



The
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**Evaluating the role of biodiversity, ecosystem services and
real-world management in the restoration of wetlands in multi-
functional landscapes**

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Declaration

The work described herein was conducted at The University of Sheffield between November 2012 and September 2016 under the supervision of Dr. Helen Moggridge, Professor Philip Warren, Dr. Liz Sharp and Professor Lorraine Maltby. This thesis and the work described within it was completed solely by the author and has not been submitted in whole or part for any other degree at this or any other institution.

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Abstract

Wetlands are important for supporting biodiversity and providing ecosystem services, which has provided impetus for landscape scale wetland restoration. Whilst the extensive restoration of wetlands is desirable, this is rarely feasible in multi-functional landscapes; rather, patches need to be integrated with other land uses and consider real-world factors. Much uncertainty exists over patch configuration and integration within the landscape, which raises issues for effective delivery. This study addresses this research gap with an examination of biodiversity and ecosystem services delivery in the Humberhead Levels (HHL) Nature Improvement Area (NIA) landscape. A combination of field data and models were used to map biodiversity and ecosystem services, alongside current wetlands, to develop a series of restoration options, based on physical suitability and optimising biodiversity and ecosystem services.

The influence of real-world factors on wetland restoration was also investigated, through examination of documentation and interviews with practitioners working within the NIA. The NIA restoration plan was compared with the restoration options developed, to understand the relative influences of biodiversity, ecosystem services and real-world factors on decisions regarding wetland restoration in the landscape.

Biodiversity provided founding principles for many decisions, but real-world factors, such as existing conservation work, agriculture and resources, exerted strong influences on final decisions. Ecosystem services was considered, but was less embedded in decision making, due to uncertainty and hesitation over its application. Our study highlighted the need for better evidence to inform decision making and further guidance on the application of ecosystem services. Real-world factors were influential and this needs to be more explicitly incorporated into landscape scale design and guidance. This study furthers understanding of how multi-functional landscapes can be designed to optimise biodiversity and ecosystem services, and other considerations that are needed for practical delivery, which is of value to the HHL and other landscape scale conservation projects.

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List of abbreviations

ARIES – ARTificial Intelligence for Ecosystem Services

BAP – Biodiversity Action Plan

BSR – Business for Social Responsibility

BOAs - Biodiversity Opportunity Areas

CCI - Cambridge Conservation Initiative

CEH – Centre for Ecology and Hydrology

CROW – Cycle Right Of Way

DEM – Digital Elevation Model

DO –Dissolved Oxygen

DTM – Digital Terrain Model

EcoServ – Ecosystem Services model

EEA - European Economic Area

GIS – Geographical Information System

HHL – Humberhead Levels

IPCC - Intergovernmental Panel on Climate Change

IDBs – Internal Drainage Boards

InVEST - Integrated Valuation of Ecosystem Services and Trade-offs

JNCC - Joint Nature Conservation Committee

KINEROS - KINEmatic wave overland flow, channel, Routing and erOSion model

LCM – Land Cover Map

LiDAR- Light Detection And Ranging

LNRs – Local Nature Reserves

LUCI - Land Utilisation and Capability Indicator model

LULC – Land Use Land Cover

MEA – Millennium Ecosystem Assessment

NBN – National Biodiversity Network

NCA – National Character Area

NGOs – Non-Governmental Organisation

NIA – Nature Improvement Area

NNRs – National Nature Reserves

OS - Ordnance Survey

PM- Programme Manager

PROW – Public Right Of Way

QUANGO - Quasi-Autonomous Non-Governmental Organisation

RSPB - Royal Society for the Protection of Birds

SCAs - Special Conservation Areas

SPAs – Special Protection Areas

SSSIs - Sites of Special Scientific Interest

TEEB- The Economics of Ecosystems and Biodiversity

TESSA – Toolkit for Ecosystem Services Site-based Assessment

WBCSD – World Business Council for Sustainable Development

Chapter One – Introduction and literature review: Issues of wetland restoration at a landscape scale

1.1 Restoration of Wetlands: Rationale and Issues

The official definition of wetlands from the 1975 Ramsar Convention is that they are “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt” (Hume et al., 2008). Figure 1.1 shows the global distribution of wetlands, illustrating how important they are as an ecosystem. Wetlands are typically found in the transitional area between terrestrial and aquatic environments (Dinnin and Van De Noort, 1999). They are highly adaptive and dynamic (Maltby, 1991; Turner et al., 2000), but also extremely vulnerable to change (Fletcher et al., 2011), making them complex to study. Unfortunately, many wetland areas, including those in the UK, are under harsh anthropogenic pressures and have been experiencing drastic destruction, fragmentation and habitat loss for some time (Feld et al., 2009). There are very few ecosystems that have not been affected by human activity (MEA, 2005), and globally up to 50% of wetlands have now been lost (Feld et al., 2009; Cui et al., 2012).

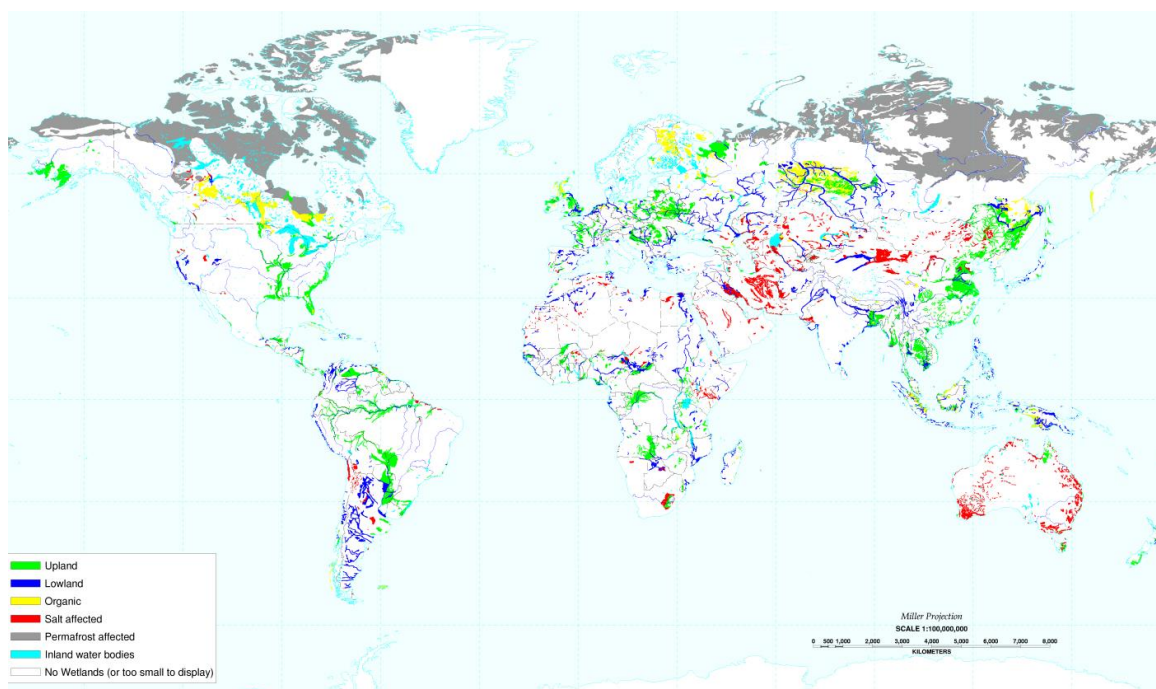


Figure 1.1: Global distribution of wetland environments (USDA, 1997).

Wetlands are essential environments for many reasons, including for biodiversity. Biodiversity was defined by the Convention on Biological Diversity (CBD, no date) as the “the variability among living organisms from all sources...and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems”. Biodiversity can increase the productivity of ecosystems (Al-Mufti et al., 1977), and wetlands can provide organisms that contribute to increasing biodiversity at a wider scale (Gibbs, 2000). Therefore, these environments create key habitats (UK National Ecosystem Assessment, 2011; Defra, 2012). However, biodiversity has been in decline for many years and in the last four decades the rate of loss has accelerated (Butchart et al., 2010).

As well as the intrinsic biodiversity value, wetlands also have many important physical characteristics, such as their ability to store water and therefore provide flood protection. Kayranli et al. (2009) state that wetlands also contain a large proportion of the global carbon stored in soils. There are many other similar functions that wetlands provide (Elmhagen et al., 2015b), and these are grouped together under the concept of ecosystem services. Ecosystem services is an important topic currently; a buzzword within both literature and policy (e.g. MEA, 2005; Maltby and Acreman, 2011; Norris, 2011; UK National Ecosystem Assessment, 2011). The concept is used as a way of displaying the benefits of an environment to society, with the hope that it will lead to humans taking more notice of their local environment and more responsibility for its protection. However, the processes that occur in a wetland, which underpin these services, are influenced by the characteristics and structure of the environment (Turner et al., 2000), which we do not yet fully understand.

Since wetlands have been identified as essential and are under harsh human pressures they require conservation and restoration. Conservation is defined as reducing or preventing change, whereas restoration is to mitigate against change and push the system into recovery of its historical state to increase composition and function (Swart et al., 2001; Hobbs et al., 2009). Traditional approaches for both include creating protected areas and conserving rare species (Goldman et al., 2008), which previously occurred at a site by site scale. However, these approaches appear to have been only partly successful (MEA, 2005), as we are still losing wetlands, levels of biodiversity and services, and so new approaches

are needed. The concept of ecosystem services is now playing a larger role and starting to appear in legislation, as it is seen as a way to better communicate issues around ecosystem loss, by showing what they provide for us (Goldman et al., 2008; Fisher and Brown, 2014; Eastwood et al., 2016).

Landscape scale work has also been suggested as an approach to overcome issues with wetland conservation. An agreed definition of landscape scale is difficult to find; Forman (1995) describes it as a mosaic of spatial units of patches, but there appears to be no agreement over the size it covers. The size appears to be context dependent rather than a known fact, but Forman's definition will be used for this project. From a nature conservation perspective, landscape scale refers to large-scale restoration/conservation that works between/outside of protected areas. By working at a landscape scale it could be possible to link already existing projects and to protect the biodiversity that already exists, which would extend out any successful conservation to a much greater scale, and so achieve more joined up thinking and actions (McKenzie et al., 2013). Landscape scale management appeared in the 1990s (Bennett and Mulongoy, 2006; Maltby and Acreman, 2011), but still more academic interest is needed to inform the process. Many groups do see the benefit of working at the landscape scale (Botequilha Leitão and Ahern, 2002). However, it is difficult to integrate the landscape approach into current policy, partly because the knowledge base is poor (McKenzie et al., 2013).

Actions for restoration decision making do not happen in isolation; landscape scale working is linked with biodiversity, ecosystem services and management, and the opportunities and constraints they provide for improving wetland functionality. Maltby (2010) has highlighted the need for more interdisciplinary work, in both academia and decision making, to bring together the different aspects and aid restoration decisions, as displayed in Figure 1.2. Figure 1.2 shows that biodiversity, ecosystem services and governance all interact when making restoration decisions, but how this occurs is relatively unknown, and will be investigated in this work. It does appear that restoration work is slowly becoming interdisciplinary (Jasonoff, 2003), but still more is needed to inform decisions on landscape scale restoration, including research on biodiversity and ecosystem services, which is the central aim of this thesis. Consequently, the rest

of this chapter will look at the issues surrounding biodiversity, ecosystem services and management in more depth, including with landscape scale work, and the links between the different aspects. The following discussion will outline the importance of the topic and the need for further attention, especially on how decisions surrounding conservation are made on the ground.

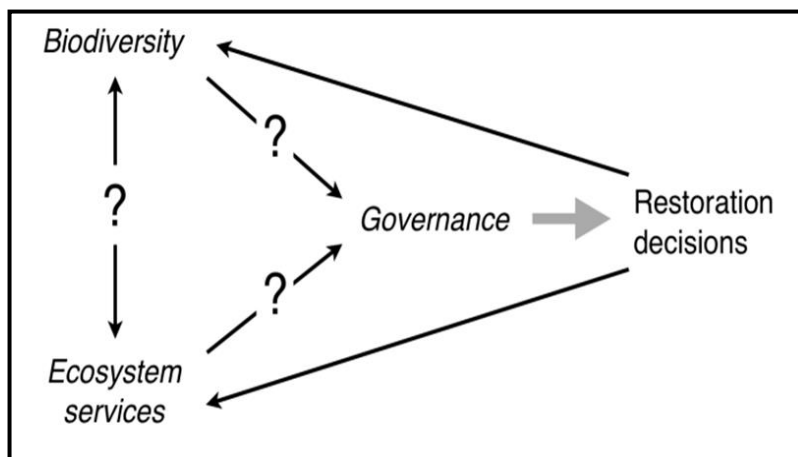


Figure 1.2: The factors and linkages to be considered for wetland restoration decisions.

1.2 Biodiversity

Biodiversity has both benefits to ecosystems and humans (MEA, 2005), but has been in decline for years (Butchart et al., 2010). The concept covers a range of issues and has many definitions, often generating controversy in the research community (Franklin, 1993; Henik and Kowarik, 2010). However, as stated previously, it was defined by the CBD (no date) as the “the variability among living organisms from all sources...and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems”. One debate is whether biodiversity is concerning species richness or abundance, but both can be important depending on the traits and processes required (Cardinale et al., 2012). In most cases, the literature considers the diversity of species, as will be used in this work, but there is also genetic, population, habitat, ecosystem and global diversity, with intrinsic value at all levels (TEEB, 2010; Pearson, 2016).

There is also diversity at different spatial scales, with alpha (diversity within a site), beta (between sites) and gamma diversity (at a regional scale). The lack of data, especially at the alpha level, means that data is often averaged or interpreted from another scale, leading to a false impression of biodiversity levels (Maskell

et al., 2013). Although typically wetlands do not support high alpha diversity, because of local conditions, they can provide species that contribute to beta and especially gamma diversity by providing variation (Gibbs, 2000, Maskell et al., 2013). Therefore, by themselves wetland sites do not tend to high productivity, because of the low alpha diversity (Al-Mufti et al., 1977, Maskell et al., 2013). Levels of biodiversity in an area are effected by the potential productivity of that site, and the levels of disturbance (Al-Mufti et al., 1977, Maskell et al., 2013).

Biodiversity plays a very important role in ecosystem functioning, which in turn underpins ecosystem services, highlighting the strong link between the two concepts (Norris, 2011; Van Oudenhaven et al., 2012). Much of the previous literature dealt with the mechanisms linking biodiversity and ecosystem functionality, and now the link with the concept of ecosystem services is starting to play a bigger part. Mace et al. (2012) suggests that the biodiversity discipline is currently in flux because of this rising role of the concept of ecosystem services, but further research can hopefully lead to a better understanding of the role of biodiversity and show its continued importance as a distinct concept.

One example of the importance of biodiversity is that of a study by Cardinale et al. (2012), who believed that higher plant diversity increases the resistance to pressures, such as invasive species. Invasive species, such as Himalayan Balsam, are found in many wetland areas (Smart et al., 1986; Hume et al., 2008), and are a landscape scale issue. Biodiversity is said to overcome the issue of invasive species and other pressures because organisms have particular traits that influence the functions of an ecosystem, and if there is a greater diversity of species then more traits should be available. Interactions between organisms also create more processes, displaying again the importance and complexity of species diversity (UK National Ecosystem Assessment, 2011). Thus, increased biodiversity should act as a buffer against change (Mace et al., 2012), which is important considering that many papers (e.g. Piper et al., 2006; Gonzalez et al., 2011) highlight the potential future pressure of climate change, especially to wetland environments.

Biodiversity affects humans through “security, resiliency, social relations, health and freedom of choices and action” (MEA, 2005). One particular paper that looks

at the links between biodiversity and humans is that of Cardinale et al. (2012). They discuss that biodiversity influences ecosystem functions, in turn impacts on what an ecosystem can provide for us as humans, but that the relationship is extremely complex (Cardinale et al., 2012). Also, it is often not clear to the public what benefits biodiversity provides, especially in wetland environments. Therefore, biodiversity is important to humans, but more academic work is needed to understand how it works, which is partly why the concept of ecosystem service is becoming more popular.

Biodiversity has been in decline for many years, but in the last four decades the rate of loss has accelerated, with habitat fragmentation increasing (Butchart et al., 2010). Land use change, climate change, pollution, invasive species and overexploitation are all causing biodiversity to decline, and the costs and risks associated are only going to increase (MEA, 2005). Many papers (e.g. Peacock, 2003; Butchart et al., 2010), state that as habitats become smaller and more disconnected, rare and specialised species, and landscape scale patterns are being lost, therefore reducing biodiversity. There was a global aim to halt biodiversity loss by 2010, but many papers and reports (e.g. Lawton et al., 2010; Larsen et al., 2014) state that this was not achieved. The Environment Agency (2009) suggests that this was because not enough was known about how to address the problem. There is now an apparently more realistic aim of the EU 2020 Biodiversity Strategy, with much clearer targets and methods (Defra, 2011). This strategy aims to halt the “loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss” (European Union, 2011). However, academic attention is needed to assist with meeting this aim and the protection of biodiversity in general.

1.2.1 Biodiversity conservation

There are many traditional approaches to biodiversity conservation, such as hotspots and rare species protection (Fisher and Brown, 2014), which have resulted in small protected areas and endangered species management (Poiani et al., 2000). It been identified that these are not enough, more needs to be done

globally to protect biodiversity, on the ground and in policy (MEA, 2005; Fisher et al., 2011). Most activities have previously been at a site by scale, which has been identified as insufficient to halt the degradation of land and the loss of biodiversity (MEA, 2005; Verburg et al., 2016). Problems with protected areas include that there was, and still is to a degree, debate over what is the right size, location and type to conserve biodiversity (Tjorve, 2010). Also, Larsen et al. (2014) believe that even though there is a global coverage of protected areas the area covered is not enough, and many core areas are not actually included. The idea now is to make these conservation attempts more holistic and physically connected, as will be discussed in the following section.

The issues around biodiversity conservation are partly why landscape scale working and the concept of ecosystem services are becoming more prominent. However, a significant section of the literature believes that the conservation of biodiversity should be the main reason ecosystem restoration, but is not currently occurring. Wu (2006) suggests that biodiversity management is another research area moving towards a more anthropocentric view, with the question often being asked of what can biodiversity do for humans, rather than the importance of biodiversity as its own entity; “new conservation” (Pearson, 2016). Therefore, biodiversity is often protected due to human choices and livelihoods rather than intrinsic value (Zhang et al., 2010; Reyers et al., 2012b). For example, by looking at functional biodiversity, rather than structural, the focus becomes on the economic gains, which makes humans more likely to protect biodiversity (Bengtsson, 1998). However, to a point, human based judgements is what is already occurring with protected areas and species-specific conservation. Value judgements have to be made about where or what to protect, which will always be connected to economic and social issues. This anthropogenic-driven conservation is more connected to the concept of ecosystem services, which is essentially driven by the needs of humans.

1.2.2 Landscape scale working/Ecological networks

Landscape scale work now appears to be a very important part of this research field in attempting to restore wetland functionality as a replacement for the

traditional unsuccessful approaches. However, it is still a research area in its infancy, as was discussed in Section 1.1. Landscape ecology is the study of how the structure of the environment on a larger scale is affected by the diversity and abundance of organisms (Opdam et al., 2002), and the complexities it creates across a landscape (Wu, 2006). Species distribution is dynamic and will vary across a landscape (Lawton et al., 2010) and as the scale alters, diversity is likely to change (Peacock, 2003), both of which are important considerations. By using a landscape scale approach to protect biodiversity it is hoped that loss will be reduced or even halted, because it allows more resources and habitats to be linked together and a greater proportion of species to be under protection (Otte et al., 2007).

The need for the landscape scale approach has been addressed in academic and policy literature. One major document is the independent review “Making Space for Nature” (Lawton et al., 2010). This stated that “we have to allow more space for nature”, because what has occurred previously, mostly at a smaller scale, has not been enough (Lawton et al., 2010). Lawton et al. (2010) argue that we need large-scale restoration, thinking about both biodiversity and ecosystem services. As a result the document “ThinkBIG” was produced (English Biodiversity Group, 2011). The document highlights that landscape scale conservation is the best way to achieve multiple benefits to wildlife and people, by integrating a range of land uses sympathetically (English Biodiversity Group, 2011). This idea of multi-functional landscapes is also covered in the UK National Ecosystem Assessment (2011) through using ecosystem services to maintain quality of life. Therefore, policy documents appear to have been reacting to the literature call for more landscape scale work, recognising its importance and attempting to fit it into policy.

It is identified in many studies that an important component for making biodiversity decisions at a landscape scale is connectivity (Pringle, 2003; Piper et al., 2006; Lawton et al., 2010; Gonzalez et al., 2011). Connectivity is defined as the “degree to which the landscape facilitates or impedes movement among resource patches” (Taylor et al., 1993), and wetlands are highly dynamic with many important connections (Amezaga et al., 2002). Connectivity allows dispersal for foraging, breeding, migration and adaption to climate change

(Welborn et al., 1996; Finlayson, 1999; Peacock, 2003; Catchpole, 2006; The Wildlife Trusts, 2007; Hume et al., 2008). No single habitat patch will provide everything that an organism requires (Ma et al., 2010; Lookingbill et al., 2010), and so they also rely on the functionality and resources of connected habitats (Amezaga and Santamaria, 2000; Roe and Georges, 2007). Habitat loss leads to smaller patches and greater distances between them, making it harder to locate suitable environments (Kindlmann and Burel, 2008), which needs to be considered designing a landscape (Peacock, 2003; Pringle, 2003; Piper et al., 2006; Lawton et al., 2010; Gonzalez et al., 2011; Cui et al., 2012). Patches do not have to be physically connected, but the probability of movement decreases with increasing distance and with increasingly poor quality intervening land (Attum et al., 2008; Roe et al., 2009). Therefore, connectivity with the surrounding land uses is also important and so integrated management approaches are needed at a larger scale (Amezaga and Santamaria, 2000; Verhoeven et al., 2008).

Many papers highlight the fact that connectivity is dynamic and changes across spatial and temporal scales, and each species will respond individually (Welborn et al., 1996; Haig et al., 1998; Gonzalez et al., 2011; Baguette et al., 2012). Also, both current and historic connectivity play out in a landscape, as Lindborg and Eriksson (2004) found that species richness and density was positively correlated with connectivity in the landscape from 50 years ago. Therefore, what happens to the landscape now could have big implications in terms of landscape connectivity in the future. There are disadvantages identified with connectivity, including: spreading disease, predators, fires, human activity, lead poisoning, high costs and unexpected gene flow, but these are often ignored (Simberloff and Cox, 1987; Jordan, 2000; Van der Windt and Swart, 2008). However, in general it is a useful and important concept for considering the restoration of wetland functionality.

Landscapes are susceptible to habitat fragmentation, which is defined as “a catch-all term for a multi-scale process that alters habitat isolation, quality, arrangement and connectivity”, and is not spatially uniform (Gonzalez et al., 2011). The concept of fragmentation comes from island biogeography and can include: a reduction in habitat amount, an increase in the number of patches, a

decrease in the size of patches and an increase in the isolation of patches (Fahrig, 2003). Drastic changes due to anthropogenic pressures has significantly increased fragmentation (Kindlmann and Burel, 2008; Mitchell et al., 2013). Theoretical and empirical studies predict that habitat loss and fragmentation contributes to local population extinctions (Kindlmann and Burel, 2008; Baguette et al., 2012), as organisms will spend more time moving between habitats in matrix areas of lower quality (Moilanen and Hanski, 2001). However, there is still limited knowledge about the way different species behave, what influence landscape structure will have on them and what level of fragmentation may cause thresholds of population persistence to be overcome (Belisle, 2005; Gonzalez et al., 2011). The interaction with climate change is also likely to have further adverse effects (Gonzalez et al., 2011).

