Flooding at Burley Car Park* [online]. Available from: http://www.geograph.org.uk/photo/3269926 [Accessed 04/11/2013].

Lovegrove, A., 2013. Fen, marsh and swamp image. In Gosal, A. (Ed.) (Email.).

Lyburn Farm, 2013. *Lyburn Pumpkin and Squash* [online]. Available from: http://www.lyburnfarm.co.uk/pumpkin2/ [Accessed 04/11/2013].

Moody, C., 2013. Dwarf shrub heath image. In Gosal, A. (Ed.) (Email.).

Newton, A., 2013. Neutral grassland image. In Gosal, A. (Ed.) (Email.).

Perrin, C., 2006. *Ancient Grassland Hengistbury Head Dorset* [online]. Geograph. Available from: http://www.geograph.org.uk/photo/184296 [Accessed 04/11/2013].

Smith, A., 2007. *Coniferous woodland, Clarendon Park* [online]. Available from: http://www.geograph.org.uk/photo/483710 [Accessed 04/11/2013].

Thomas, G., 2007. *The Ford in Brockenhurst Village* [online]. Available from: http://www.geograph.org.uk/photo/344205 [Accessed 04/11/2013].

Woodland Trust, 2013. *Arable cereals* [online]. Available from: http://www.woodlandtrust.org.uk/ [Accessed 04/11/2013].

Appendix II: Supplementary material for Chapter 3

II.1. Participant information sheet

New Forest Visitor Survey (Summer 2015)

Project: Assessing Cultural Ecosystem Services in a Historic Natural Landscape: The New Forest

You are being invited to take part in a research project. Before you decide it is important for you to understand why the research is being done and what it will involve. Please take time to read the following information carefully and discuss it with others if you wish. Ask us if there is anything that is not clear or if you would like more information. Take time to decide whether or not you wish to take part.

What is the purpose of the project?

I am a PhD student at Bournemouth University, and I am studying recreational use of the New Forest. I am keen to understand what people do when they visit the New Forest, and what attracts them to the area. I am inviting a selection of visitors to carry a GPS unit with them on their visit so that I can find out where people go, and which parts of the New Forest they prefer to visit.

Do I have to take part?

It is up to you to decide whether or not to take part. If you do decide to take part you will be given this information sheet to keep (and be asked to sign a consent form) and you can still withdraw up to the point of the GPS unit being handed back to the survey. You do not have to give a reason.

How long does the survey and GPS carrying part of the research take?

The survey is short and should take no more than 5 minutes. You are asked to carry the GPS unit with you on your route through the New Forest, there is no minimum or maximum duration, and is guided by you. All we ask is that you finish your trip and return to the car park by 5pm the same day and return the GPS unit to the survey team. If this is not possible, you will not be able to take part on this occasion.

Will my taking part in this project be kept confidential?

All the information that we collect about you during the course of the research will be kept strictly confidential. You will not be able to be identified in any reports or publications.

What type of information will be sought from me and why is the collection of this information relevant for achieving the research project's objectives?

Your demographic information, your views and collection of GPS tracking data will allow us to determine what visitors to the New Forest value aesthetically and recreationally and where they visit.

Who is organising and funding the research?

This project is funded by Bournemouth University and the Natural Environment Research Council (NERC) under the Biodiversity & Ecosystem Service Sustainability (BESS) research programme.

Contact for further information

The supervisor of this research project is Prof. Adrian Newton, Professor and Director Conservation Ecology at Bournemouth University (anewton@bournemouth.ac.uk).

II.2. Consent form

New Forest Visitor survey and GPS research consent form

Project: Assessing Cultural Ecosystem Services in a Historic Natural Landscape: the New Forest

Researcher: Arjan Gosal, PhD student, agosal@bournemouth.ac.uk

Supervisor: Prof. Adrian Newton, Professor And Director Conservation Ecology, anewton@bournemouth.ac.uk

Please Initial Here

I confirm that I have read and understood the participant information sheet for the above research project and have had the opportunity to ask questions.	
I understand that my participation is voluntary and that I am free to withdraw up to the point where the data is anonymised, without giving reason and without there being any negative consequences. In addition, should I not wish to answer any particular question(s), or complete the GPS tracking research, I am free to decline.	
I give permission for members of the research team to have access to my anonymised responses. I understand that my name will not be linked with the research materials, and I will not be identified or identifiable in the report or reports that result from the research.	
I agree to take part in the above research project.	
I agree to return the GPS unit back to the survey team.	