To reduce the negative consequences of fragmentation and encourage connectivity, the concept of ecological networks has been suggested as a landscape scale approach for restoration in a human dominated landscape, because it is not feasible to restore everywhere (Bennett and Mulongoy, 2006; Lawton et al., 2010; Harrington et al., 2010). This approach first appeared in the 1970s and is now becoming global (Lawton et al., 2010). Figure 1.3 (Defra, 2011) depicts an ecological network. There are four key elements to Figure 1.3: core areas, restoration areas, corridors and stepping stones, and buffer zones (Lawton et al., 2010; Defra, 2011). Lawton et al. (2010) also suggest that sustainable land use areas around ecological networks are important.

The core areas are already existing wetland areas with habitat that supports a wide range of species, and therefore already have high ecological quality (Harrington et al., 2010), or important ecosystem services (Lawton et al., 2010; Defra, 2011). Often they are designated as a protected area (Harrington et al., 2010), but are always areas where the conservation of biodiversity is the main priority (Bennett and Mulongoy, 2006). They form the heart of the network and provide space for species to thrive and disperse to other elements (Lawton et al., 2010). Core areas for the future also need to be created to increase the provision of space and resources, and so creating the restoration areas (Lawton et al., 2010; Defra, 2011). These are often situated close to core areas to enhance them or are at least connected (Lawton et al., 2010).

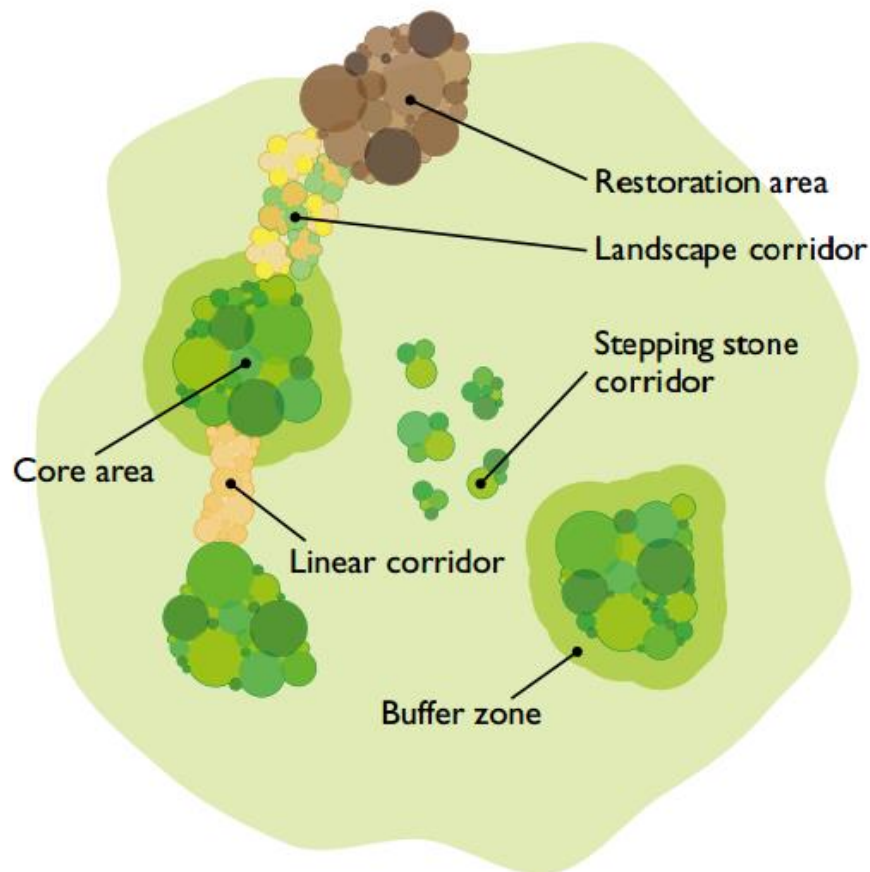


Figure 1.3: The components of ecological networks (Defra, 2011).

With the addition of a buffer area (or transitional zone) the other elements become increasingly protected from surrounding activities in the wider environment (Harrington et al., 2010; Lawton et al., 2010; Defra, 2011; Cui et al., 2012), increasing the likelihood of continued conservation. Land use in these areas are restricted to those that are compatible with conservation (Bennett and Mulonguy, 2006). However, by restricting what can happen in this zone costs are imposed on the land owners and users (Bennett and Mulonguy, 2006). Also, Bennett and Mulonguy (2006) identified that it is difficult to decide on is an appropriate “safe” land use for a buffer area) and in some environments buffer areas are not practical (Roe and Georges, 2007). Land-use management is therefore important for effective conservation (Bennett and Mulonguy, 2006).

The different elements are linked by corridors to allow species to move between them (Harrington et al., 2010; Defra, 2011); often rivers (Lindenmayer and Nix, 1993; Lawton et al., 2010), hedgerows or tunnels (Bennett and Mulonguy, 2006).

The stepping stones also provide a corridor type approach, as long as the distance between them is not greater than species can move (Rodriguez-Iturbe et al., 2009). Van der Windt and Swart (2008) define a corridor as “a physical or biological strip connecting areas that allows the movement of species”. The movement allows for foraging, dispersal, migration, genetic exchange and to move away from poor or degrading habitat (Bennett and Mulongoy, 2006). However, it can also spread disease (Bennett and Mulongoy, 2006). According to Tjorve (2010), corridors increase the likelihood of individual survival for the majority of species, but we still do not fully understand the complex interactions between the organisms and the landscape, due to the lack of theoretical and empirical foundations (Van der Windt and Swart, 2008). Also, creating a corridor is not simple, what looks to be connected to the human eye may not be connected for other organisms; each has different requirements, and in some cases corridors can act as sinks or barriers (Hess and Fischer, 2001).

Sustainable use areas are the areas in the wider landscape that cover appropriate sustainable economic activities and consider ecosystem services, to make the environment less hostile to wildlife (Bennett and Mulongoy, 2006; Lawton et al., 2010; Defra, 2011). They have a similar role to buffers, but are less defined areas and allow a greater variety of land uses (Lawton et al., 2010). Overall, it is the whole network that is important, as species have different requirements (Lindermayer and Nix, 1993), and so often respond differently to the individual elements, but more research is needed on their development to inform restoration decisions.

Creating ecological networks is a less damaging approach to conservation than many traditional engineering solutions have been in the past (Peacock, 2003). More traditional methods of management, which often involve hard, more permanent engineering solutions, can destroy habitats and the chances of recolonisation (Peacock, 2003). Creation of softer schemes is reversible and so does not constrain future options for land use (Hume et al., 2008). However, it does create a human led landscape (Pringle, 2003). The concept of ecological networks has come from land use planning theories, but is not widely applied in conservation (Cui et al, 2012), but could be very useful in restoring wetland functionality at a landscape scale. Before designing a network, it is important to

consider the required function, for which landscape managers will have to work with the scientific community (Hess and Fischer, 2001). Some researchers are so sceptical about the concept that they are not recommending its usage in practice (Van der Windt and Swart, 2008), but there does not appear to be many other options for landscape scale conservation available at this time.

1.2.3 Synthesis/Research gaps

Biodiversity has great benefits to wildlife, environmental functions and human health, and so there is a clear rationale for its protection. It is also clear that biodiversity is being lost globally, partly as we lose more wetland habitats. However, it is still uncertain how to halt this loss. Landscape scale work has been suggested as an approach to halt biodiversity loss; creating connectivity across different patches, rather than trying (and inevitably failing for a number of reasons) to restore great swathes of land to its former state, which is the classic way to inform restoration locations. However, how to deliver landscape scale work was identified previously as a question that still remains unanswered in both academia and policy. Another approach for conserving biodiversity is through the concept of ecosystem services. There is some doubt over using the concept as amongst other issues it is felt that it detracts from biodiversity, but this will be considered further in the following section. Overall, biodiversity is an extremely important concept, but there are still many unknowns, such as how to achieve it on a landscape scale, and so more research is needed, which will be attempted by this work. Maltby and Acreman (2011) believe that wetlands provide a good opportunity to study biodiversity, as they are a diverse ecosystem containing a wide range of species.

1.3 Ecosystem services

Ecosystem services are defined as the “benefits provided by ecosystems that contribute to making human life both possible and worth living” (UK National Ecosystem Assessment, 2011), examples of which can be seen in Figure 1.4. It is a continuously growing research area, sometimes termed ecosystems functions or ecosystem benefits in the literature (English Nature, 2006), used in essence as a concept to describe and explain ecosystems (Fisher et al., 2008).

The study of ecosystems previously focused on the conservation of individual species, so was more biodiversity driven, but since the 1970s and 1980s the concept of ecosystem services has become much more popular (MEA, 2005; Luck et al., 2009; Maltby and Acreman, 2011; Norris, 2011; Bull et al., 2016). This popularity is partly because of the continuing loss of biodiversity and global problems with poverty (Posner et al., 2016b). Several studies (e.g. Rey Benayas et al., 2009; Maltby, 2010) have highlighted the shift from just classifying environments, to researching what services they can provide, in order to encourage more ecosystem protection. More recent literature has also incorporated an understanding of the element of disservices, where the ecosystem causes problems for humans (Sandbrook and Burgess, 2015). As a concept, ecosystem services, shows how human society depends on nature and that we define it “in anthropocentric terms” (Fisher and Brown, 2014; Silverton, 2015). It is now the “dominant paradigm framing research and policy makers in biodiversity, ecology and conservation biology” (Silverton, 2015).

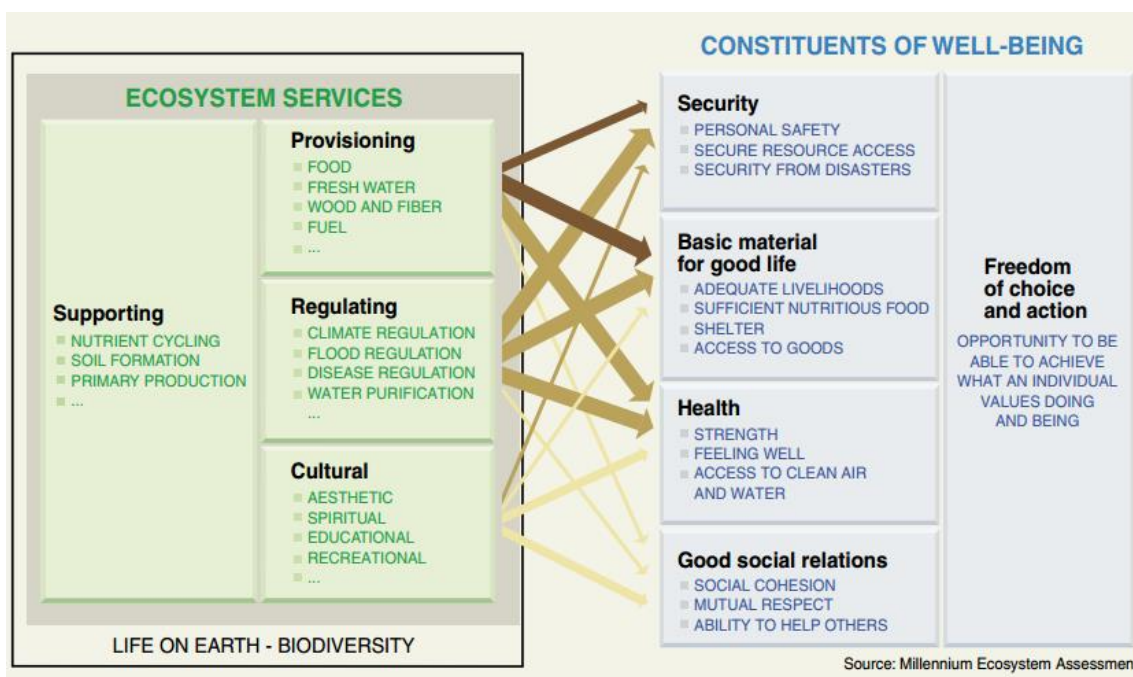


Figure 1.4: Ecosystem services, their categories and links to human well-being (MEA, 2005).

Ecosystem services derive from the natural functions and processes within the environment (Catchpole, 2006), that directly or indirectly produce goods and services (Di Sabatino et al., 2013). However, the natural capital to produce ecosystem services is declining (Shapiro et al., 2015). Ecosystem services are not uniform or stationary (Syrbe and Walz, 2012), but the details are largely

unknown (Mitchell et al., 2013). Individual ecosystems produce different services; the greatest differences often observed between urban and rural environments (Larondelle and Haase, 2013). More protection is needed of ecosystems if this concept is to be utilised. However, there are still many unknowns, such as the spatial distribution, which are stalling integration into policy and use on the ground.

Ecosystem services can be split into four categories: regulating (benefits from the regulation of processes), provisioning (products obtained from ecosystems), cultural (recreation and aesthetic experiences) and supporting (services needed for the production of all others) (see Figure 1.4) (Luck et al., 2009; Harrington et al., 2012). Wetlands have many physical and biological characteristics that can provide a wide range of ecosystem services. These particularly include: oxygen production, nutrient cycling, carbon storage, flood risk mitigation, education and recreation (English Nature, 2006). Wetlands therefore provide services from all four categories, many of which safeguard livelihoods (Tanner et al., 2013). In the literature, carbon storage is an ecosystem service often discussed in association with wetlands and they are also renowned for providing little food or timber (Pan et al., 2013; Tanner et al., 2013; Van der Biest et al., 2014). It is important to identify the most important ecosystem services for a particular landscape (Peh et al., 2013), as not every service can always be investigated.

The sheer amount of ecosystem services that exist is one reason why there is a large amount of literature on this subject, but perhaps also why there are still many unknowns. There has been much research on regulating and provisioning services, but less on the other two (Fisher et al., 2008; Quijas et al., 2012; Cardinale et al., 2012). Supporting services provide the basis for all other ecosystem services, as they are essential for the ecosystem processes and functions (Wu, 2013), and so therefore more research is needed. Norris (2011) has stated that most of the current management strategies focus on provisioning services, often at the expense of other types of ecosystem services.

Whether something is regarded as a service or not depends on the location, time, scale and perception of it, and is based on preferences, demands, needs and values (English Nature, 2006; Willeman et al., 2013). There is a difference between the supply and demand of ecosystem services, with the supply side

more involved in the biophysical world and the demand side concerning socio-cultural and economic issues (Martin-Lopez et al., 2014). Therefore, the need for the integration of science and society is becoming more widely recognised (Potschin and Haines-Young, 2013).

Ecosystem services are often studied on a landscape scale. Indeed, Wu et al. (2013) talk about the broader concept of landscape services in order to highlight the larger scale and to integrate natural and artificial systems. There is a hope that both concepts will lead to greater conservation and protection of natural environments (Stewart and Downing, 2008), but as identified previously they are not yet fully integrated into policy and decision making. There are still many unknowns about ecosystem services, including basic questions, such as “where are ecosystems producing benefits?” (Bagstad et al., 2013a) and what is the spatial distribution? (Wu, 2013). The interactions between land use dynamics and ecosystem services are also still largely unknown, but changes to the former could have a big impact on the supply of the latter (Larondelle and Haase, 2013). However, typically ecosystem services are difficult to observe and there is a lack of monitoring data (English Nature, 2006), and so it will be a big challenge to answer some of the unknowns.

1.3.1 Ecosystem services and biodiversity

It is well documented in the literature that there is evidence of a relationship between biodiversity and ecosystem services (Loreau et al., 2001; Balvanera et al., 2006; Rey Benayas et al., 2009; Norris, 2011; Van Oudenhaven et al., 2012). However, it is extremely difficult, if not currently impossible, to quantify this relationship as the research is still in its infancy (Luck et al., 2009; Norris, 2011; UK National Ecosystem Assessment, 2011). Also there is a separate, but often confusing, debate over whether biodiversity is a cultural ecosystem services (Fisher and Brown, 2014; Jax and Heink, 2015), but it will be considered as a separate concept for this thesis. The basic idea of the relationship is that organisms have certain traits that produce ecosystem processes and functions, which underpin the ecosystem services and create the landscape composition (Schneiders et al., 2012; Jones et al., 2013). Luck et al. (2009) suggested that

greater biodiversity provides a buffer to change for ecosystem services, as there will be an increased chance of a species occurring with the required functional trait. Therefore, the greater the biodiversity, the greater the possibility of increased ecosystem services in a landscape.

However, the relationship between the two concepts is actually more complex. Some ecosystem services require specific organisms, whereas others work with many, but this is still little understood, and so there is an argument for protecting everything in order to be precautionary (Schneiders et al., 2012). Reyers et al. (2012a) states that there are particularly strong links between regulating services and biodiversity that lead to win-win situations for both, but that in the case of protecting provisioning services, biodiversity often loses out. Also, Naidoo et al. (2008) found that when the focus was solely on the conservation of biodiversity it did not in general help to conserve optimal levels of ecosystem services, but there were some win-win cases. There are very few studies that look at the spatial relationships between what areas are optimal for ecosystem services and what are optimal for biodiversity (Anderson et al., 2009), especially for wetland environments. Some publications (e.g. Fisher et al., 2008) suggest that ecosystem services and biodiversity are positively related, so management for one should enhance the other, but Bullock et al. (2011) state that this should not be assumed. Also, Eastwood et al. (2016) discovered from the literature that sometimes there is a good association between the two, sometimes a bad association and sometimes no association at all. Therefore, it is unclear what the relationship is between the two concepts (Jax and Heink, 2015), but that compromises will often need to be made if attempting to conserve for both.

As the previous and current approaches to conserving biodiversity have not had much success it is now sometimes being recognised under the concept of ecosystem services in the hope that the link with human well-being and economics will increase interest and action (Farber et al., 2002; Mace et al., 2012). As identified previously there can be some areas and occasions when conserving for ecosystem services can also mean conserving for biodiversity, but Rey Benayas and Bullock (2012) believe that biodiversity often loses out because it has different restoration requirements to ecosystem services and now is often seen as less important by humans. Many in the biodiversity community feel that

the concept of ecosystem services is a distraction from biodiversity conservation (Reyers et al., 2012a) and it is better protected on its own. However, more protection is needed if we want to continue to receive the functions they deliver (Mitsch and Gosselink, 2000). Also, we do not know what the threshold is of biodiversity that below which we would lose functions (Swift et al., 2004). Reyers et al. (2012a) suggest that the links that exist between biodiversity and ecosystems services could have potential in trying to face the sustainability challenges of the future, and deal with landscape scale projects, such as wetland restoration. However, any management decisions made using the concept of ecosystem services need to be carefully thought through, as any changes could potentially propagate back to the supporting biodiversity (CCI and Birdlife International, 2011).

1.3.2 Ecosystem service management

Due to the increasing popularity of the concept of ecosystem services, there has been a rise in ecosystem services in management, because it is seen as an approach for planning and decision making, to bridge the gap between ecology and economics (Braat and De Groot, 2012; Verburg et al., 2016). It has also been shown that ecosystem service projects are good at securing funding (Goldman et al., 2008) and so they could be a useful way to get more money into the natural environment. There is a hope that the concept will lead to the greater conservation of natural environments (Stewart and Downing, 2008), especially in the case of wetlands. The management of biodiversity and ecosystem services often comes under the same policy making decisions, but normally it is the final service that is focused upon, rather than the underpinning processes (Mace et al., 2012). The concept has a very anthropogenic view, used as a global communication tool to increase interest in the quality of degrading ecosystems by presenting people with what they could lose out on (Fisher et al., 2008; Koschke et al., 2012). Management strategies to protect ecosystem services vary depending on the service, but examples include coupling services, such as recreation, with biodiversity conservation, giving them a value or creating market incentives (Willeman et al., 2013).

There has been a rise in policy that involves the concept, such as the Nature Improvement Area Programme (Defra, 2011) and the EU Biodiversity 2020 Strategy, with targets that include mapping ecosystem services (Maes et al., 2016). Decision makers are recognising that solutions to human based problems that use nature (e.g. wetlands for flood protection), could be better than technical solutions (Maes et al., 2012). However, Koschke et al. (2012) stated that ecosystem services are not yet fully integrated into policy and decision making. In order for this to occur there needs to be more information for decisions from research (Posner et al., 2016a). However, the science behind the concept, in terms of location and quantification, is still not fully understood (Villa et al., 2009; Nelson et al., 2009). Questions, such as where are the services providing benefits and who are the people that use them, still remain unanswered (Bagstad et al., 2013a). The complexity of human behaviour adds further complications (Bagstad et al., 2013a).

An important report in terms of ecosystem services management in the UK is the UK National Ecosystem Assessment (2011). One of the main points from the report is that natural resources are under-valued and are not included appropriately in decision making, leading to the situation now where approximately 30% of ecosystem services are in decline and more are already degraded (UK National Ecosystem Assessment, 2011). Also, the report discusses the fragmentation of wetlands, which is altering the provision of ecosystem services (UK National Ecosystem Assessment, 2011). The majority of the public are not familiar with the concept of ecosystem services and therefore it is difficult to demonstrate the benefits (UK National Ecosystem Assessment, 2011). All three of these points need improvement if the concept is to be fully implemented into policy, with a range of stakeholders involved within partnerships for good communication and increased knowledge. The report also suggests that research is currently focused on economic issues (UK National Ecosystem Assessment, 2011), which needs to change if we are to drive landscape scale restoration forward and implement the concept into policy better. Robust methods to define and quantify ecosystem services are needed, which can then be used for defensible decision making (Crossman et al., 2013).

A major issue in the literature is trade-offs, where services are negatively correlated (Tallis and Polasky, 2009; HHL Partnership, 2011; Pan et al., 2013), often as the result of management decisions (Wu et al., 2013). Trade-offs occur as services are not independent, compete for space, are mechanistically linked or have very different requirements (Lautenbach et al., 2012; Richards et al., 2015). They can lead to the degradation of some ecosystem services in the restoration of others (Rey Benayas and Bullock, 2012), and Maltby and Acreman (2011) and Martin-Lopez et al. (2014) suggest that often regulating services miss out to provisioning services. Many regulating services are vital, such as flood mitigation, to wetland environments, but they are harder to visualise and so are often overlooked. Buckhard et al. (2012) has stated that when trade-offs occur decisions need to be transparent and driven by socio-ecological information. Using landscape scale scenarios in order to identify and overcome trade-offs has been highlighted as an approach that will also allow synergies, defined as the “phenomena that occur when multiple services are enhanced simultaneously” (Pan et al., 2013), to be better understood (Opdam et al., 2002; CCI and Birdlife International, 2011). However, most studies so far have found that there are few ecosystem service synergies (Jones et al., 2013).

Linking to the idea of ecological networks, Schneiders et al. (2012) have suggested three focal zones for environmental management that leads to the provision of ecosystem services, and also the protection of biodiversity, which can be used to restore wetland functionality. They suggest small core zones for restoration that strive to achieve high ecological status, which are then surrounded by rural areas that focus on the sustainable provision of ecosystem services, and finally built up areas that strive towards lowering their ecological footprint (Schneiders, et al., 2012). This seems a very practical way of managing a large scale environment, taking ecosystem services, and biodiversity, into consideration, but would require a great deal of work, time and financial backing to achieve. Egoh et al. (2008) has suggested that focusing on ecosystem service hotspots might be the best way to manage a landscape, whilst keeping resources and efforts lower.

The value of ecosystem services is often important in management decisions, and can be expressed in economic gains, access to resources, social valuation

or contribution to human well-being, but most commonly tends to be expressed economically (Farber et al., 2002; Posthumus et al., 2010, Acreman et al., 2011). Economists really first attempted to value the natural environment in the 1960s and 1970s, seeing it as “natural capital” (Lawton et al., 2010), with mainly provisioning services involved (Daily et al., 2009). This research area continues to be very popular in current literature as the concept of ecosystem services rises, with many methods of economic valuation existing (Luck et al., 2009; Nelson et al., 2009). There is a large amount of literature that attempts to put an economic value on a single ecosystem service, in order to provide a reason for its protection, but services vary and will not be found in isolation. Economic valuation is also often used to overcome trade-offs (English Nature, 2006; TEEB, 2010), but Buckhard et al. (2012) suggests this could lead to groups with little scientific knowledge becoming involved in the valuation of ecosystem services that have little obvious economic value losing out.

One of the major papers to look at the issues surrounding ecosystem service valuation is Contanza et al. (1997). One of the main points is that a value is just a very crude starting point to highlight the need for further research, analysis and debate, but is often used for much more (Contanza et al., 1997). Valuation is just a static snapshot of a complex system, and so the ecosystem services are not fully captured in many of these quantifications (Contanza et al., 1997). The issue of complexity is further highlighted, as in most cases there is not a one to one relationship between functions and ecosystem services (Contanza et al., 1997). Overall, Contanza et al. (1997) conclude that valuation is not a good idea and that protection should be for moral or aesthetics reasons.

Van Oudenhaven et al. (2012) identified that different perceptions of monetary value amongst different groups has the ability to change decision making, and so therefore another method for valuing or understanding ecosystem services is needed. However, it is very difficult to demonstrate non-financial benefits and change thinking, especially with land owners (UK National Ecosystem Assessment, 2011). The problem with monetary valuation is highlighted further with cultural services, because these are virtually impossible to put an economic value on (Tanner et al., 2013), and so Frank et al. (2012) believes they are often forgotten about in current decision making. It becomes even more complex with

the question of how to compare a cultural service, such as recreational activity, to a regulatory service, such as carbon sequestration. This causes complex decisions, as both are important in wetlands, further displaying issues with economic valuation of ecosystem services.

Another economic issue to consider with ecosystem services, is the incentives given to put them into practice, of which there are few viable options currently in place (Tallis and Polasky, 2009). There are some incentive schemes, such as the Higher Level Stewardship, but Lawton et al. (2010) states that these are not available to everyone, and so further policy work is needed to make the system more accessible and transparent (TEEB, 2010). Also, the practice is normally on a farm by farm basis (Hodge and McNally, 2000), which is unsupportive of landscape scale working. Cardinale et al. (2012) believe that payments need to be based on the understanding of the links between biodiversity, functioning and ecosystem services, not just the economic value, with moral decisions also included (English Nature, 2006). However, incentives also just focus on commercially viable goods (Mitsch and Gosselink, 2000; Durigon et al., 2012), which means that the issues surrounding economic valuation continue.

A few studies have looked at societal values of ecosystem services (e.g. Ding and Nunes, 2014), but they have struggled as it is extremely complex and there is little data in support (Acharya, 2000). Human well-being is often mentioned as an important issue, which is supposed to be a major advantage of the concept (MEA, 2005), but often they just return to economic valuation (Acharya, 2000). Overall, currently the management of ecosystem services is dominated by the issue of economic valuation, but there are so many problems that we need to look for alternatives, but there is currently no consensus to the best way to focus ecosystem services management.

1.3.3 Mapping and modelling ecosystem services

Globally, the spatial estimation of ecosystem services is crude and will require great interdisciplinary effort to overcome (Naidoo et al., 2008; Bull et al., 2016). One outcome of the MEA (2005) is that more measuring, mapping and modelling

of ecosystem services is needed, which is also supported by other literature (e.g. Wolff et al., 2015). However, there is no consensus over which methods to use (Nahuelhual et al., 2015; Rasmussen et al., 2016). There is currently a lack of transparency and user-friendly methods to map and assess ecosystem services (Van der Biest et al., 2014) and also a lack of data (Holt et al., 2015), which causes problems for decision makers (Nahuelhual et al., 2015). However, if done well maps can act as powerful communication tools and provide information about restoring wetland functionality, with trade-offs and synergies (Jiang et al., 2013; Buckhard et al., 2013; Vorstitis and Spray, 2015).

Policy makers have tried to map the supply and demand of services in order to identify the impact of potential management decisions (Shroter et al., 2012; Wu et al., 2013), but as Maltby and Acreman (2011) have highlighted, there is very limited knowledge on how much of an ecosystem is needed to support each service. Ecosystem services are not equal or static, making spatial estimation complex (Syrbe and Walz, 2012; Schneiders et al., 2012), which most maps do not deal with (Peh et al., 2013). Maps can also represent the spatial differences between where there is supply and demand, and how people in different locations are effected (Bagstad et al., 2013a), but more detail at smaller spatial scales is required from research (Peh et al., 2013). Maps are also often used to produce different scenarios with alternative land uses (Brauman et al., 2007), which is useful for understanding potential future landscape dynamics, caused for example by policy or land-use change (Posthumus et al., 2010; Buckhard et al., 2013; Jiang et al., 2013).

Many studies use land cover attributes as proxies for ecosystem services, often using secondary data (Villa et al., 2009; Lautenbach et al., 2012; Nedkov and Burkhard, 2012; Di Sabatino et al., 2013; Crossman et al., 2013; Willeman et al., 2013). These proxies frequently come from a crude data source and the errors that can propagate through the analysis are often not considered (Eigenbrod et al., 2010). In Eigenbrod et al.'s study (2010) on biodiversity and carbon storage in England they found that proxy maps were only broadly similar to primary data maps, and so should only be used for broad scale patterns. Also, most ecosystem services are the result of complex interactions between many elements in the landscape that do not fit into specific land use categories (Van der Biest et al.,

2013). Therefore, each case may require a different approach, based on data availability and landscape composition (English Nature, 2006), and if land use is used it needs to be done so with caveats to the results.

Regulating services appear to have been mapped the most frequently and the individual services that are mapped the most are carbon storage, food production, recreation and water provision (Crossman et al., 2013), many of which are key in wetlands. Additional research is needed on mapping multiple services and cultural services (Crossman et al., 2013). Even though there is little academic work mapping cultural services (Hernandez-Morcilloa et al., 2013), there are many suggestions on how to measure recreation and tourism, including: number of visitors, accessibility, maintenance of green and blue space, trails and footpaths, the landscape naturalness, the number of overnight stays, values from stakeholder engagement and expenditure (Crossman et al., 2013; Larondelle and Haase, 2013; Nahuelhual et al., 2013; Martin-Lopez et al., 2014). However, of the MEA indicators (344 in total) only 38 cover cultural services, of which recreation and tourism account for 32 (Hernandez-Morcilloa et al., 2013). There is no methodological consistency with cultural services, which has hampered their integration into policy (Hernandez-Morcilloa et al., 2013). One reason given by Nahuelhual et al. (2013) is that the spatial definition of cultural services is often subjective and value laden, and so there is no single correct answer. Hernandez-Morcilloa et al. (2013) adds that there is currently no clear relationship between ecosystem functioning and cultural services.

Many approaches using modelling exist, but in most cases they require more development, as a better understanding of the links between landscape management and ecosystem service supply and demand is needed (Buckhard et al., 2013). Tools range from simple models using spreadsheets to complex suites of models, on a range of scales, from global to site specific, with many uses, such as scenario-building and economic valuation (Gret-Regamey et al., 2008; Bagstad et al., 2013b). Most have been developed to aid decision making. A variety of models have appeared in recent years (Bagstad et al., 2013a), from both private companies and research initiatives (WBCSD, 2013). Models include: InVEST (the integrated tool to value ecosystem services and their trade-offs) and ARIES (the artificial intelligence for ecosystem services) (Crossman et al., 2013).

However, as more tools appear there is increased confusion over which is appropriate to use (BSR, 2011). Kram et al. (2009) carried out an inventory and concluded that no one model can do everything, a combination is needed. In general, they found that many ecosystem services were covered, but that the analysis of cultural services was the weakest (Kram et al., 2009).

Many models deal with the important issue of scenarios, which are not predictions of the future, but feasible outcomes to explore (Carpenter et al., 2006; Ding and Nunes, 2014). Scenarios can help answer questions about how to deal with biodiversity and ecosystem services at a landscape scale. In general, an even balance of a few ecosystem services is desired, because often the loss of one service is not compensated by an increase in others (Richards et al., 2015). Ideally we want to look for synergies between more than one service in order to create service bundles (Wu et al., 2013), which some models do. An example comes from Swetnam et al. (2011), who created a range of scenarios from models, changing energy, economy, agriculture, forestry and population, using stakeholder engagement in Tanzania to see the effect on ecosystem service delivery.

Many issues exist with current ecosystem service models; such as scale, a reliance on large scale fieldwork, lack of a user-friendly interface, using land cover as a proxy, reliance on what studies there are in the literature, assumptions and often using data collected from other locations (Troy and Wilson, 2006; Villa et al., 2009; Lautenbach et al., 2012; Peh et al., 2013; Ding and Nunes, 2014; Van der Biest et al., 2014). Many models depict services as static and do not consider flows, which limits their application (Bagstad et al., 2013a). In some cases biodiversity is considered as an ecosystem service, but there is no consensus over which indicator to use for biodiversity, with over 40 existing (Overmars et al., 2014; Ding and Nunes, 2014). Many different outputs can come out of these models, with many different value domains, which makes comparison difficult.

Decision makers often find that the models take too much time, money or require skills they do not have (Bagstad et al., 2013b; Gardner et al., 2015), which should be important considerations when designing models. Fontana et al. (2013) states that a tool for decision making should combine ecology, economics and social

issues, in order to be more realistic, and so should be considered in any model selection. This paper also concluded that models need to be more linked to peer-reviewed ecological and biophysical models (Fontana et al., 2013). Another big issue to consider when using any model is the assumptions made and whether there is robust science behind them, which could partly be overcome by using peer-reviewed work. However, if used well these models and maps can act as powerful communication tools (Jiang et al., 2013; Buckhard et al., 2013), which can then be fed into decision making.

1.3.4 Synthesis/Research gaps

The concept of ecosystem services has been shown to have a useful role in conservation and landscape scale wetland restoration. The concept provides an important framework for managing the natural environment, especially in showing people our reliance on natural systems (Palmer et al., 2014; Bull et al., 2016). However, there are still many questions regarding ecosystem services that remain unanswered, such as what is the spatial distribution of ecosystem services (Wu, 2013) and how can we deliver multiple ecosystem services (Pan et al., 2013)? There is also a need for further research on valuing ecosystem services. Attempts to put monetary values on ecosystem services are popular, but appear perilous, as such calculations seem to be conducive to some services being side-lined and others solely thought of as part of a money driven process.

The concept of ecosystem services is applied in policy (Naidoo et al., 2008) and governments are more aware of the growing links with human well-being (Peh et al., 2013), but there is a demand for more research to address the incomplete scientific basis (Posner et al., 2016a; Bull et al., 2016). By increasing knowledge of ecosystem services it will help with decision making (Elmhagen et al., 2015a). It is impossible to conserve all ecosystem services across a landscape in restoring wetlands, and so we need to play to the strengths and needs of the particular environment. A framework needs to be created that addresses the biophysical, socio-cultural and economic values of ecosystem services, using dialogue between biophysical scientists, social scientists, academics and policy makers (Martin-Lopez et al., 2014). Simple and reliable indicators need to be

developed for decision making support so that risks can be minimised and public acceptance can be increased (Van der Biest et al., 2013). Therefore, it is a major challenge (Ding and Nunes, 2014) that needs to involve both scientists and policy makers (Gret-Regamey et al. 2008; Martin-Lopez et al., 2014), which will be addressed more in the following section on real-world management.

1.4 Real-world management

Both biodiversity and ecosystem services feed into the management of landscape scale wetland restoration, because acts of conservation occur anthropogenically and intrinsically (Faith, 2012). It is real-world management or governance that is key; what actually happens on the ground, the complex networks of real life, rather than what we aspire or imagine to happen. Thus, issues, such as funding opportunities and land availability, are included in decision making (Van Kersbergen and Van Waarden, 2004). The governance side of restoration and conservation is not as covered in the literature, but is where everything comes together to produce decisions and management that drives the wetland restoration process (Smith et al., 2007). Governance is the process of collectively solving society's problems and needs (Osbourne and Gaebler, 1992). Since the 1970s it is believed that there has been a shift from government to governance, where power is moving from hierarchical systems to inclusive networks of multiple stakeholders for restoration based decisions (Ruiz et al., 2011). There has also been some recognition that society and the process of governance itself are part of the problem (Voss and Bornemann, 2011).

1.4.1 Adaptive management

One type of environmental management becoming more common since the 1970s is adaptive management (Susskind et al., 2012; Westling et al., 2014), which has been widely advocated as a suitable approach for the protection of biodiversity and ecosystems (Westgate et al., 2013). Adaptive management aims to reduce the vulnerability of ecosystems, including wetlands, whilst understanding that they are full of uncertainty, involve socio-ecological links, and

that as more experience and knowledge is gained management actions need to be adapted (Jasonoff, 2010; Voss and Bornemann, 2011; Sharp et al., 2011; Westgate et al., 2013; Westling et al., 2014). It is a shift away from trying to control the environment, to having the ability to deal with change and uncertainty (Folke, 2006). Voss and Bornemann (2011) suggest that adaptive management can be a more “reflective” approach, which takes into account a range of perspectives, expectations and strategies, moving away from the assumption that the way we do things is always objective. By undertaking collaborative adaptive management, a range of stakeholders are involved, to use local knowledge and achieve more accepted outcomes (Susskind et al., 2012).

From these discussions and other literature, a series of criteria can be put together to understand adaptive management and whether a group on the ground are carrying it out. One of the criteria is the recognition that knowledge is incomplete and that different types of learning needs to occur, which then should be fed into continuously developing management actions (Voss and Bornemann, 2011; Rychlewski et al., 2014; Westling et al., 2014). This includes integrating research into the process (Westgate et al., 2013). Leading on, another criterion is that participation and integration of many stakeholders needs to occur, and in order to achieve collaboration, incentives, such as funding, are required (Voss and Bornemann, 2011; Susskind et al., 2012). These stakeholders need to be both social and scientific, although the approach has been criticised for not realising the potential tensions between different groups (Westling et al., 2014). These tensions can be overcome by being “reflective” (Westling et al., 2014), another criteria for good adaptive management. Finally, an important issue is to create goals to measure against before the process starts (Susskind et al., 2012; Westgate et al., 2013).

Folke (2006) believes that the best way to secure ecosystems for the future is to use the adaptive management concept, which links in well with landscape scale work. It also fits in with this idea of real-world governance where what we aspire or imagine to happen does not often occur on the ground, due to a range of different factors. However, there are very few examples of on the ground adaptive management (Westgate et al., 2013), which makes it difficult, but extremely useful to study, using the range of criteria identified previously.

1.4.2 Stakeholders and partnerships

As defined by Harrington et al. (2010), a stakeholder is a person with an interest in an ecosystem or who could be affected by it. The involvement of stakeholders and partnerships is a key part of adaptive management, where people and organisations share their diverse knowledge and experiences, and learn from each other. The role of stakeholders is increasing, partly because of the recognition of how useful local knowledge is (Palacio-Agundez et al., 2014). However, each stakeholder and organisation will have different ways of thinking, and how a problem is framed will go a long way to defining the solution (Jasonoff, 2003). Different issues of identity, definitions and priorities will play out in negotiations between stakeholders and organisations in a partnership during wetland restoration, with some complexities and controversies always occurring (Lems et al., 2011).

More open policy styles are needed that include the ever increasing knowledge within science and social systems in order to achieve more effective management, and therefore better wetland restoration (Arts and Buzier, 2009; Jasonoff, 2010). By working with a variety of organisations and stakeholders, Lems et al. (2011) believes that relationships can be formed and discussions more widely developed, achieving wetland restoration policy that is more open and accepted. Also, by using relevant stakeholders on the ground they are often in a much better position to carry out the work (Knight et al., 2007). However, there is the issue of trust to overcome; working relationships need to be built up (Tallis and Polasky, 2009). De Groot et al. (2002) believes that a framework for integration between the different aspects is still elusive, and so more research is needed to aid with this process.

One of the main groups of stakeholders that could be involved in landscape scale wetland restoration is farmers, alongside other local groups, in order to increase knowledge and approval (Maltby, 1991; Catchpole, 2006; Fletcher et al., 2011). If conservation is to move outside the traditional protected areas approach when the dominate land use is agriculture, then farmers need to be involved. However, agriculture can be very profitable, which is something that would need to be considered in any restoration plans. Without the farmers on side there would be problems; Wu and Hobbs (2002) have identified that the current land use

determines current dynamics and future development of a landscape. Land use activities can have far reaching effects in wetland environments (e.g. eutrophication (Maltby, 1991)), and so this might be one of the bigger hurdles to overcome in management of wetland restoration (Fisher et al., 2011). Rey Benayas and Bullock (2012) even go as far as stating that “agriculture and conservation are in permanent conflict”, and so this needs to be considered in any wetland restoration project in a farming landscape.

In recent years, the number of conservation projects that have been delivered by organisations working in partnership have increased, specifically for restoration work, but mostly on a case-specific basis (Hamblen and Canney, 2013). The idea of partnership working is to get many stakeholders from a wide range of organisations involved, in order to increase knowledge, engagement, buy-in and economic security (RSPB, 2001; Hume et al., 2008). Good communication helps engage people (Wu and Hobbs, 2002), as well as increasing trust by displaying acknowledgement of different views (Tallis and Polasky, 2009). A paper by Hodge and McNally (2000) highlighted the importance of using detailed local knowledge that can be a very rich source of information if coming from existing partnerships. Therefore, the concept of a partnership encourages adaptive management and may be a useful framework, alongside the concept of ecosystem services, for successful wetland restoration. Overall, there is the belief that a partnership means more joined up management, which in turn will facilitate more joined up conservation (and therefore wetland restoration).

1.4.3 Links to biodiversity and ecosystem services: Interdisciplinary decision making for wetland restoration

There are many links between landscape scale management, biodiversity and the concept of ecosystem services in restoring wetland functionality, as none of these concepts occur in isolation. Biodiversity has been involved in decision making for some time, and there has been a rise in the last decade of landscape scale perspectives (Jones et al., 2013), but the explicit use of the concept of ecosystem services in decision making is still in its infancy. However, it has often been accepted that the concept is beneficial, with many authors discussing the

big advantages to landscape scale management (Sitas et al., 2013; Albert et al., 2014a; Pagella and Sinclair, 2014). Also, it is useful as a communication tool to stakeholders, for looking at the synergies and trade-offs between economic, social and environmental interests and to act as a starting point for discussions amongst stakeholders (Albert et al., 2014a, b; Hatton MacDonald et al., 2014). However, there are a range of reasons that need to be further investigated as to why it is a difficult concept to use, the lack of research on practical tools and applications, the lack of good data and maps and there are problems with trying to understand how the environment and social issues are connected (Carpenter et al., 2006; Casado-Arzuaga et al., 2013; Hatton MacDonald et al., 2014; Fletcher et al., 2014; Pagella and Sinclair, 2014).

There are still issues with implementing the biodiversity concept into decision making. Decisions about biodiversity can be made by a wide range of people, each with a varying focus, including: the government, environmental organisations, planners and partnership stakeholders (Verissimo et al., 2014). Most decisions that impact biodiversity are actually primarily aimed at other issues, such as farming or infrastructure, but they often have unintended consequences for biodiversity. Only occasionally are decision makers beginning to think about these consequences in advance. Also, it has tended to be assumed that ecological processes are understood and can therefore be easily manipulated, and are isolated from other factors (Carpenter et al., 2006). Overall, a major problem with integrating biodiversity and ecosystem services into decision making, specifically in this case wetlands, is that there is barely any precedence for it and so decision makers do not know what to do as it is so novel (Rucklehaus et al., 2015). The cases where biodiversity is used for wetland restoration decisions, tend to be very site specific (Smith et al., 2008), and so they need to be understood in more general terms.

Decision making for wetland restoration and conservation in a real-world context does not just involve consideration of biodiversity and ecosystem services. There are a number of other factors that need to be understood and linked, such as economic, social, political, human and institutional issues (Knight et al., 2011a). This rarely occurs in reality (Albert et al., 2014b), but doing so could reduce the risk of conflict after decisions have been made and make plans more relevant for

real-world implementation (Nhancale and Smith, 2011; Metcalfe et al., 2015). Cotter et al. (2014) state that an interdisciplinary approach is needed to understand different demands and options, which in the face of potential future climate change is going to become more challenging (Struebig et al., 2015).

In the literature there are important discussions over what factors are involved in conservation decision making, including for wetland restoration. For many of those making decisions about the environment, Dicks et al. (2014b) suggests that their objective is to use the best available science to create “evidence-informed conservation”. However, the science is not widely used, creating a “science practice implementation gap” (Walsh et al., 2015). The reasons behind this include that research is not always relevant to decisions or that there is poor access to it (Dicks et al., 2014b; Walsh et al., 2015). Also, resources may be an issue; the decision makers have to think about what is actually achievable with the time and money they have (Pullin et al., 2013, Dicks et al., 2014b). This can also mean they are reluctant to spend this time and money learning, especially when there are many stakeholders (Pullin et al., 2013). As this is an important part of adaptive management it would suggest that many conservation decision makers do not carry out adaptive management.

However, Dicks et al. (2014a) state that science is only one of several drivers of decision making to consider. Successful decisions need the integration of a range of factors, including social and economic (Pullin et al., 2013; Dicks et al., 2014b; Cook et al., 2014). Also, any information that comes from scientists needs to take into account the other factors that might be involved (Sutherland and Freckleton, 2012). Often experiences, personal opinion and subjective values are used over science to inform decisions (Pullin et al., 2013; Walsh et al., 2015). However, these can be important, because alongside other factors, local knowledge and values are key to making more realistic decisions (Dicks et al., 2014b). Sutherland et al. (2012) state that scientists also need to learn from the experiences of those who are doing on the ground conservation, which links back to the idea of adaptive management. Decisions are often made with much uncertainty over whether they will work or not (Pullin et al., 2013), and so more work is needed to understand the factors in the wetland restoration decision making process better and stop this from occurring.

1.4.4 Synthesis/Research gaps

Finlayson (2012) states that this area of management is a paradox, as even though environmental policy is increasing, the levels of wetland degradation are also increasing, partly because the relationship between humans and nature is complex (Borie and Hulme, 2015). However, we are moving slowly towards solution based research, rather than just investigations (Wu, 2006). Final decisions on wetland restoration are currently often political and influenced by a range of non-scientific factors (Maltby, 2010), but everything needs to be integrated to be successful, with more open access science (Jasonoff, 2003). There does appear to be an increasing awareness of the need for action and management at the landscape scale, and the integration of the different factors, such as social issues, but there is still much uncertainty in how it should be delivered. This is partly because there is currently very little evidence on how successful different conservation interventions have been, with limited monitoring and evaluation (Bottrill and Pressey, 2012).

1.5 Outline of academic rationale

The discussion here has identified that there are strong links between biodiversity and ecosystem services (Loreau et al., 2001; Norris, 2011; Van Oudenhaven et al., 2012), but that even though there is a huge amount of literature, the research is still very much in its infancy with many questions remaining unanswered (Luck et al., 2009; Norris, 2011; UK National Ecosystem Assessment, 2011). Reyers et al. (2012a) have suggested that the links that exist between biodiversity and ecosystems services could have a lot of potential in terms of trying to face the sustainability challenges of the future, and therefore inform landscape scale wetland restoration. However, there also needs to be practical considerations taken alongside biodiversity and the concept of ecosystem services (Wilson and Law, 2016), such as land availability and funding (Van Kersbergen and Van Waarden, 2004). This feeds into the management side, in terms of how to make decisions that drive the restoration process. By involving different stakeholders and working in partnerships some of the issues around management will be overcome and bring together a range of skills and experiences to solve problems.

In reality it is probably even more complex than Figure 1.2 displays, especially at the landscape scale of working, as it is not just a case of up-scaling current processes. Whilst landscape scale restoration is acknowledged as desirable, in reality, the complete restoration of continuous large areas of wetlands is not feasible, due to other land uses, such as urban areas and valuable agriculture. Wetland restoration needs to be embedded within landscapes that serve multiple purposes. This requires careful decision making to preserve existing areas and expand wetlands within landscapes to seek maximum benefit, whilst considering the whole range of factors discussed in this chapter. De Groot et al. (2002) believe that a framework for integration between the different concepts and factors is still elusive, but is needed for successful restoration. This framework needs to consider the drivers and feedback within an interdisciplinary setting (Elmhagen et al., 2015a), and find a place for biodiversity and the concept of ecosystem services (Jax and Heink, 2015).