_______________________ _________________ _______________________________

Name of Participant Date Signature

_______________________ _________________ _______________________________

Name of Researcher Date Signature

II.3. New Forest visitor survey

<u>New Forest Visitor Survey (Summer 2015)</u>

Surveyor	____________________________
GPS unit	____________________________
GPS Start time	____________________________
GPS end time	____________________________
Date	____________________________

As part of PhD research at Bournemouth University, we are studying recreational use of the New Forest. We are keen to understand what people do when they visit the New Forest, and what attracts them to the area. I am inviting a selection of visitors to carry a GPS unit with them on their visit so that we can find out where people go, and which parts of the New Forest they prefer to visit. I would be very grateful if you would help me in our research, by reading and signing the consent form and then completing the survey and carrying a portable GPS device during your visit.

1. What is your age?

☐ Up to 10
☐ 10-17
☐ 18-30
☐ 31-40
☐ 41-50
☐ 51-60
☐ 61-70
☐ 71+

2. What gender are you?

☐ Male
☐ Female

3. What type of area best describes the area you live in?

☐ City
☐ Town
☐ Village
☐ Countryside

4. How close do you live to the New Forest National Park?

☐ Within the National Park boundaries
☐ Within 10 km of the National Park boundaries
☐ Within 30 km of the National Park boundaries
☐ More than 30 km away from the National Park boundaries

If distance unknown, please provide your postcode or town of residence ________

5. How did you get here today?

☐ Car
☐ Foot
☐ Bike
☐ Train

Other ____________________

6. What is the main activity you are undertaking today?

☐ Dog walking
☐ Exercise (including jogging and running)
☐ Horse riding
☐ Outing with children/ family
☐ Walking
☐ Cycling
☐ Bird watching / wildlife watching
☐ Other ____________________

7. Please look at the following five images. You will see woodland in various stages of decline, as you may see on a walk through the New Forest. Please score each photo on a scale of 1 - 5, with 1 being very unappealing to 5 being very appealing for both aesthetic (visual) appeal and for recreational usage. Please circle your answers.

Image 1

for aesthetic (visual) appeal	1	2	3	4	5
for recreational usage	1	2	3	4	5

Please circle a score for each line (with 1 being very unappealing to 5 being very appealing).

Image 2

for aesthetic (visual) appeal	1	2	3	4	5
for recreational usage	1	2	3	4	5

Please circle a score for each line (with 1 being very unappealing to 5 being very appealing).

Image 3

for aesthetic (visual) appeal	1	2	3	4	5
for recreational usage	1	2	3	4	5

Please circle a score for each line (with 1 being very unappealing to 5 being very appealing).

Image 4

for aesthetic (visual) appeal	1	2	3	4	5
for recreational usage	1	2	3	4	5

Please circle a score for each line (with 1 being very unappealing to 5 being very appealing).

Image 5

for aesthetic (visual) appeal	1	2	3	4	5
for recreational usage	1	2	3	4	5

Please circle a score for each line (with 1 being very unappealing to 5 being very appealing).

8. Please score how important you think woodlands are for providing the following, on a scale of 1-5, with 1 being very unimportant to 5 being very important. Please circle your answer.

Conservation value
The conservation value is a measure of how important a habitat is for wildlife. It reflects how strongly you feel a habitat should be protected, or conserved, for the purposes of conserving wildlife.

1 2 3 4 5

Recreation value
The recreation value of a habitat can be described how important you feel it is for recreation including hiking, dog-walking, horse-riding, cycling and picnicking.

1 2 3 4 5

Aesthetic value
The aesthetic value of a habitat can be described as the value you place on a habitat for its beauty or visual appearance.

1 2 3 4 5

9. How concerned are you that mature beech woods of the New Forest are dying because of climate change on a scale of 1-5, with 1 being not at all concerned to 5 being very concerned? Please circle your answer.

1 2 3 4 5

10. In terms of their aesthetic (visual) value, do you prefer broadleaved trees or conifer trees?

☐ Broadleaved
☐ Conifer
☐ No preference

11. In terms of their aesthetic (visual) value, do you prefer smaller or larger (veteran) trees?

☐ Smaller
☐ Larger (veteran)
☐ No preference

Thank you

For survey team use only

On return of the GPS unit, tick the appropriate weather categories during the participant's route with the GPS unit (choose one from each column).

	Cloud cover		Temperature		Rain
	Sunny		Cold		No rain
	Sunny spells		Mild		Rain less than 1/4 of time
	Overcast		Warm		Rain less than 1/2 of time
	Windy		Hot		Rain more than 1/2 of time

Appendix III: Supplementary material for Chapter 5

III.1. Climate mapping reclassification categories

Table III.1.1: Habitat reclassification of LANDIS-II outputs used to create climate change maps of the New Forest National Park.