Clear and rigorous science with increased knowledge is also required (Maltby, 2010; Retford et al., 2012), which not only needs to consider the scientific quality, but also the needs of decision makers (Rose, 2014). Therefore, it is vital to develop knowledge and research for the effective restoration of wetlands at a landscape scale. This study will address this research gap of understanding the role that biodiversity, the concept of ecosystem services and the real-world factors have in wetland restoration and their potential contribution in understanding work at the landscape scale. The Humberhead Levels (HHL) provides a great opportunity for investigating this, a suitable case study, the context of which will be explored in the next chapter. It appears that such a study has not been previously carried out, but the importance of doing so has been shown.

Chapter Two: The context of the Humberhead Levels and the NIA Programme

2.1 Introduction

The HHL provides a great opportunity for exploring the issues and research gaps identified in the previous chapter. This chapter provides a summary of the HHL (including some of its major physical characteristics), the HHL Partnership and the Nature Improvement Area (NIA) Programme, before setting out the aims, objectives and structure for this thesis.

2.2 The Humberhead Levels

The HHL is located in North East England (Figure 2.1), and covers parts of Yorkshire, Nottinghamshire and Lincolnshire (HHL Partnership, 2011; Yorkshire Wildlife Trust, no date). It is defined by Natural England (2012a) as a National Character Area (NCA) of 171,805 hectares (Natural England, 2012a). NCAs are “areas that share similar landscape characteristics, and which follow natural lines in the landscape rather than administrative boundaries, making them a good decision making framework for the natural environment” (Natural England, 2012a). The HHL provides a good example of a wetland landscape that has been experiencing degradation for a long period, but has the potential to provide a whole range of biodiversity and ecosystem services through restoration (Natural England, 2012b; Yorkshire Wildlife Trust, no date). This landscape currently delivers a large number of ecosystem services, including food provision, climate regulation, soil erosion regulation, flood mitigation and recreation. It was formerly covered in wetlands, but has had a long history of human management, including several major drainage programmes. This means that the landscape in the past, and still now, has had other priorities than protecting the environment, and requires significant restoration in order to conserve and provide more biodiversity and ecosystem services.

The HHL is a flat, low lying area covering over 2000 square kilometres, made up of agricultural land (also a major employer), large drainage networks, urban areas,

heavy industry, several Sites of Special Scientific Interest (SSSIs) and other protected areas (Environment Agency, 2009; HHL Partnership, 2011; Natural England, 2012a; Yorkshire Wildlife Trust, no date). The landscape includes some highly fertile soils of the best agricultural grades, which in combination with the low, flat expanses of land means that the landscape is farmed intensively for arable (Natural England, 2012a). These fertile soils are caused by the geomorphological and fluvial history of the area, which includes some important glacial and alluvial deposits laid down by glacial Lake Humber across large parts of this landscape (Natural England, 2012a). The HHL is one of the most productive areas for agriculture in the UK (Natural England, 2012a).

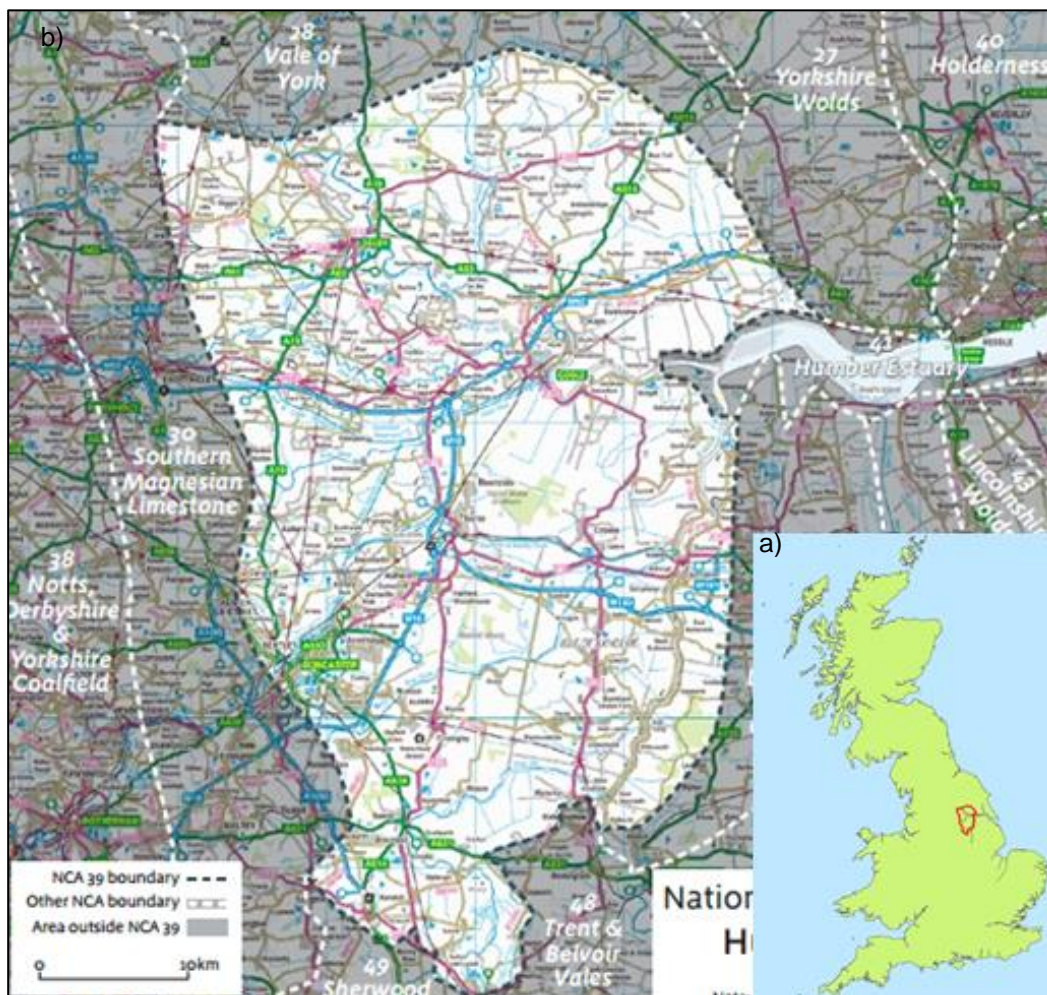


Figure 2.1: Location of the HHL NCA (a) within the UK (b) in the local regional (Natural England, 2012a).

The urban areas are fairly dispersed and connected by some major motorways (Natural England, 2012a). There are a range of protected areas across the landscape, including two National Nature Reserves (NNRs) – Thorne and Hatfield Moors (Natural England, 2012a). There are also several SSSI's, Local Nature Reserves (LNRs), Ramsar sites, Special Protection Areas (SPAs) and Special Conservation Areas (SCAs). Details of these designations can be found in Appendix A. These protected areas, and the surrounding landscape, provide habitats for over 200 species of birds (Natural England, 2012a), amongst other important groups, which demonstrates the biodiversity importance of the landscape. Some of these areas also protect the important raised peat bogs found mostly in central parts of the landscape, which are of historical interest, as well as ecological importance (Natural England, 2012a). Other historical features include the large strip field systems on the Isle of Axholme and the archaeological deposits found in certain soils (Natural England, 2012a). The long history of drainage, especially by the Dutch in the 17th century (Natural England, 2012a), has meant that the landscape has been heavily managed in the past. This has continued and formed the dykes, ditches and large areas of farming that can be seen across the landscape today (Natural England, 2012a).

Water management is a big issue in the area, partly due to flat low lying land and the numerous rivers that run across the HHL into the Humber Estuary (such as the Ouse, Don and Trent) (Natural England, 2012a). The network of water courses form part of the habitat network across the landscape (Natural England, 2012a). Most of the land is less than 10 m above sea level, with some areas even actually below sea level (Yorkshire Wildlife Trust, no date). Flooding has the potential to affect many people and businesses in the HHL landscape. Therefore, flood mitigation is a big issue, and could open up potential for wetland restoration that focuses on this particular ecosystem service.

There is a history of conservation work already in the landscape, including protected areas (as discussed earlier), the Wetland Vision project and Biodiversity Opportunity Areas (BOAs). The Wetland Vision project ran from 2008 until 2011 and was seen as highly successful (HHL Partnership, 2011). The project “created or restored 140 ha of wetland habitats”, encouraged more sustainable agriculture schemes (see Chapter Three, Section 3.5.3 for more

information), started to repair some of the damage from historical peat extraction and developed many people based projects (HHL Partnership, 2011). It also investigated the future potential of restoration in the landscape based on ecological and historical environmental characteristics (Hume et al., 2008), as can be seen in Figure 2.2. The BOAs came about after, as an outcome of a series of workshops as part of the Regional Biodiversity Strategy, to try to carry on the impetus of Wetland Vision by identifying areas of opportunity for restoration (HHL Partnership, 2011). They were defined under biodiversity principles by their landscape and biodiversity assets, with a robust mapping process that used local data (HHL Partnership, 2011), and can be seen in Figure 2.3. More detail could not be found on the selection process of the BOAs in the HHL, because there is no standard approach, but were probably based on using information on habitats, species and physical characteristics.

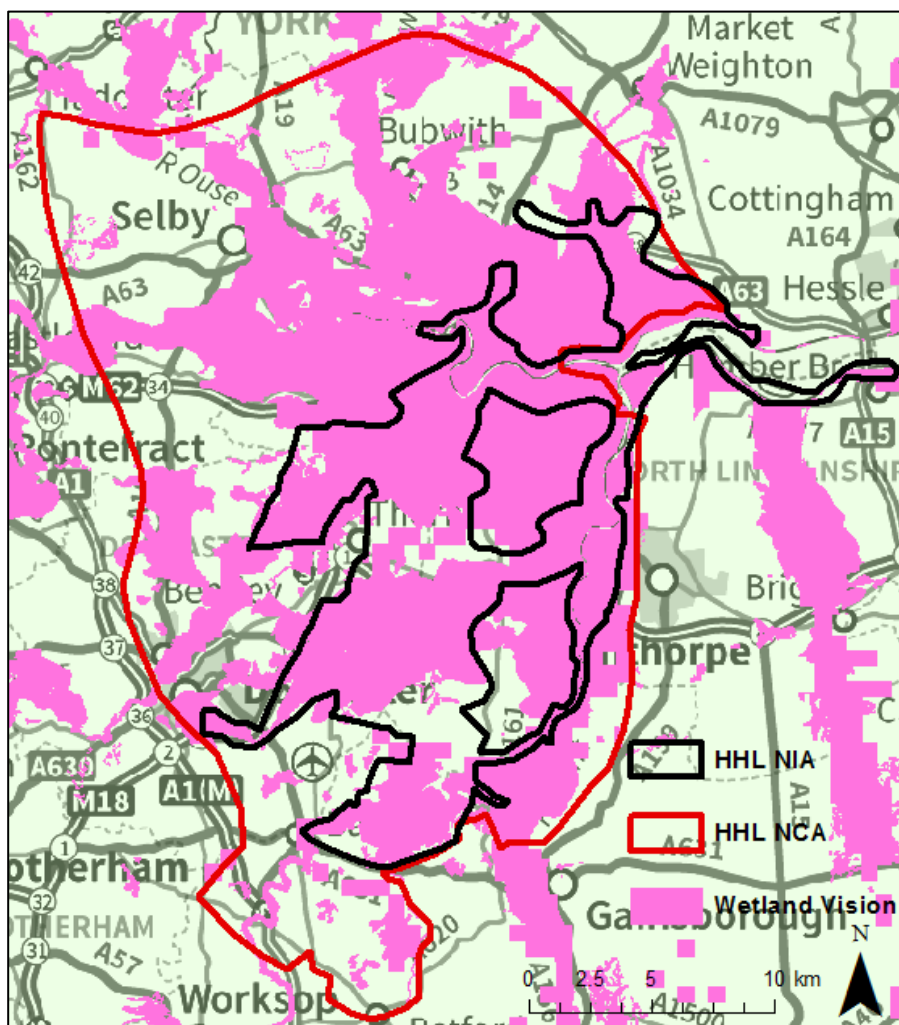


Figure 2.2: Wetland Vision project restoration potential for the HHL NCA, based on ecological and historical environmental characteristics (Hume et al., 2008).

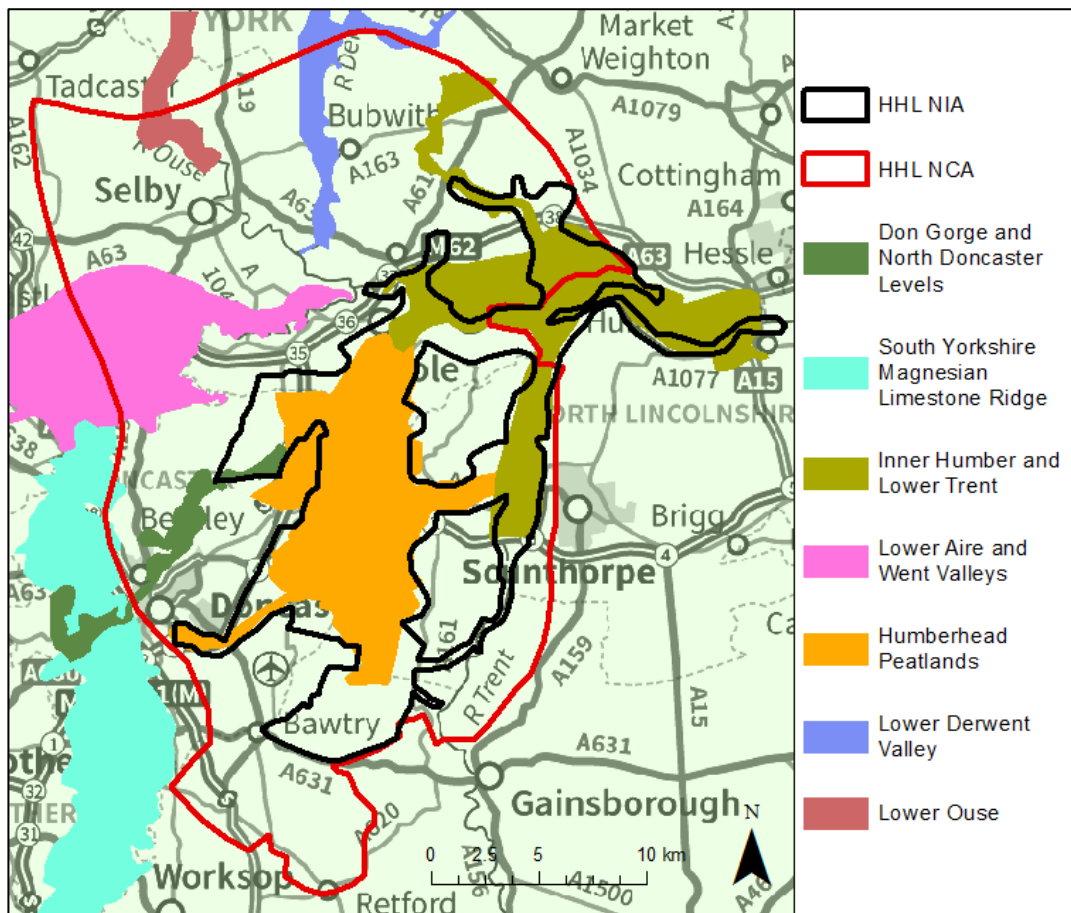


Figure 2.3: BOAs in the HHL NCA.

There are various pressures on the biodiversity and ecological functioning of the HHL, including the fragmentation of habitats, the increasing pressure for high intensity food production and the appropriate management of water resources (HHL Partnership, 2011). However, the landscape also offers a great opportunity to implement a programme of restoration applying principles of biodiversity conservation and ecosystem services, and building upon existing conservation work, to develop a multi-functional landscape, with wetland habitat interspersed with other land uses, such as agriculture (HHL Partnership, 2011). Such opportunities formed the basis for why the HHL was chosen as one of the projects funded under the Nature Improvement Areas (NIA) scheme undertaken by Natural England in order to restore large scale habitats (see Section 2.3).

2.2.1 Physical characteristics of the HHL

It is important to understand the physical characteristics of the HHL, as these affect patterns of biodiversity and ecosystem services (Nelson et al., 2009). In the

literature the importance of understanding the context of a landscape is discussed (Botequilha Leitao and Ahern, 2002; Jones et al., 2013) and management decisions are frequently made based on the spatial distribution of physical characteristics (Casado-Azuaga et al., 2013). Here four key features of the area are summarised: land use, soil type, nutrient levels and topography.

2.2.1.1 Land use

Land use is an important landscape characteristic as it defines patterns of human activities across an area, it provides the background to biodiversity and ecosystem services (Berger, 1987; Allan, 2004) and “determines the structure, function and dynamics of a landscape” (Wu and Hobbs, 2002). Figure 2.4 (page 63) shows the land use across the HHL area, providing detailed information on 23 types of land use. The landscape is dominated by the “arable and horticulture”. There are also some “improved grassland”, “broadleaved woodland” and “suburban” areas, and within the areas covered by the two NNRs there is mainly the “bog” land use.

2.2.1.2 Soil conditions

Many patterns of biodiversity and ecosystem services are influenced by soils (Berger, 1987) (and vice versa). For example, the type of soil would have an influence on the amount of carbon stored at a particular location. Figure 2.5 (page 64) presents the distribution of soil types across the HHL. There are many different soil types; the dominant one within the core of the landscape is “loamy and clayey soils of coastal flats with naturally high groundwater”. In wider parts of the landscape, soils are dominated by “slowly permeable seasonally wet acid loamy and clayey soils”. Again, as with the land use map, Thorne and Hatfield Moors (the two NNRs) stand out in the landscape, and are classed as “raised bog peat” soils. Using soils information means that we can understand more of what is going on under and on the surface of the HHL landscape, as soils influence water flow, plants and habitats, amongst a range of other things, an important consideration for wetland restoration.

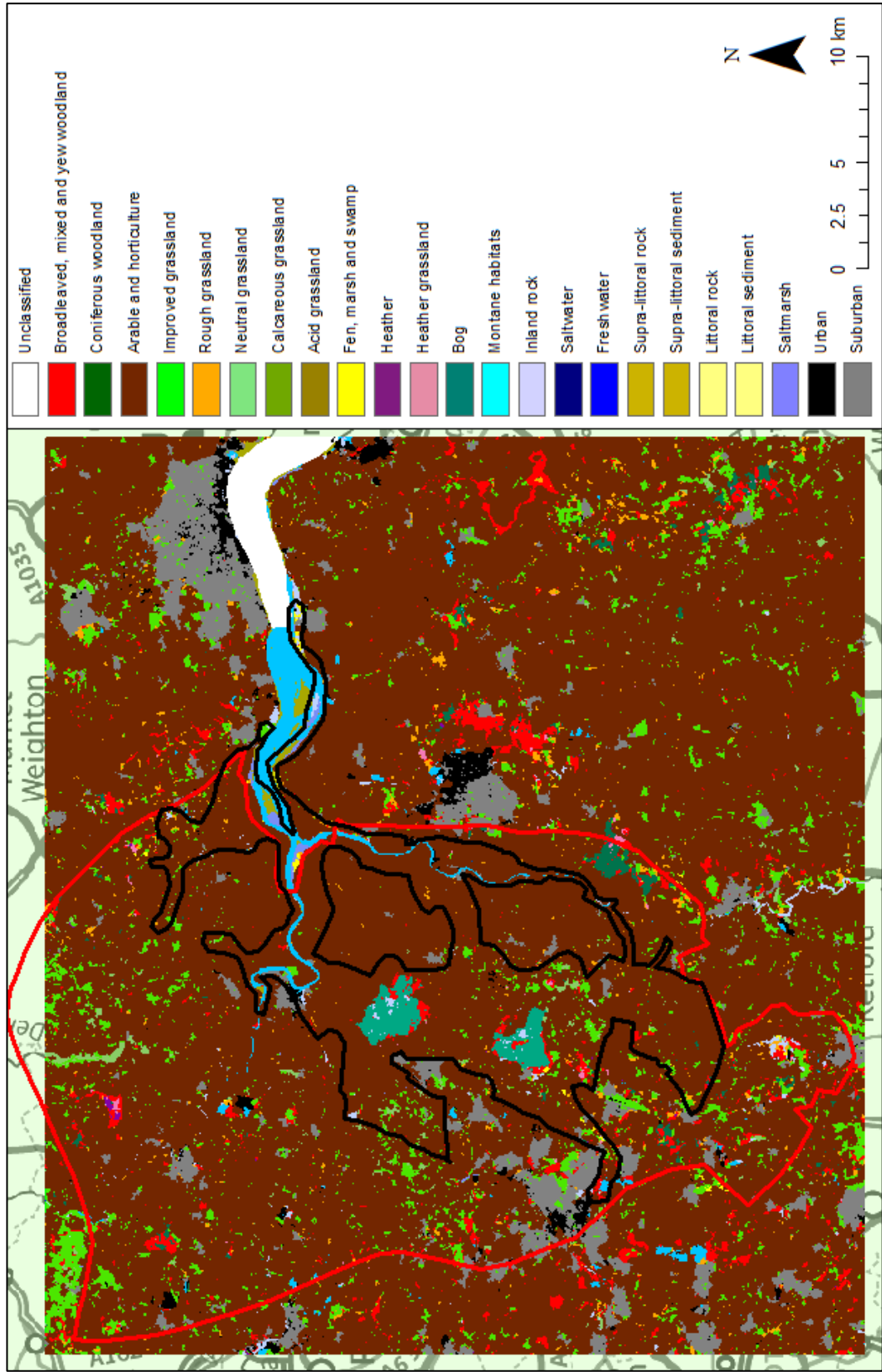


Figure 2.4: Land Use map from 2007 (the most up to date), acquired from Edina, but produced by the Centre for Ecology and Hydrology (CEH). The data is produced using satellite images and classification techniques (which may not always provide accurate information).

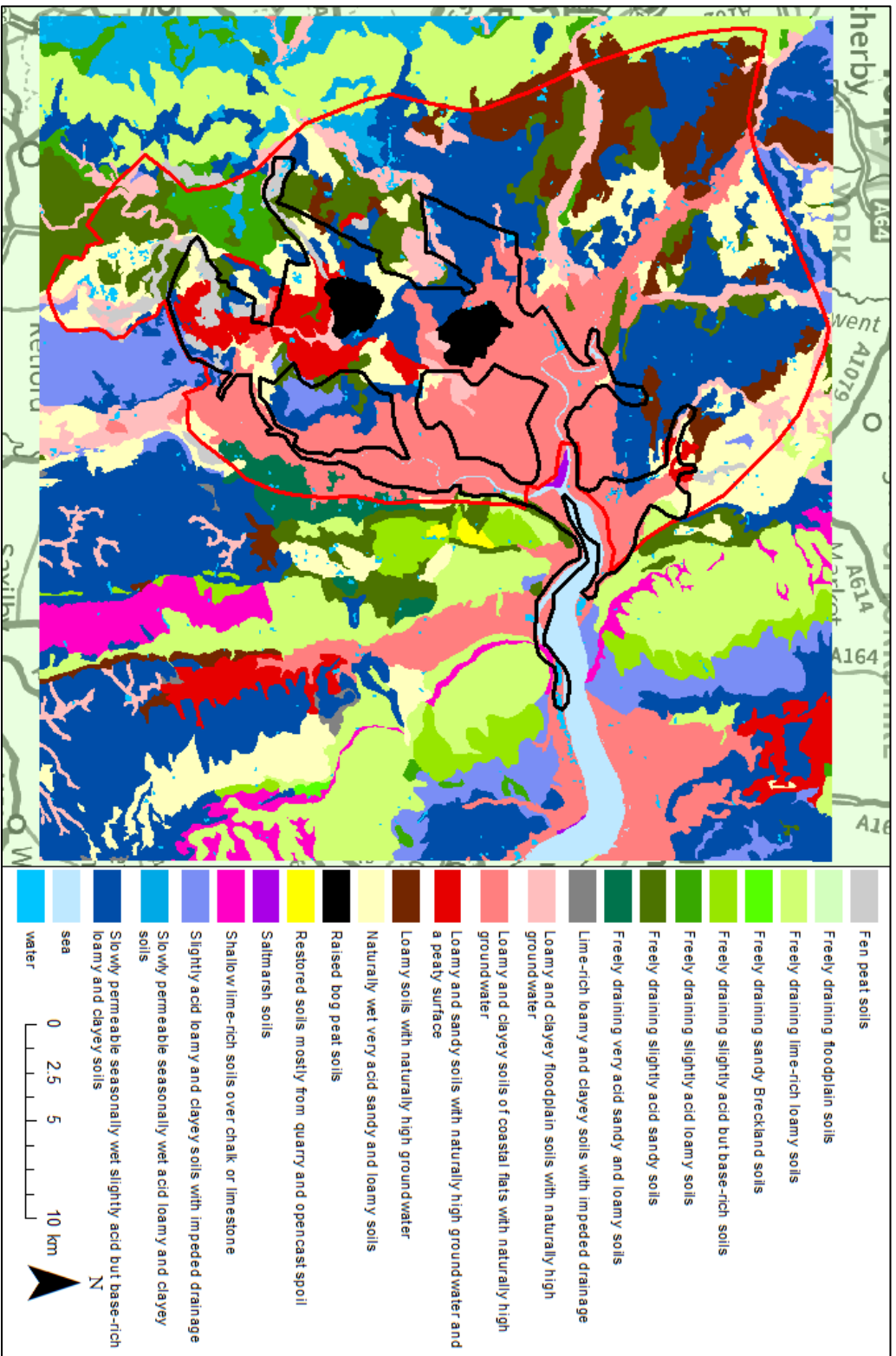


Figure 2.5: Soils map of the HHL NCA produced by Cranfield University.

2.2.1.3 Nutrient levels/Water quality

There are many ways proposed for studying nutrient levels within a landscape, including by analysis of water quality (Thiere et al., 2009; Syrbe and Walz, 2012; Kandziora et al., 2013). Here the levels of nutrients within rivers and streams were investigated to consider the water quality of the HHL landscape, which is vital for wetland environments. Data was sourced from the Environment Agency and comprised: ammonia, dissolved oxygen (DO), nitrates and phosphates. In the case of ammonia, the measurements were categorised into grades ranging from A (lowest pollution level) to E (highest pollution level). As seen in Figure 2.6 there is a lot of variation in ammonia across the HHL, but in most cases the results lie in Grades A to C. There are however some rivers that are Grades D and E, which could be an issue for any restoration plans.

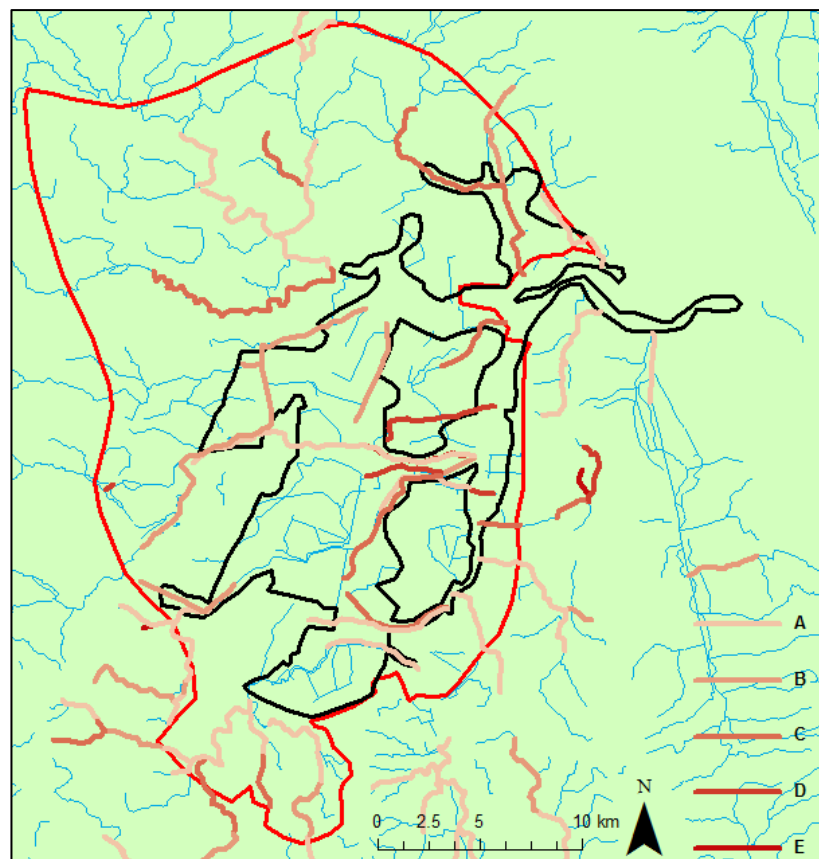


Figure 2.6: Ammonia levels of some rivers in the HHL NCA (sourced from the Environment Agency), ranging from A (lowest pollution level) to E (highest pollution level).

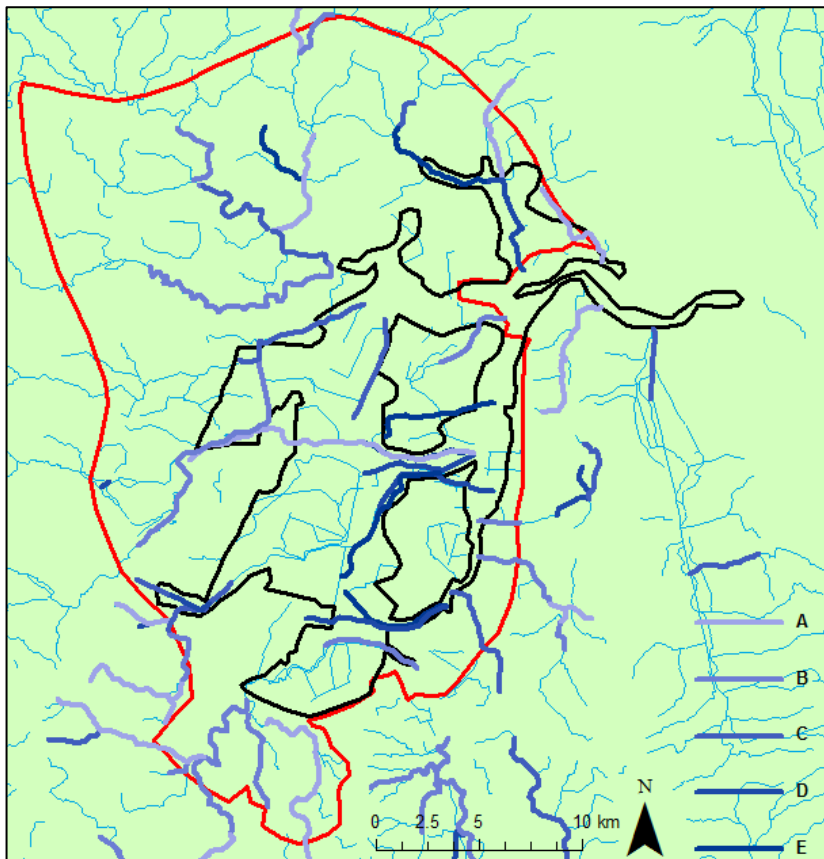


Figure 2.7: DO levels of some rivers in the HHL NCA (sourced from the Environment Agency), ranging from A (lowest pollution level) to E (highest pollution level).

Levels of DO are graded the same way as ammonia, from A to E, and the results for the HHL can be seen in Figure 2.7. Again, there is a lot of variation across the landscape, but there are many rivers classed within the higher grades. The results for nitrates and phosphates are graded differently, with the system running from Grade One (lowest) to Grade Six (highest). Figure 2.8 displays the results for nitrates and shows that most rivers are graded 4-6, with many at Grade Five. The results for phosphates are in Figure 2.9, with a lot of variation across the landscape, seeing results vary from Grade One to Grade Six. This information is useful as it allows a better understanding of water quality in the landscape, but is a complex factor to deal with, partly to do with the distribution of the available data from the Environment Agency, which does not cover the whole river network of the HHL.

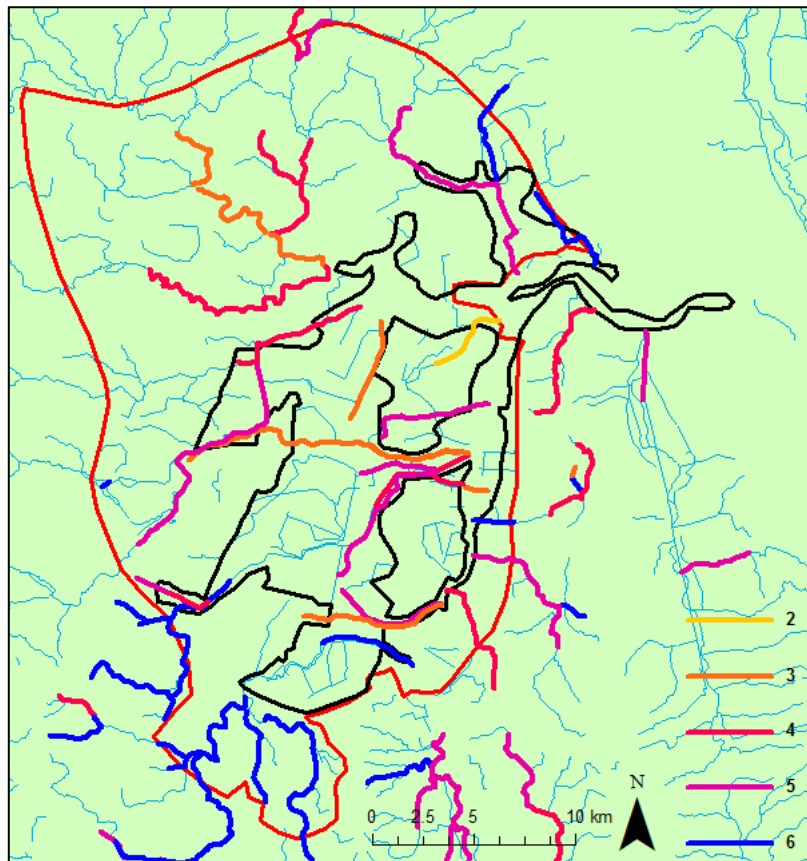


Figure 2.8: Nitrate levels of some rivers in the HHL NCA (sourced from the Environment Agency), ranging from Grade One (lowest pollution level) to Grade Six (highest pollution level).

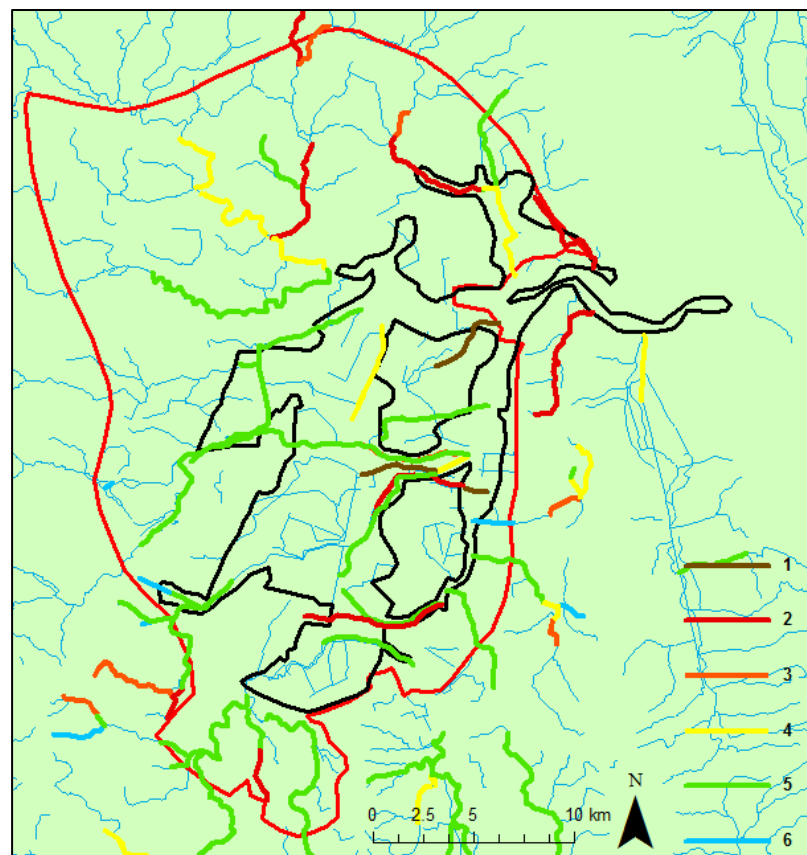


Figure 2.9: Phosphate levels of some rivers in the HHL NCA (sourced from the Environment Agency), ranging from Grade One (lowest pollution level) to Grade Six (highest pollution level).

2.2.1.4 Topography

The topography of a landscape will clearly be a major factor to consider for wetland restoration, as it will have a huge influence on water levels and soil depth, amongst a range of other factors (Cook and Hauer, 2007; Thiere et al., 2009). There are two sources of information on topography for the HHL landscape: a Digital Terrain Model (DTM) and contours (both sourced from Edina). The DTM for the landscape in Figure 2.10 shows that the area is very flat and low lying, meaning that water from the surrounding higher areas will flow into the landscape. The contours, also displayed in Figure 2.10, clearly show that the landscape is at a topographic extreme. These data sources of topography for the HHL allows further understanding of many aspects of the landscape, including height, slope and water flow direction, which will all be important for wetland restoration.

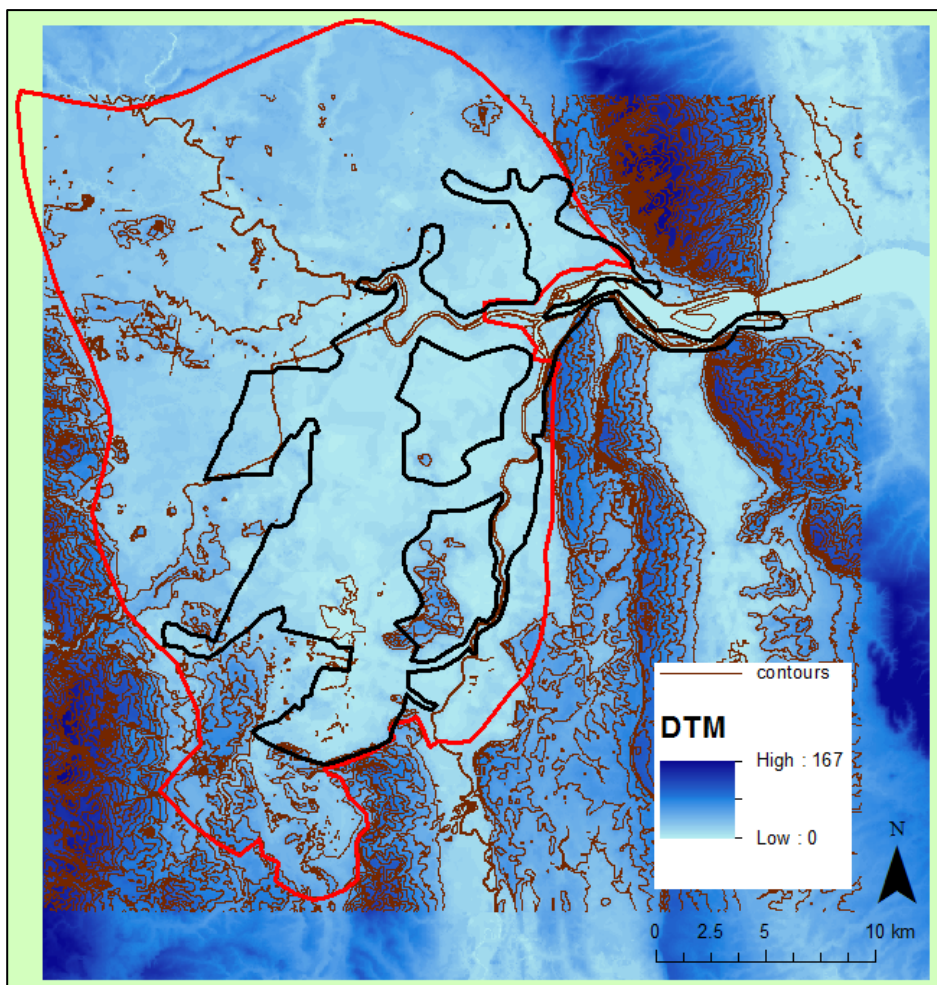


Figure 2.10: DTM and contours map for the HHL NCA (both sourced from Edina).

2.3 The Nature Improvement Area (NIA) Programme

In 2010 the independent review “Making Space for Nature” was published, written to investigate the ability of wildlife and environments in the UK to deal with climate change and other pressures (Lawton et al., 2010). This document stated that landscapes need to become “more, bigger, better and joined”, whilst thinking about the multiple benefits of the environment, through the concept of ecosystem services and using partnership working (Lawton et al., 2010). Partly in response to this review the White Paper “The Natural Choice: securing the value of nature” was published in 2011 by Defra, and established the NIA Programme. The main aim of NIAs was to “enhance and reconnect nature on a significant scale” in the long term, pursuing multiple benefits (Defra, 2011; Defra and Natural England, 2012). The NIA project created a need to refocus science and policy, and generated a new set of challenges, from which Defra (2011) aimed to capture the learning. There was a requirement to carry out a Monitoring and Evaluation Programme as part of each NIA, in order to allow evidence to be shared and help with the evaluation of the project at the end of each year and the overall end (Natural England, 2016). Some of the measures were compulsory and some optional, selected based on the specific context of each project, around the themes of biodiversity, ecosystem services, partnership working and socio-economic issues (Natural England, 2016).

2.3.1 The HHL Partnership

The HHL has had a partnership in place overseeing the management of the landscape since 2001, which has grown to consist of multiple organisations and stakeholders (Yorkshire Wildlife Trust, no date), and was used to manage the NIA Programme. They are considered to be one of the best examples of a working partnership in the environmental sector in the UK (Natural England, 2012b). Their goal is to use their shared visions, and different skills and experience to achieve landscape scale restoration “in a predominantly agricultural landscape, whilst supporting thriving communities and wildlife” (Yorkshire Wildlife Trust, no date). The main aim of their work is “to create an internationally renowned, unique wetland landscape, supporting thriving communities, ecosystem services, economy and wildlife” (HHL Partnership,

2011). Membership of the Partnership has varied since its inception, but the current active partners include (HHL Partnership, 2015):

- Natural England
- Defra
- RSPB
- Shire group of Internal Drainage Boards (IDBs)
- Yorkshire Wildlife Trust
- East Riding of Yorkshire council
- Nottinghamshire Wildlife Trust
- Lincolnshire Wildlife Trust
- Environment Agency
- The University of Sheffield
- The University of York
- English Heritage
- North Lincolnshire Council
- Doncaster Metropolitan Borough Council.

The Partnership is made up of a series of working groups, such as the Executive and Steering Groups, which each contain different partners and have different responsibilities towards the various funded projects the Partnership is involved with (HHL Partnership, 2011). The general workings and aims, and those of each group, are agreed under a Memorandum of Agreement (HHL Partnership, 2011). Following the success of the Wetland Vision project, the Partnership is considered to be a very effective partnership within the conservation community (HHL Partnership, 2011).

2.3.2 The HHL NIA

After an application process the HHL project was awarded £587,295 from Defra in 2012, which covers 49,700 hectares of the HHL NCA (Natural England, 2012b), as seen in Figure 2.11. They were one of 12 projects chosen, that all started on the 1st April 2012 and finished on the 31st March 2015 (Natural England, 2016). The successful projects were selected based on a range of factors, including: having aspects of ecological networks already managed in wildlife friendly ways,

opportunities to expand on that network, a good partnership already in place and obvious benefits to people (Natural England, 2016). As a result of the application process the HHL Partnership put together a business plan, which set out the HHL specific aims for the NIA Programme. The main aim was to deliver wetland creation and restoration to improve and extend the ecological network of the landscape, using the wide range of protected areas already in place (HHL Partnership, 2011), as seen in Figure 2.12. This includes the globally important wetlands of Thorne and Hatfield Moors and the Humber estuary (HHL Partnership, 2011). They also wanted to develop a volunteer programme, improve education and develop the green economy (HHL Partnership, 2011). The Partnership also identified a range of ecosystem services in their Business Plan, including carbon storage and recreation (HHL Partnership, 2011). The leading responsibility was given to the Yorkshire Wildlife Trust, with a full time Programme Manager appointed to oversee and manage the programme (HHL Partnership, 2011).

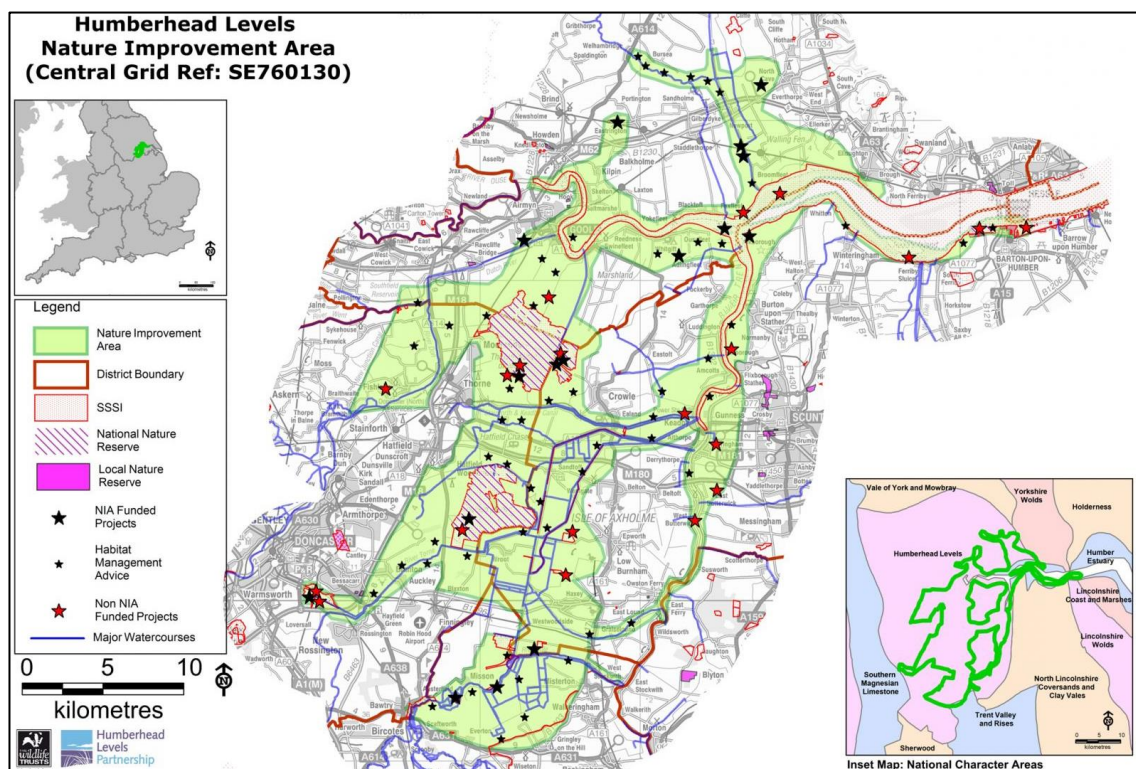


Figure 2.11: The NIA boundary, with the projects and protected site designations (YWT, 2012).