NVC classification	Non-native and heath classification	Raster value	Final classification category
<u>**Single classification**</u>			
-	Calluna vulgaris or Ulex europaeus	100	Dwarf shrub heath
-	Larch (*Larix decidua*) Grand Fir (*Abies grandis*) and Douglas-fir (*Pseudotsuga menziesii*),	200	Coniferous woodland
-	Sitka spruce (Picea sitchensis) and Norway spruce (Picea abies)	200	Coniferous woodland
-	Ponderosa Pine (*Pinus ponderosa*), Black pine (*Pinus nigra*) and Scots pine (*Pinus sylvestris*)	200	Coniferous woodland
-	Coast redwood (Sequoia sempervirens)	200	Coniferous woodland
-	Western Hemlock (Tsuga heterophylla)	200	Coniferous woodland
-	Lawson cypress (Chamaecyparis lawsoniana)	200	Coniferous woodland
-	Sweet chestnut (*Castanea sativa*)	300	Broadleaved, mixed and yew woodland
-	Hornbeam (Carpinus betulus)	300	Broadleaved, mixed and yew woodland
-	Sycamore (Acer pseudoplatanus).	300	Broadleaved, mixed and yew woodland
W1 Salix cinerea – Galium palustre woodland or W2 Salix cinerea – Betula pubescens – Phragmites australis woodland		1	Broadleaved, mixed and yew woodland

W8 Fraxinus excelsior – Acer campestre – Mercurialis perennis woodland	-	2	Broadleaved, mixed and yew woodland
W9 Fraxinus excelsior – Sorbus aucuparia – Mercurialis perennis woodland	-	3	Broadleaved, mixed and yew woodland
W10 Quercus robur – Pteridium aquilinum – Rubus fruticosus woodland	-	4	Broadleaved, mixed and yew woodland
W12 Fagus sylvatica – Mercurialis perennis woodland	-	5	Broadleaved, mixed and yew woodland
W14 Fagus sylvatica – Rubus fruticosus woodland	-	6	Broadleaved, mixed and yew woodland
W15 Fagus sylvatica – Deschampsia flexuosa woodland	-	7	Broadleaved, mixed and yew woodland
W13 Taxus baccata woodland	-	8	Broadleaved, mixed and yew woodland
W16 Quercus spp. – Betula spp. – Deschampsia flexuosa woodland	-	9	Broadleaved, mixed and yew woodland
W18 Pinus sylvestris – Hylocomium splendens woodland	-	10	Coniferous woodland
W21 Crataegus monogyna – Hedera helix scrub	-	11	Dwarf shrub heath (as a shrub community)
W22 Prunus spinosa – Pteridium aquilinum scrub	-	12	Dwarf shrub heath (as a shrub community)
W23 Ulex europaeus – Rubus fruticosus agg. scrub	-	13	Dwarf schrub heath
W12.1 fagusylv fraxexce acerpseu sorbaria coryavel cratmono	-	14	Broadleaved, mixed and yew woodland
<u>Combined classification</u>			
W1 Salix cinerea – Galium palustre woodland or W2 Salix cinerea – Betula pubescens – Phragmites australis woodland	Dwarf shrub heath	101	Broadleaved, mixed and yew woodland
W8 Fraxinus excelsior – Acer campestre – Mercurialis perennis woodland	Dwarf shrub heath	102	Broadleaved, mixed and yew woodland
W9 Fraxinus excelsior – Sorbus aucuparia – Mercurialis perennis woodland	Dwarf shrub heath	103	Broadleaved, mixed and yew woodland