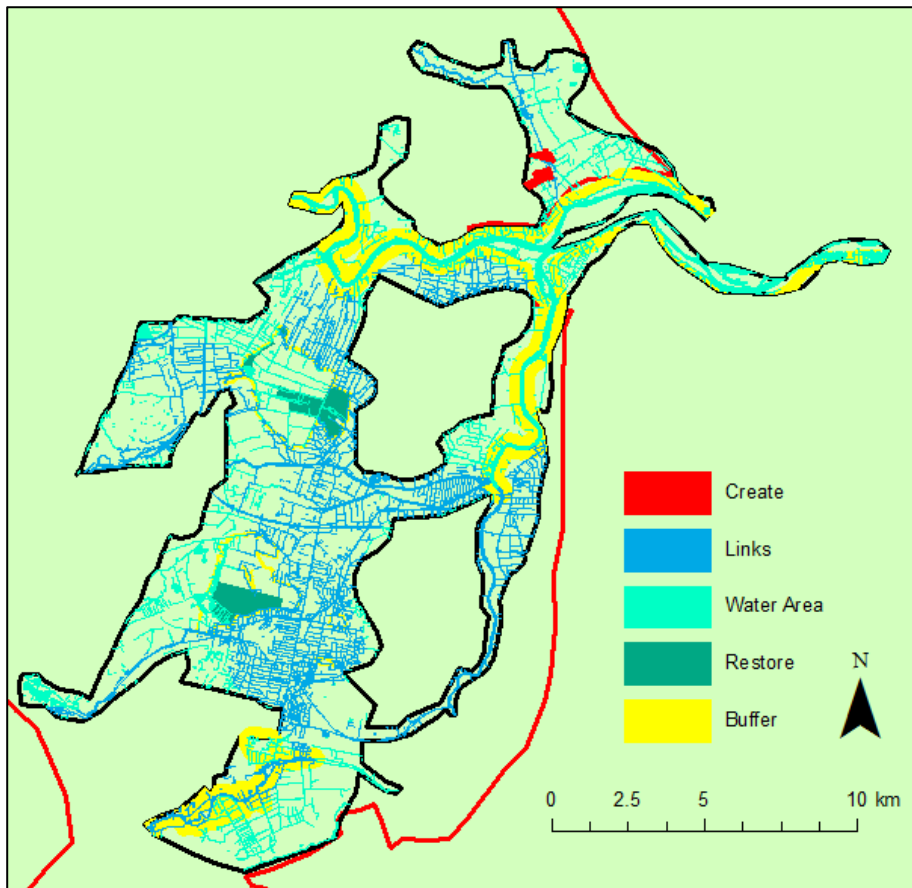


Figure 2.12: HHL NIA ecological restoration priority zones (information obtained from Yorkshire Wildlife Trust).

2.4 Aim and objectives

It was identified in the previous chapter that in order to achieve more successful wetland restoration biodiversity, the concept of ecosystem services and the real-world issues at the landscape scale all need to be further understood. In reality wetland restoration cannot cover a whole landscape, and so these three aspects need to work alongside other land uses. However, there are still many unknowns about how to deliver landscape scale wetland restoration on the ground. The HHL provides an ideal case study, because there are areas that could potentially be restored to wetlands, with many competing land uses. The existence of the NIA Programme and the effective HHL Partnership mean that restoration work is occurring. Therefore, there is an opportunity to study these issues on the ground, and it is important to learn from this (Sutherland et al., 2012). Due to these research gaps and the context of the HHL the following aim and objectives have been identified.

The aim of this project is to: *explore the potential and limitations of biodiversity, ecosystem services and “real-world” management in enhancing decision making for landscape scale wetland restoration.*

Maltby (1991) and Catchpole (2006) suggest that the future of wetlands depends on the gap between economics, society and ecology being bridged, and the aim of this project is to explore these in the context of the HHL NIA, with the improved use of environmental data. This involves exploring the current role of each factor, which will involve substantially increasing our understanding of each, before exploring the interactions and differences between them, and the opportunities and constraints this provides for restoring wetland functionality. This will create a truly interdisciplinary project, in order to increase knowledge and understanding. The literature reviewed in the previous chapter identified the need for more academic work on all of these aspects, in order to feed into how to deliver landscape scale restoration. The HHL NIA project provides an excellent case study for exploring the interaction of these factors, partly as they are recognised as one of the best examples of an environmental management partnership in the UK and the landscape is data rich (Natural England, 2012b).

The aim will be addressed with the following four objectives:

1. *To map the main types of ecological knowledge that could potentially inform wetland restoration, as well as the spatial structure of current wetland patches in the HHL area. This ecological knowledge will take the following forms:*
 - (a) *Major biodiversity patterns*
 - (b) *Indicators of ecosystem service provision.*
2. *To use the information from Objective One to address:*
 - (a) *What biodiversity and ecosystem services are delivered by the current wetland patches in the HHL landscape?*
 - (b) *Identify a range of potential locations for wetland restoration for the HHL.*
3. *To identify the actual decision making processes used in the NIA project, focusing especially on the factors used to choose the location of intended wetland restoration.*
4. *To compare the information gathered in the previous objectives to inform wetland restoration, and consider the potential and limitations for biodiversity, the concept of ecosystem services and landscape scale work in wetland restoration.*

2.4.1 Objectives

1. *To map the main types of ecological knowledge that could potentially inform wetland restoration, as well as the spatial structure of current wetland patches in the HHL area. This ecological knowledge will take the following forms:*

(a) Major biodiversity patterns

(b) Indicators of ecosystem service provision.

The optimal locations for wetland restoration to create the maximum gain for biodiversity and ecosystem services is unknown, but would provide useful information for many restoration schemes and as well as increasing knowledge within the HHL area. Information on biodiversity is useful for restoration because there are many benefits for humans (MEA, 2005), as well as those to species and the environment (Mace et al., 2012). However, there are still unknowns about the spatial distribution of biodiversity, especially in the face of the rapid rate of decline (MEA, 2005; Butchart et al., 2010). Ecosystem services is a useful concept to consider with wetland restoration because it is raising awareness of what the environment can do (Bull et al., 2016) and understanding the delivery of ecosystem services can be a framework for undertaking conservation (Goldman et al., 2008). There are also still many unknowns about this concept, including the spatial distribution of the services (Wu, 2013), and so by mapping this alongside biodiversity it will provide useful information to academic questions on these concepts, as well as for restoration in the HHL.

In order to carry out this objective, patterns of biodiversity and ecosystem services will be mapped for the HHL landscape, alongside the spatial distribution of the current wetland patches. A GIS map will be produced, with each layer displaying information on biodiversity or ecosystem services, in order to understand the spatial structure, form and context of the landscape. There have been many attempts to map various metrics and indicators (Holl et al., 2003; Bowne and Bowers, 2004), but involving many different types within one study is unusual. Selection of which aspects of biodiversity and ecosystem services to map will be complex because there is a huge range to choose from, both are very difficult to measure, there is no consensus in the literature over what is suitable and there are still many unknowns (Dinnin and Van de Noort, 1999; Holl et al., 2003; Rey Benayas et al., 2009; Paetzold et al., 2010). In order to choose the

appropriate metrics the literature and policy will be considered, alongside the available data and the context of the HHL landscape (Karjalainen et al., 2013).

2. To use the information from Objective One to address:

(a) What biodiversity and ecosystem services are delivered by the current wetland patches in the HHL landscape?

(b) Identify a range of potential locations for wetland restoration for the HHL.

The first part of this objective is to identify the level of agreement between the various different metrics mapped in Objective One and current wetland patches in the HHL landscape. It was identified in Chapter One that we do not fully understand biodiversity and ecosystem services, especially their spatial distribution across a landscape (Wu, 2013). Therefore, looking at what the current wetland patches in the landscape deliver in terms of biodiversity and ecosystem services is novel and very important. Also, it has been identified that in order to aid landscape restoration, clear and rigorous science is needed (Maltby, 2010; Retford et al., 2012), which is what will be attempted in this objective. Investigating the relationship between biodiversity, ecosystem services and areas already designated as wetlands is useful for a framework for understanding restoration potential. From this a range of potential future restoration options will be identified, based on data that could realistically be used to make decisions in a landscape and the results from part (a) of this objective. Important options that will be investigated include targeting biodiversity, targeting ecosystem services and then considering if it is possible to target restoration for both of these at the same time, for which current literature struggles to answer (Fisher et al., 2008; Anderson et al., 2009; Eastwood et al., 2016). Overall, these first two objectives are about understanding biodiversity and ecosystem services in landscape restoration and their ideal role in making decisions about the landscape.

3. To identify the actual decision making processes used in the NIA project, focusing especially on the factors used to choose the location of intended wetland restoration.

Whilst the optimal locations for wetland restoration can be predicted using physical and biological criteria, in reality, there are a number of additional considerations which impact where restoration actually occur and which will have

influenced the work of the HHL Partnership. This objective will further the understanding of the decision making process of where and how wetlands are restored, and the anticipated outcomes. This will be achieved by interviewing members of the HHL Partnership, and by examining relevant documentation, such as the HHL NIA Business Plan and the Making Space for Nature document, and attending partnership meetings. A particular focus will be the role that biodiversity and ecosystem services had in decision making process, as well as trying to understand the influence of the partnership. Therefore, the decision making process in the real-world will be explored, focusing on the roles of biodiversity and ecosystem systems in reality, as opposed to their ideal use, as with the previous two objectives.

4. To compare the information gathered in the previous objectives to inform wetland restoration, and consider the potential and limitations for biodiversity, the concept of ecosystem services and landscape scale work in wetland restoration.

This objective uses the results of the first three to examine what causes the differences and similarities between the two different types of information gathered to inform wetland restoration; the use of biodiversity and ecosystem services to construct restoration options and the real-world decision making processes for the HHL NIA. Maltby (2010) has identified that most decisions are often political or influenced by a range of other non-scientific factors, but Fisher et al. (2008) believes that science should be the main driver, and so these different viewpoints will be considered, as well as looking at a mix of both. Also, a balance has to be made between what is desirable and what is feasible in the real-world (Adams et al., 2004). The issues of ecosystem services will particularly be investigated, due to its growing popularity and ongoing levels of uncertainty. Once the similarities and differences have been identified, the implications of these for landscape scale restoration will need to be considered. De Groot et al. (2002) believe that a framework for integration between the different discourses is still elusive, but this is needed for successful restoration. Therefore, it is vital to develop the knowledge and research for the effective restoration of wetlands at a landscape scale by integrating all the various aspects in and to attempt to find the right perspective for it all.

2.5 Structure and approach

This section deals with the structure of the thesis; which chapters the objectives are found in. In Chapter Three, the first objective will be addressed, where various metrics of biodiversity and ecosystem services will be mapped across the HHL. Chapter Four covers the first part of Objective Two, which will use the information from Objective One to understand what the current wetland patches are delivering in terms of biodiversity and ecosystem services. Following on, Chapter Five will use the information from the previous chapters to look at a range of potential restoration options for the HHL, which is the second part of Objective Two. These first two objectives together look at how the concepts of biodiversity and ecosystem services could be used in an ideal way to inform landscape scale wetland restoration. Chapter Six covers the third objective, looking at restoration decisions in the real-world and all the different factors that are involved, including the role that biodiversity and ecosystem services have. The final chapter, Chapter Seven, through Objective Four, will investigate the similarities and differences between the two different approaches looked at in this work, as well as concluding the thesis. Due to the nature of this work there is no specific methods chapter, but each will have its own methods section, explaining the specific method developed and used throughout that chapter.

Here an outline of the methods to be used in Chapters Three to Seven will be provided, to better orientate the reader through the overall project. However, most of the details on the methods will be included in the individual chapters. An important issue to consider throughout this thesis, especially in terms of the methods, is the intentional pragmatism placed on choosing to use data, models and methods that reflect the potential engagement with these by practitioners, specifically looking at ease of access and ease of use. This work is supposed to consider the idea of “real-world” conservation and so it makes sense that it should use methods that could potentially be replicated for many other similar projects.

The first strand of the thesis involves mapping patterns of biodiversity and ecosystem services to construct restoration options for the HHL. Metrics of biodiversity and ecosystem services are chosen for the landscape in Chapter Three, and then mapped using GIS and modelling techniques. In Chapter Four these metrics are further analysed compared to current wetland patches in the

HHL using basic statistical analysis. This strand is concluded in Chapter Five, where restoration options for the HHL are developed using information from the previous two chapters, and further basic statistical and GIS techniques.

The second strand of the thesis involves using three different methods to collect information to understand how “real-world” decisions are made, and is carried out in Chapter Six. Firstly, there is the attendance of meetings and workshops using participant observation, and the collection of agendas and other documentation. This documentation and other important texts, such as the HHL Business Plan, will then be analysed. Finally, interviews with members of the HHL Partnership will be carried out. In Chapter Seven the two strands of the thesis are brought together and compared in order to understand the disparity between the ideal way of planning restoration and what happens in the “real-world”. This is carried out again using basic GIS techniques along with a visual assessment of the differences.

Chapter Three: Mapping biodiversity and ecosystem services in the HHL

3.1 Introduction

The purpose of this chapter is to address the first objective, which is as follows;

To map the main types of ecological knowledge that could potentially inform wetland restoration, as well as the spatial structure of current wetland patches in the HHL area. This ecological knowledge will take the following forms:

(a) Major biodiversity patterns

(b) Indicators of ecosystem service provision.

The optimal locations for wetland restoration in order to create maximum gain for biodiversity and ecosystem services are unknown, but would provide useful information for many restoration schemes, as well as specifically increasing knowledge within the HHL. The aim here is to create a GIS map, with each layer displaying a metric or indicator of biodiversity or ecosystem services, in order to understand the spatial structure, form and context of the landscape. There have been many attempts to map various metrics and indicators (Holl et al., 2003; Bowne and Bowers, 2004), but an investigation of different forms within one study is unusual. Also, very few studies use scientific methods to predict or test restoration (Holl et al., 2003), and so by mapping the two different types of scientific knowledge the ideal locations for restoration can be identified. GIS has enhanced the investigation of conservation strategies and outcomes, by providing the capacity to accumulate spatial data and run spatial analysis (Nielson et al., 2005), allowing a study of this type to occur, integrating many sources of information on one landscape.

The first step is to map the current and previous wetland patch structure across the HHL landscape, before deciding on which metrics and indicators to use for biodiversity and ecosystem services. There is considerable debate in the literature over which metrics and indicators are suitable (Paetzold et al. 2010), but no consensus has been reached, creating a gap between theory and application (Holl et al., 2003). However, the most useful are identified here by considering the literature, policy documents, data availability, context and using

the NIA Monitoring and Evaluation regulations in order to create parallels with the HHL work. This landscape is data rich and provides a great opportunity for research. Once the various metrics and indicators are chosen and mapped there will also be brief discussion over their worth in decision making, before being fed into further analysis in Chapter Four and Chapter Five.

3.1.1 Indicators of biodiversity

It is important to thoroughly consider the correct metrics to use to understand biodiversity. Dinnin and Van de Noort (1999) state that biodiversity is very difficult to measure, partly due to the arguments of whether the number of species or the abundance of particular species is more appropriate, and what is the most suitable for landscape scale work. Other issues include how will different indicators relate to the different stakeholders and what data already exists in the particular landscape (Normander et al., 2012). The EEA (2007) propose 26 indicators of biodiversity in line with combating biodiversity loss by the original 2010 deadline, including the fragmentation of habitats, nitrogen levels and public awareness, using a very detailed set of criteria to select each. The Biodiversity 2020 Strategy (Defra, 2012) also provides useful recommendations for indicators to use. As part of the NIA programme, the HHL Partnership also had a series of Monitoring and Evaluation metrics to measure and report back on (Defra and Natural England, 2012).

The EEA (2007) states that different combinations of indicators will answer different questions, which is why it is also important to think about the end use before making a selection (Feld et al., 2009). Henik and Kowarik (2010) suggest that it is better to choose a few indicators that represent more and are sensitive to change, rather than use many. However, no one perfect indicator currently exists. Popular techniques include measuring species composition, richness, abundance and diversity (Feld et al., 2009; Schneiders et al., 2012). Displaying biological data as maps is very useful visually on a landscape scale, such as was used in the Wetland Vision Project (Hume et al., 2008). Overall, Feld et al. (2009) has suggested that techniques need to be “rigorous, repeatable, widely accepted and easily understood”, and therefore a decision should be made only after the

area and current data had been studied. Also, the aforementioned literature and policy documents provide a useful basis to start from.

3.1.2 Mapping ecosystem services

Globally, the spatial estimation of ecosystem services is crude and requires a great interdisciplinary effort to overcome (Naidoo et al., 2008). There is currently a lack of transparency and user-friendly methods to map and assess ecosystem services (Van der Biest et al., 2014). However, if done well they can act as powerful communication tools for decision making (Jiang et al., 2013; Buckhard et al., 2013). Policy makers have tried to map ecosystem services in order to identify the impacts of potential management decisions (Shroter et al., 2012; Wu et al., 2013), but as Maltby and Acreman (2011) have highlighted there is very limited knowledge on how much of an ecosystem is required to support each service. Maps can also represent the spatial differences between supply and demand, and how people in different locations are affected (Bagstad et al., 2013a), but in general more detail at a smaller spatial scale is required (Peh et al., 2013). Also, many current methods do not deal with the variation in services through time (Peh et al., 2013). Mapping is a good method for understanding potential future landscape dynamics, for example caused by policy or land-use change (Posthumus et al., 2010; Buckhard et al., 2013; Jiang et al., 2013). It has also been used to evaluate the potential response of multiple ecosystem services to different management options (Jones et al., 2013).

Many studies use land cover attributes or other proxies for ecosystem services (Di Sabatino et al., 2013), often collected as secondary data (Crossman et al., 2013). These proxies frequently come from crude data and the errors that can propagate through analysis are often not considered (Eigenbrod et al., 2010). In Eigenbrod et al.'s (2010) study of biodiversity (using BAP observation records) and carbon storage in England, they found that proxy maps were only broadly similar to primary data maps, and so should only be used for broad scale patterns. Also, most ecosystem services are the result of complex interactions between many elements in the landscape, that do not fit into specific land use categories (Van der Biest et al., 2014). Therefore, each case may require a different

approach, based on data availability and landscape composition (English Nature, 2006).

Regulating services appear to be the most frequently mapped, and the individual services that are most commonly mapped are carbon storage, food production, recreation and water provision (Crossman et al., 2013). Much more work is needed on mapping cultural services and multiple services in a landscape (Crossman et al., 2013). Carbon storage is a big issue often studied in the literature, as a change to intensive agriculture tends to mean less carbon can be stored (Jiang et al., 2013). The most common approach for looking at carbon storage is to use land cover maps, especially the soils and vegetation, and to estimate the storage for the different biomes, for both above and below ground (Naidoo et al., 2008; Egoh et al., 2008). Land classes are also used as a proxy for food production and water storage capacity (Nedkov and Burkhard, 2012; Willeman et al., 2013).

Even though little work has been done on cultural services (Hernandez-Morcilloa et al., 2013), there are many suggestions on how to measure recreation and tourism, including: number of visitors, accessibility, maintenance of green and blue space, footpaths, landscape naturalness, number of overnight stays and expenditure (Crossman et al., 2013; Larondelle and Haase, 2013; Nahuelhual et al., 2013; Martin-Lopez et al., 2014). There is no methodological consistency with cultural services, which has hampered their integration into policy (Hernandez-Morcilloa et al., 2013). One reason suggested by Nahuelhual et al. (2013), is that the spatial definition of cultural services is often subjective and value laden, and so there is often no absolute answer. Hernandez-Morcilloa et al. (2013) add that there is no clear relationship between ecosystem function and components, and the cultural services they provide.

Many modelling approaches exist with ecosystem services, but in most cases they require more development, as a better understanding of the links between landscape management and ecosystem service supply and demand is needed (Buckhard et al., 2013). The choice of methods is vital, as different forms of analysis could create different trade-offs and synergies in a landscape (Martin-Lopez et al., 2014), but the majority of methods are still in the development and

testing stages (Nedkov and Burkhard, 2012). As with biodiversity, the HHL Partnership have a series of Monitoring and Evaluation regulations that they had to adhere to under the NIA funding, and so again these will provide some good indicators to use (Defra and Natural England, 2012). The HHL Business Plan (HHL Partnership, 2011) will also provide good suggestions for ecosystem services to focus on, as it is not feasible to do all.

3.1.3 Summary

In order to address the objective of this chapter a series of metrics for biodiversity and ecosystem services are identified and mapped. This will allow the spatial distribution of current wetland patches in the HHL to be identified, as well as the spatial distribution of biodiversity and ecosystem services, so that in following chapters these can be further investigated and utilised. As well as biodiversity and ecosystem services being useful for decision making in the landscape, the physical characteristics mapped in the previous chapter will also be valuable in further chapters. The structure of this chapter will be as follows: methods, HHL current wetland patches, biodiversity mapping, ecosystem services mapping and discussion and conclusion.

3.2 Methods

The methods involve using mapping software (ArcGIS) and modelling software to understand the spatial distribution of current wetland patches, and patterns of biodiversity and ecosystem services across the HHL. The analysis throughout this work covers not only the NIA area, but the whole of the HHL NCA. Each section involves slightly different methods. For each, the metrics are identified, using literature and data availability. These are then mapped and explained, including how they were produced, before briefly discussing their implications for decision making and their application in this study.

3.3 HHL wetland patches

It is important to identify the current wetland patches in the HHL, as it provides an understanding of the current condition of the landscape, which can be then used in conjunction with the following metrics to understand the HHL further. Figure 3.1 shows the distribution of current wetland patches in the HHL with data obtained from Natural England. This map shows that the vast majority of current wetland systems are found in the two NNRs, Thorne and Hatfield Moors, and along some of the river banks, with other patches found to the west of the NIA area. The area covered by the NIA Programme does enclose many of the wetland patches that are within the HHL landscape.

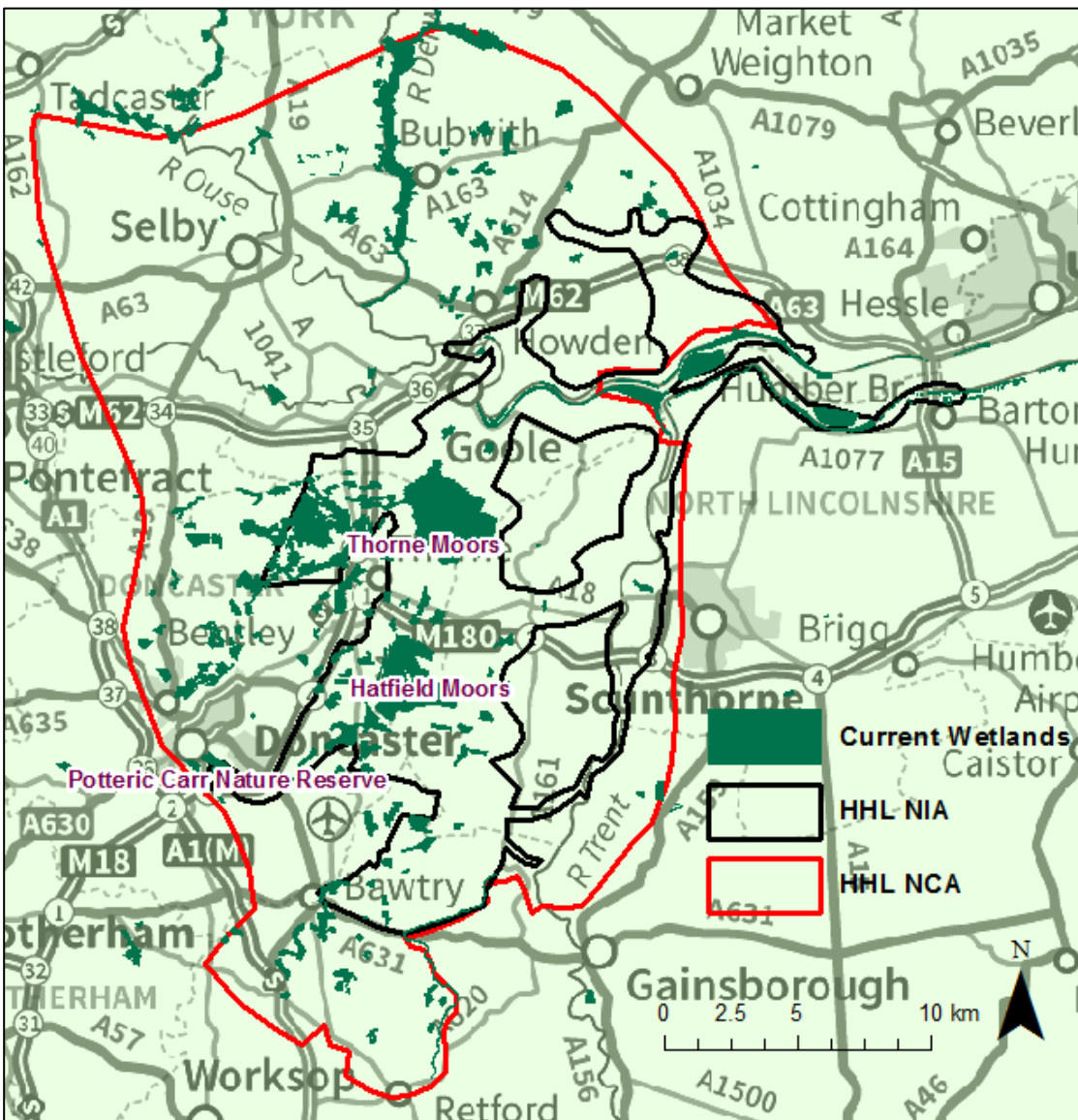


Figure 3.1 Current wetland patches (data acquired from Natural England) in the HHL landscape, with some of the key sites labelled.

In comparison Figure 3.2 shows the historic wetland patches across the HHL landscape, also sourced from Natural England. This was defined in the Wetland Vision Technical Document as “not a definitive record of previous extent, but is based instead on underlying soil characteristics, and shows the maximum former extent of wetlands” (Hume, 2008). As discussed in Chapter Two, the landscape was previously dominated by wetland patches, which is visible in Figure 3.2, and covered large parts of the NIA area. Understandably, many of the patches extend outwards from river corridors and also cover areas beyond the two NNRs. Together, these maps will be used in Chapters Four and Five to develop options for wetland restoration. Both figures, especially the current wetlands, will be a useful indicator to identify where else to restore wetlands.

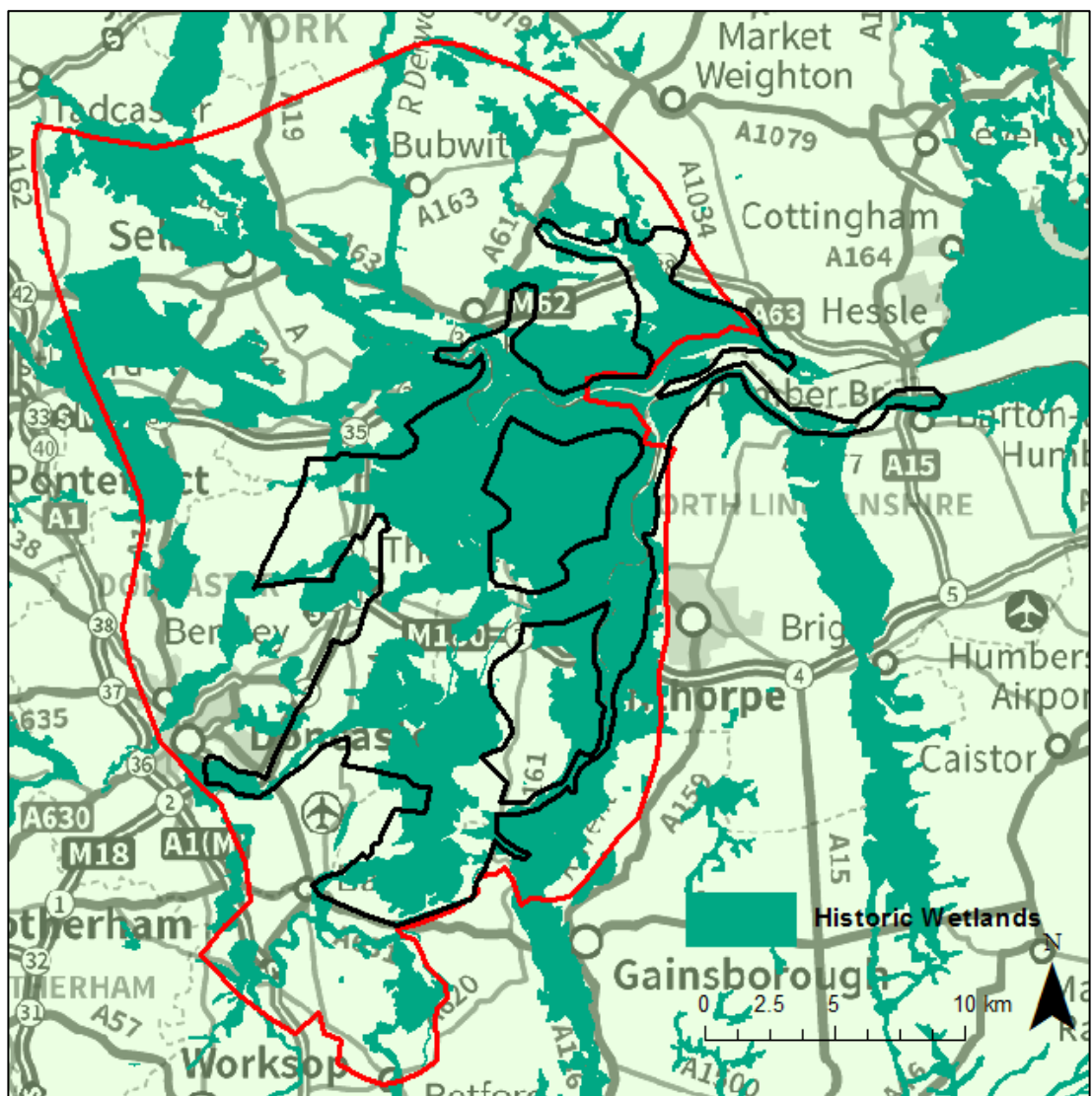


Figure 3.2 Historic wetland patches in the HHL landscape (data acquired from Natural England).

3.4 Biodiversity mapping

Biodiversity is important to landscape restoration for many reasons, including that it creates a buffer against change if there are more functional traits available through an increased range of species (Mace et al., 2012). However, Dinnin and Van de Noort (1999), amongst others, state that biodiversity is difficult to measure, and so choosing and mapping metrics is complicated. Several indicators have been identified from the NIA Monitoring and Evaluation work (Defra and Natural England, 2012), the Biodiversity 2020 Strategy (Defra, 2012), the 2010 Biodiversity target (EEA, 2007) and other literature.

3.4.1 Nationally Designated Protected Areas

The Nationally Designated Protected Areas found within the HHL are SSSIs, Ramsar, NNRs, LNRs, SPAs and SACs.

- SSSIs are designated by Natural England and are protected under the Wildlife and Countryside Act 1981, in order to safeguard the UK's best wildlife and geological sites (Natural England, 2013).
- Ramsar sites are wetlands of international importance and are designated under the Ramsar Convention, which came into force in 1975 (The Ramsar Convention Secretariat, 2014).
- NNRs were put into place to protect wildlife habitats for recreation and scientific research, and are protected under the National Parks and Access to the Countryside Act 1949, and the Wildlife and Countryside Act 1981 (Defra, 2013).
- LNRs are locally important sites of special wildlife or geological interest, and were designated under the National Parks and Access to the Countryside Act 1949 (Defra, 2013).
- SPAs were classified for rare and vulnerable birds and regularly occurring migrating species, under the EC Birds Directive in 1979 (JNCC, 2013a).
- SACs are high quality conservation sites designated under the EC Habitats Directive (JNCC, 2013b).

Nationally Designated Protected Areas were selected as many are designated due to high levels of biodiversity (see Appendix A), and was suggested by a range of literature and policy documents (e.g. Adams et al., 1994; Biodiversity 2020 Strategy (Defra, 2012); Maes et al., 2013). Therefore, these could be useful sites as core areas with the ecological networks concept. Many of them cover very similar areas, often occurring within the NIA. The SSSIs (Figure 3.3) cover Thorne and Hatfield Moors, large parts of the Humber estuary, and many other patches, mostly along river banks. Ramsar designations (Figure 3.4) cover almost exactly the same areas in the Humber estuary as the SSSIs with additional river patches. The NNR designations (Figure 3.5) cover Thorne and Hatfield Moors, as well as a few smaller patches within the NCA. LNRs (Figure 3.6) are much smaller patches with only one area within the NIA. The SPAs (Figure 3.7) cover much of the same area as the SSSIs, but less of Hatfield Moor. Finally, the SACs (Figure 3.8) again cover much of the same area to the SSSIs, but without many of the smaller patches. In terms of decision making this information allows awareness of what areas are (or should be) in good condition that could then be built on and are fairly straightforward to assess.

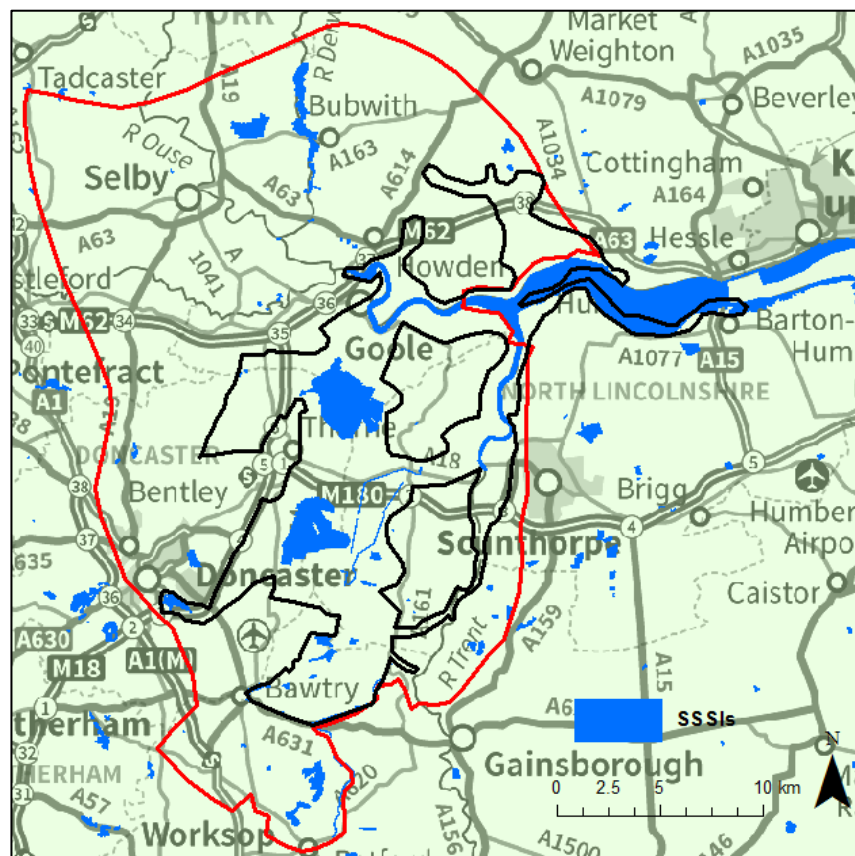


Figure 3.3: SSSIs in the HHL (data acquired from Natural England).

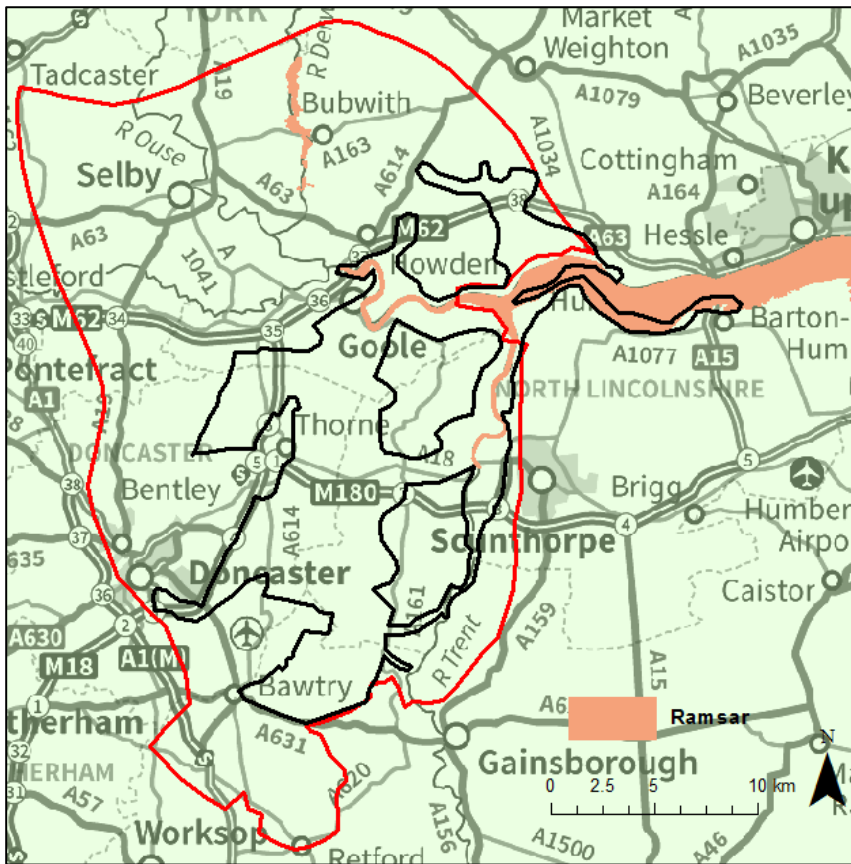


Figure 3.4: Ramsar sites in the HGL (data acquired from Natural England).

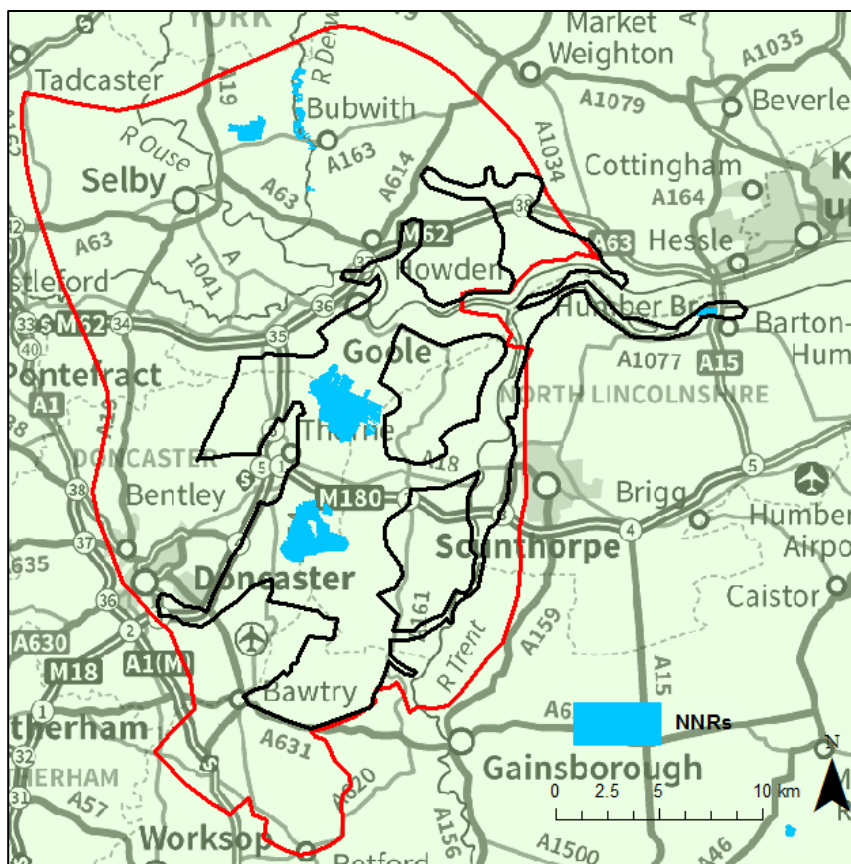


Figure 3.5: NNRs in the HGL (data acquired from Natural England).

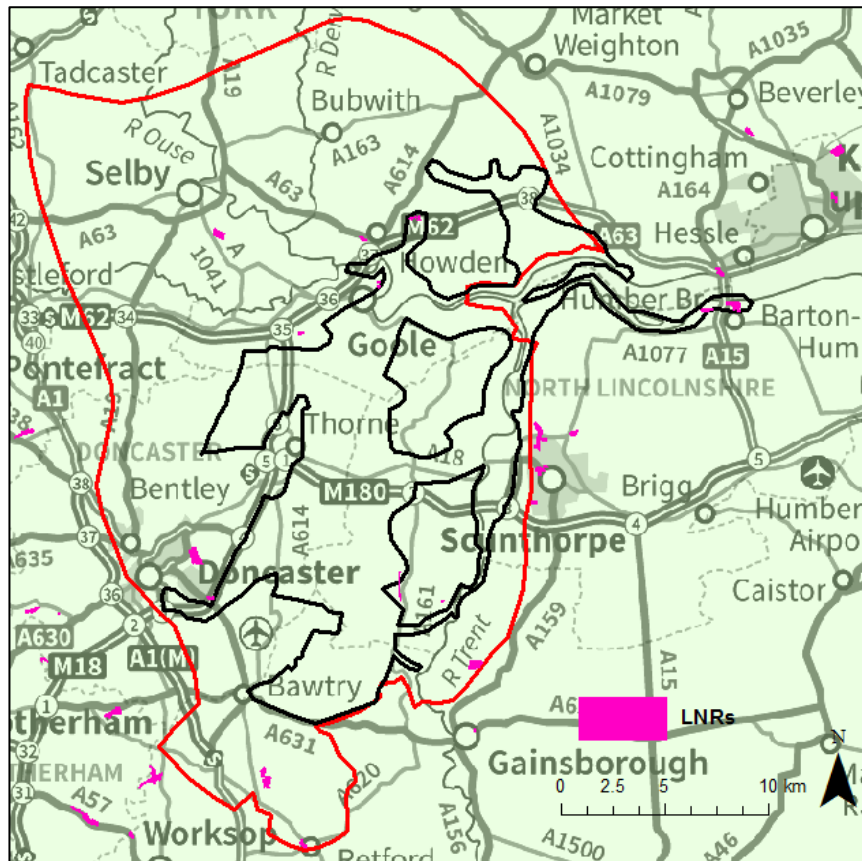


Figure 3.6: LNRs in the HGL (data acquired from Natural England).

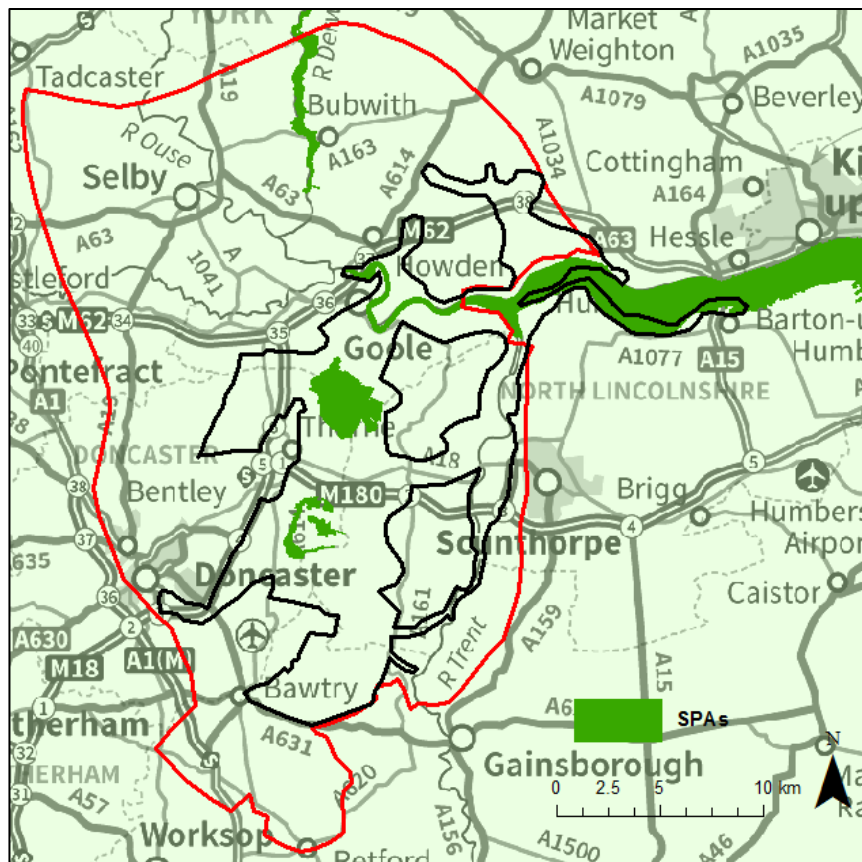


Figure 3.7: SPAs in the HGL (data acquired from Natural England).

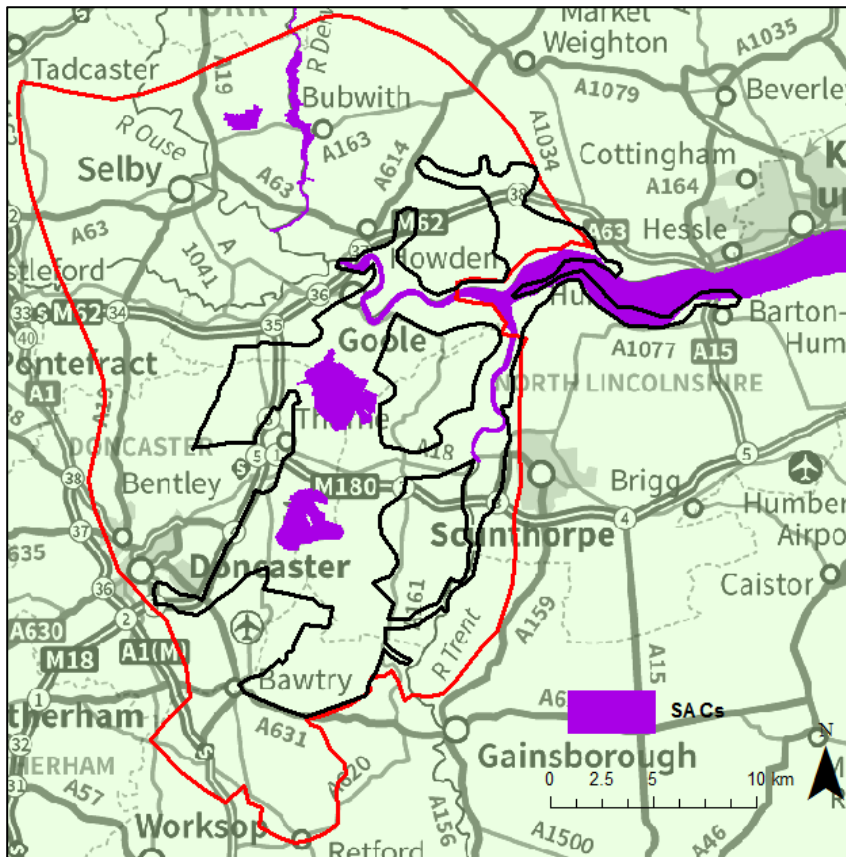


Figure 3.8: SACs in the HHL (data acquired from Natural England).

3.4.2 Priority habitats for wetlands (BAP habitats)

BAP wetland habitats were chosen as a metric in order to better understand the distribution of current wetland habitats found across the HHL with potentially high biodiversity levels, and were included in both literature and policy (Natural England, 2011; Overmars et al., 2013). These include Coastal and Floodplain Grazing Marsh, Lowland Fens, Lowland Raised Bogs and Reed Beds. All four were mapped using GIS files from the JNCC and can be seen in Figures 3.9 to 3.12. Coastal and Floodplain Grazing Marsh (Figure 3.9) is defined as “periodically inundated pasture, or meadow with ditches which maintain the water levels” (Defra, 2008a). Across the HHL there are many patches, often found near rivers, but there are also larger areas to the western side on the NIA boundary. Figure 3.10 displays the Lowland Fen habitat across the HHL with most patches found in the two NNRs or along various river banks. These are defined by Defra (2008b) as “peatlands which receive water and nutrients from the soil, rock and ground water as well as from rainfall”.