W10 Quercus robur – Pteridium aquilinum – Rubus fruticosus woodland	Dwarf shrub heath	104	Broadleaved, mixed and yew woodland
W12 Fagus sylvatica – Mercurialis perennis woodland	Dwarf shrub heath	105	Broadleaved, mixed and yew woodland
W14 Fagus sylvatica – Rubus fruticosus woodland	Dwarf shrub heath	106	Broadleaved, mixed and yew woodland
W15 Fagus sylvatica – Deschampsia flexuosa woodland	Dwarf shrub heath	107	Broadleaved, mixed and yew woodland
W13 Taxus baccata woodland	Dwarf shrub heath	108	Broadleaved, mixed and yew woodland
W16 Quercus spp. – Betula spp. – Deschampsia flexuosa woodland	Dwarf shrub heath	109	Broadleaved, mixed and yew woodland
W18 Pinus sylvestris – Hylocomium splendens woodland	Dwarf shrub heath	110	Coniferous woodland
W21 Crataegus monogyna – Hedera helix scrub	Dwarf shrub heath	111	Dwarf shrub heath
W22 Prunus spinosa – Pteridium aquilinum scrub	Dwarf shrub heath	112	Dwarf shrub heath
W23 Ulex europaeus – Rubus fruticosus agg. scrub	Dwarf shrub heath	113	Dwarf shrub heath
W12.1 fagusylv fraxexce acerpseu sorbaria coryavel cratmono	Dwarf shrub heath	114	Broadleaved, mixed and yew woodland
W1 Salix cinerea – Galium palustre woodland or W2 Salix cinerea – Betula pubescens – Phragmites australis woodland	Coniferous woodland	201	Broadleaved, mixed and yew woodland
W8 Fraxinus excelsior – Acer campestre – Mercurialis perennis woodland	Coniferous woodland	202	Broadleaved, mixed and yew woodland
W9 Fraxinus excelsior – Sorbus aucuparia – Mercurialis perennis woodland	Coniferous woodland	203	Broadleaved, mixed and yew woodland
W10 Quercus robur – Pteridium aquilinum – Rubus fruticosus woodland	Coniferous woodland	204	Broadleaved, mixed and yew woodland
W12 Fagus sylvatica – Mercurialis perennis woodland	Coniferous woodland	205	Broadleaved, mixed and yew woodland
W14 Fagus sylvatica – Rubus fruticosus woodland	Coniferous woodland	206	Broadleaved, mixed and yew woodland

W15 Fagus sylvatica – Deschampsia flexuosa woodland	Coniferous woodland	207	Broadleaved, mixed and yew woodland
W13 Taxus baccata woodland	Coniferous woodland	208	Broadleaved, mixed and yew woodland
W16 Quercus spp. – Betula spp. – Deschampsia flexuosa woodland	Coniferous woodland	209	Broadleaved, mixed and yew woodland
W18 Pinus sylvestris – Hylocomium splendens woodland	Coniferous woodland	210	Coniferous woodland
W21 Crataegus monogyna – Hedera helix scrub	Coniferous woodland	211	Coniferous woodland
W22 Prunus spinosa – Pteridium aquilinum scrub	Coniferous woodland	212	Coniferous woodland
W23 Ulex europaeus – Rubus fruticosus agg. scrub	Coniferous woodland	213	Coniferous woodland
W12.1 fagusylv fraxexce acerpseu sorbaria coryavel cratmono	Coniferous woodland	214	Broadleaved, mixed and yew woodland
W1 Salix cinerea – Galium palustre woodland or W2 Salix cinerea – Betula pubescens – Phragmites australis woodland	Broadleaved, mixed and yew woodland	301	Broadleaved, mixed and yew woodland
W8 Fraxinus excelsior – Acer campestre – Mercurialis perennis woodland	Broadleaved, mixed and yew woodland	302	Broadleaved, mixed and yew woodland
W9 Fraxinus excelsior – Sorbus aucuparia – Mercurialis perennis woodland	Broadleaved, mixed and yew woodland	303	Broadleaved, mixed and yew woodland
W10 Quercus robur – Pteridium aquilinum – Rubus fruticosus woodland	Broadleaved, mixed and yew woodland	304	Broadleaved, mixed and yew woodland
W12 Fagus sylvatica – Mercurialis perennis woodland	Broadleaved, mixed and yew woodland	305	Broadleaved, mixed and yew woodland
W14 Fagus sylvatica – Rubus fruticosus woodland	Broadleaved, mixed and yew woodland	306	Broadleaved, mixed and yew woodland
W15 Fagus sylvatica – Deschampsia flexuosa woodland	Broadleaved, mixed and yew woodland	307	Broadleaved, mixed and yew woodland
W13 Taxus baccata woodland	Broadleaved, mixed and yew woodland	308	Broadleaved, mixed and yew woodland
W16 Quercus spp. – Betula spp. – Deschampsia flexuosa woodland	Broadleaved, mixed and yew woodland	309	Broadleaved, mixed and yew woodland

W18 Pinus sylvestris – Hylocomium splendens woodland	Broadleaved, mixed and yew woodland	310	Coniferous woodland
W21 Crataegus monogyna – Hedera helix scrub	Broadleaved, mixed and yew woodland	311	Broadleaved, mixed and yew woodland
W22 Prunus spinosa – Pteridium aquilinum scrub	Broadleaved, mixed and yew woodland	312	Broadleaved, mixed and yew woodland
W23 Ulex europaeus – Rubus fruticosus agg. scrub	Broadleaved, mixed and yew woodland	313	Broadleaved, mixed and yew woodland
W12.1 fagusylv fraxexce acerpseu sorbaria coryavel cratmono	Broadleaved, mixed and yew woodland	314	Broadleaved, mixed and yew woodland