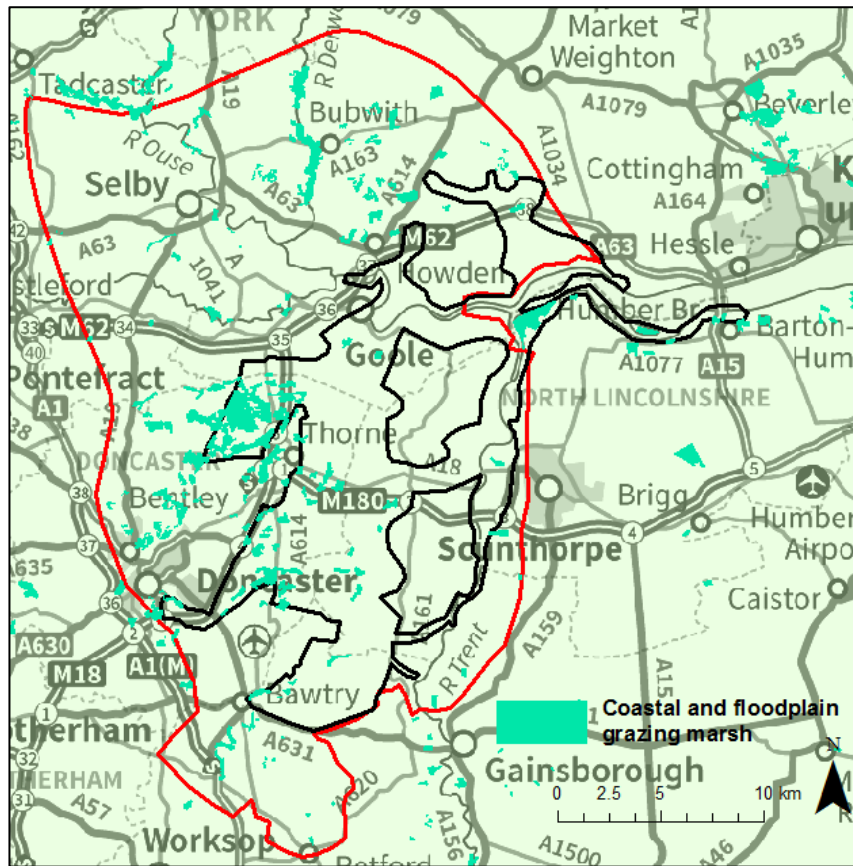


Figure 3.9: Coastal and floodplain grazing marsh in the HHL (data acquired from the JNCC).

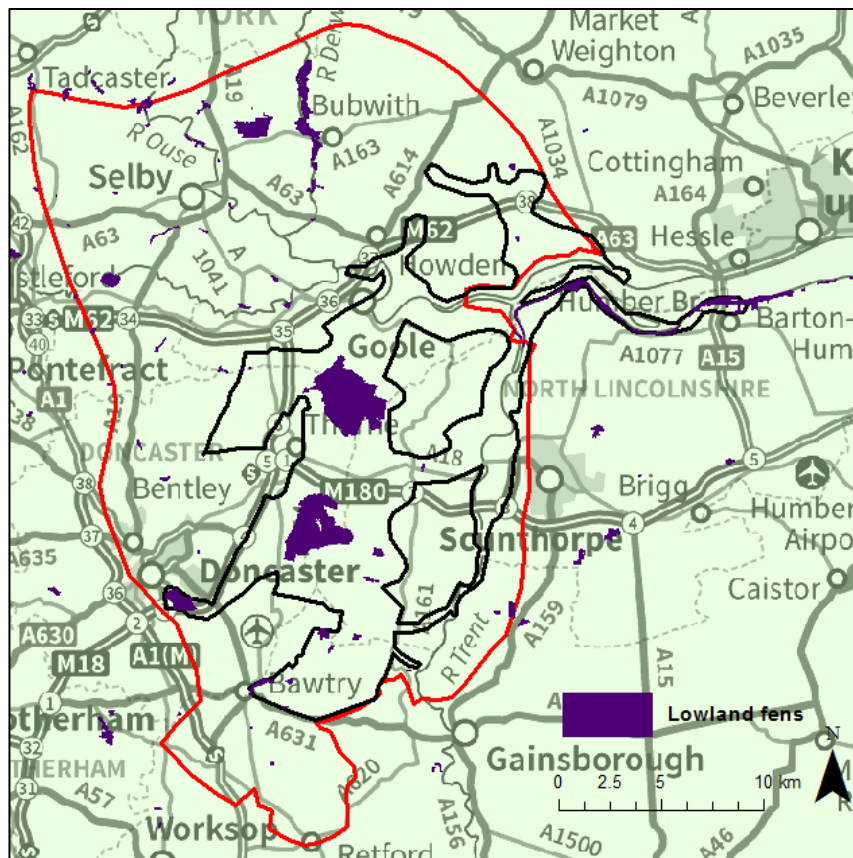


Figure 3.10: Lowland fens in the HHL (data acquired from the JNCC).

Lowland Raised Bogs (Figure 3.11) are defined as peatland ecosystems that develop in areas of poor drainage, which lead to anaerobic conditions allowing the slow decomposition of organic material (Defra, 2008c). In the HHL these are mainly found within the two NNRs: Thorne and Hatfield Moor. Figure 3.12 shows the distribution of Reed Beds across the landscape, with most patches either found along the Humber banks or to the western side of the HHL. Reed beds are defined by Defra (2008d) as “wetlands dominated by stands of the common reed *Phragmites australis*, wherein the water table is at or above ground level for most of the year”. Understanding the distribution of all of these habitats will be useful with the restoration options in later chapters, as these could be areas for wetland enhancement or expansion. In wider decision making, these habitats will be useful for building upon the concept of ecological networks, as these should be areas in relatively good condition. Again, the data is also relatively easy to read and use, meaning that decision makers should be relatively comfortable using these maps as part of their process.

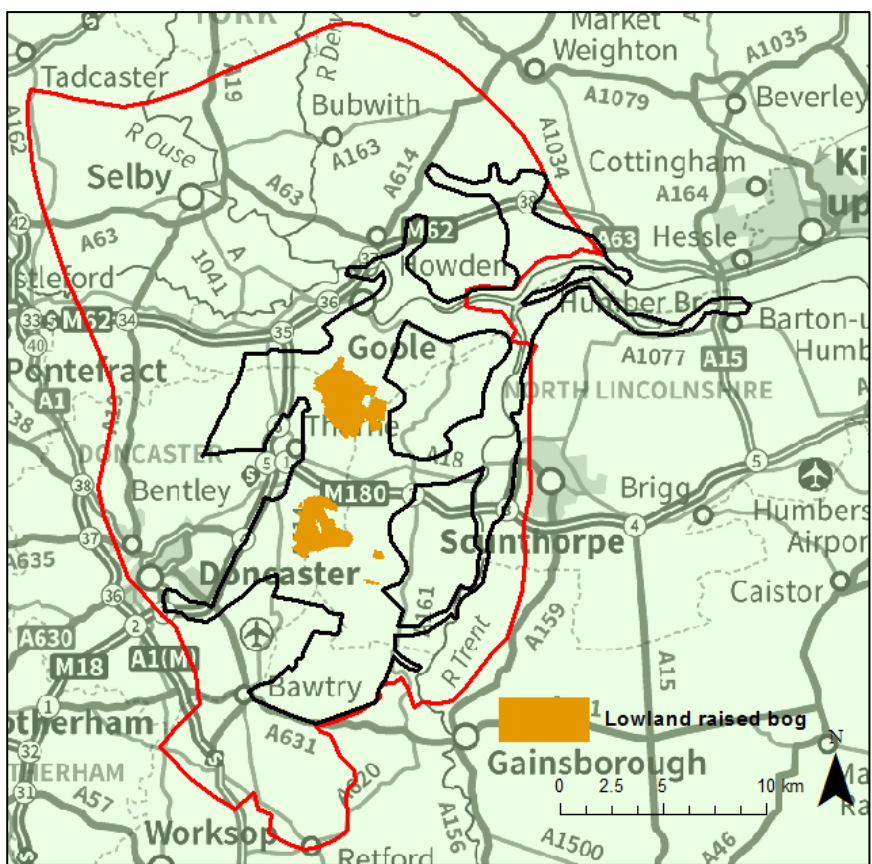


Figure 3.11: Lowland raised bogs in the HHL (data acquired from the JNCC).

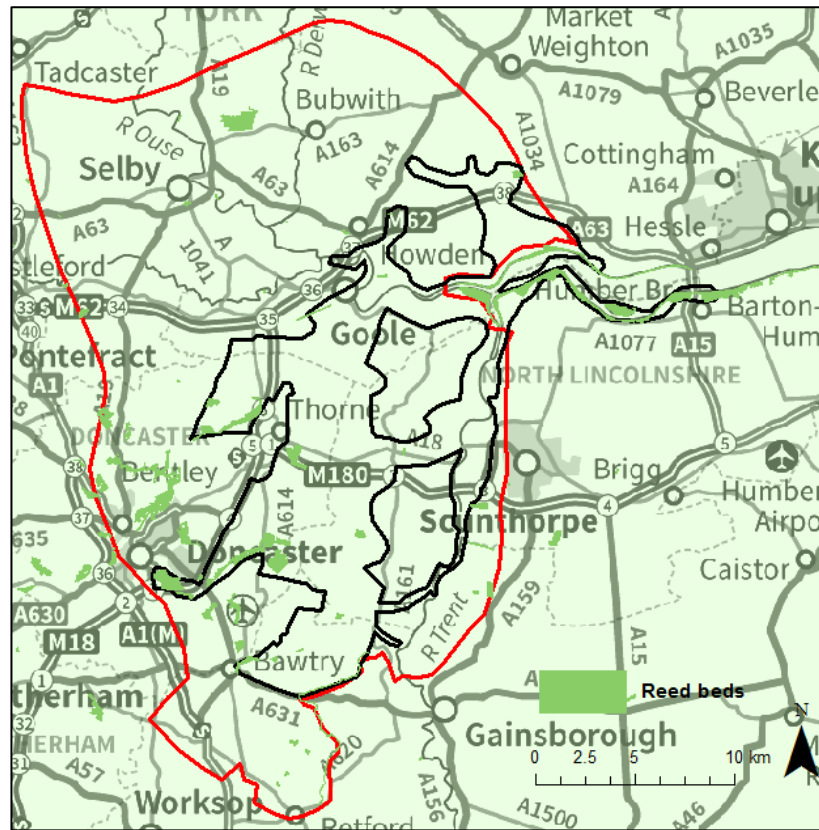


Figure 3.12: Reed beds in the HGL (data acquired from the JNCC).

3.4.3 Extent of habitat in favourable or recovering condition

Habitat in favourable or recovering condition was selected to understand how the HGL has been improved through work carried out by the Partnership. It was also set out in the NIA Monitoring and Evaluation framework, defined as the work recorded under the Biodiversity Action Recording System (BARS) to maintain habitats to a good condition (Defra and Natural England, 2012). In Figure 3.13 the work carried out by the HGL Partnership during the NIA project is displayed, obtained from the Programme Manager (based at YWT) in April 2015. Many different types of work were carried out, ranging from putting in ponds and buffer strips, to improving access for people. Not all of these improvements will have contributed to improving biodiversity, but it is difficult to understand each piece of work and its implications from this data. Overall, a large proportion of these improvements are to grasslands or implementing buffer strips, both of which could have positive implications for biodiversity. This information could help with decision making as it gives details of the current state of restoration work to then build on for the future. However, this map is unclear, as there is a lot of information and the meaning of each category is unknown, which would mean it would be harder to make decisions.

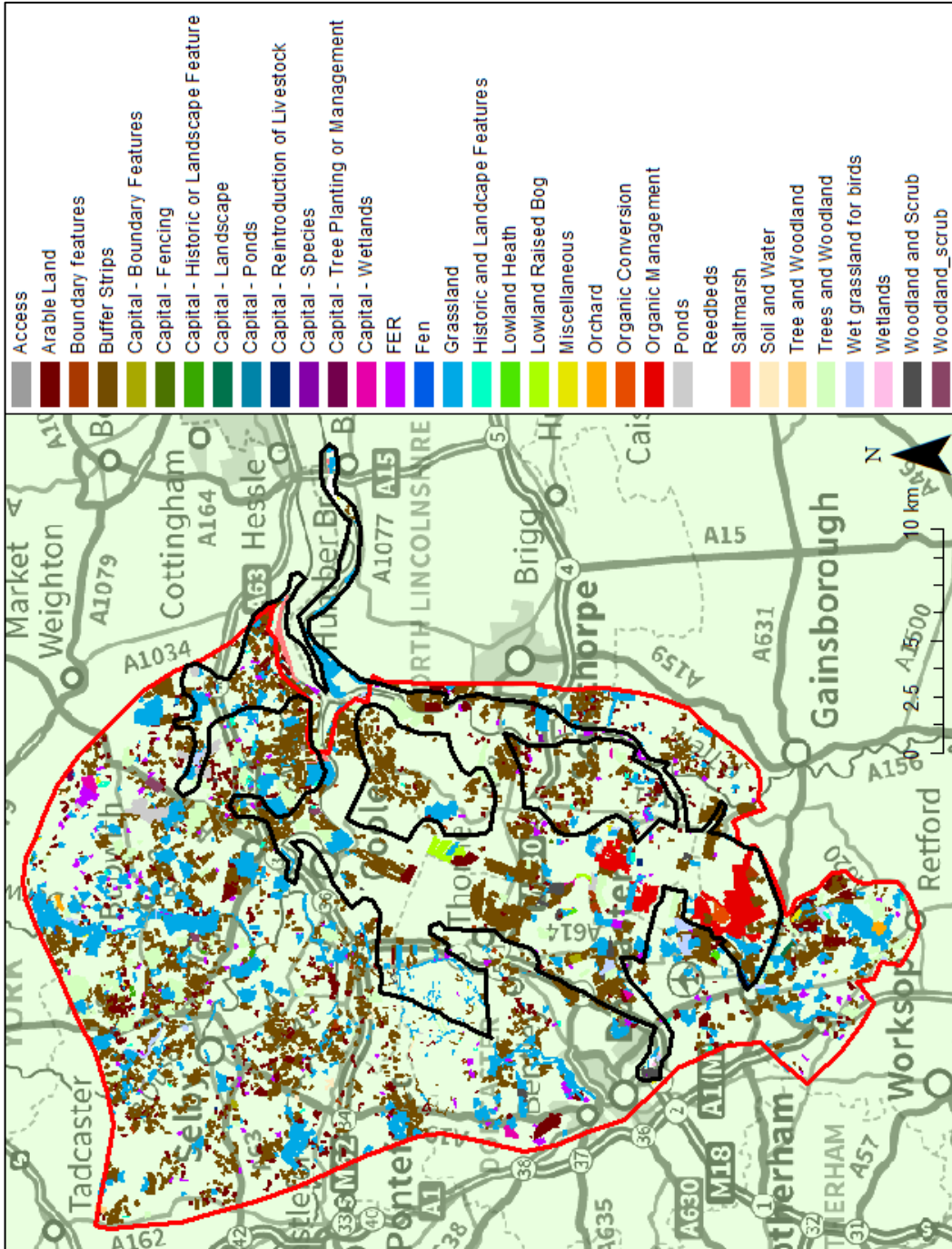


Figure 3.13: Work carried out in the HHL during the NIA programme, as recorded in BARS.

3.4.4 Status of widespread species

Widespread species were chosen as an indicator as they were also used by the HHL NIA Monitoring and Evaluation Group (Defra and Natural England, 2012). It is defined in the NIA regulations as the status of widespread species that are relevant to England Biodiversity 2020 Indicators, but the Monitoring and Evaluation regulations and the HHL Partnership were suggesting it as a way to look at groups of widespread species, for example wetland species (Defra and Natural England, 2012). Therefore, as it is useful to look at groups of important species, this is the direction this category will take for the rest of the thesis, although it will continue to be called “widespread species”.

A sub-set of seven groups of species were selected based on suggestions from the Monitoring and Evaluation Framework (Defra and Natural England, 2012), the Biodiversity 2020 targets (Defra, 2012) and taking into account the landscape context. The data was acquired from the NBN (National Biodiversity Network) Gateway. There may be some bias in the data from this source, as it depends upon who submits to the gateway, the locations where they choose to record and how often certain species are deliberately targeted, and most of the data provided to the NBN is this unstructured and opportunistic data. There have been efforts to ameliorate these problems with citizen science in other projects, but these were outside the scope of this work. Potential solutions include statistical techniques and encouraging better collection of data to be provided to the NBN.

Breeding wetland birds, wintering water birds and wetland habitat plants were chosen from the Monitoring and Evaluation information and England Biodiversity Indicators 2020 (Defra and Natural England, 2012; Defra, 2012). Figure 3.14 displays the distribution of breeding wetland birds across the HHL, of which there are very few sightings according to the data source. However, there is a particularly large group around the Potteric Carr Nature Reserve, suggesting some locational bias in the recording. Wintering water birds are displayed in Figure 3.15, which again shows few sightings, but a high distribution at Potteric Carr Nature Reserve. In Figure 3.16 there are more records of wetland habitat plants, with many of these occurring within the NIA boundary.

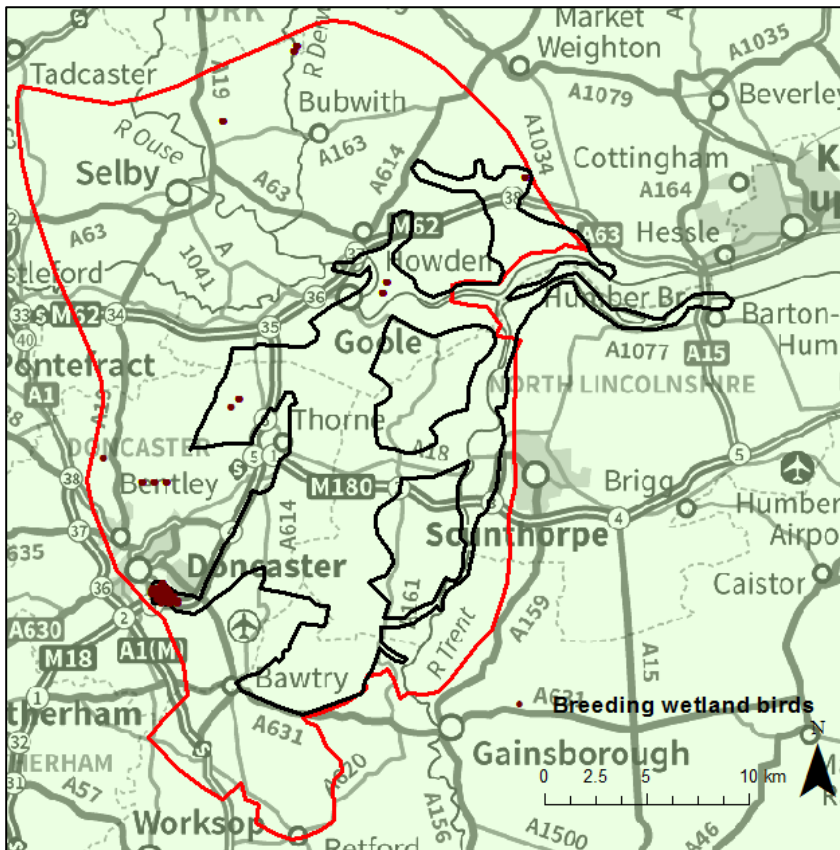


Figure 3.14: Breeding wetland birds observations in the HHL (data acquired from the NBN).

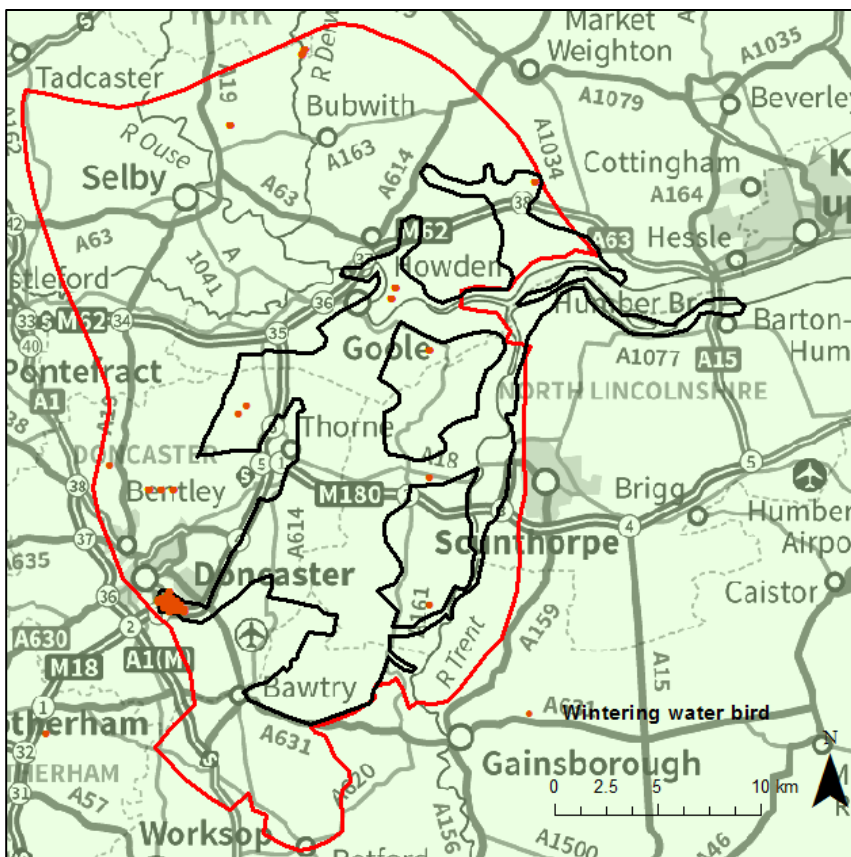


Figure 3.15: Wintering water birds observations in the HHL (data acquired from the NBN).

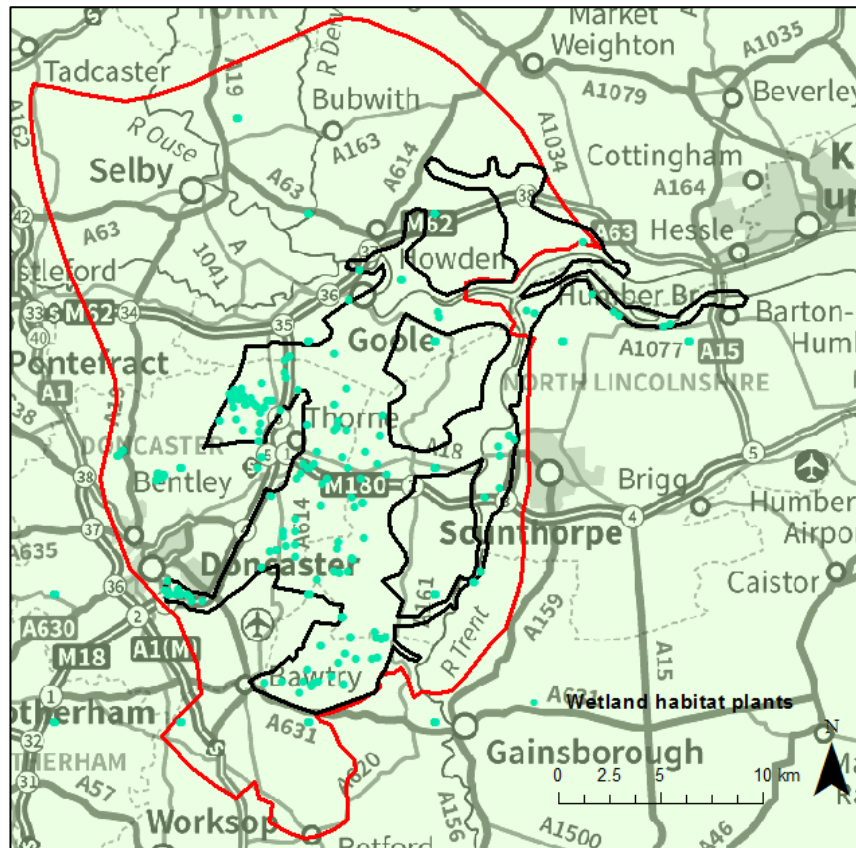


Figure 3.16: Wetland habitat plants observations in the HHL (data acquired from the NBN).

Four further groups of species were chosen after observations of the HHL Partnership meetings where they were selecting their own metrics. These were butterflies, bees, bats and dragonflies, and again all the data was obtained from the NBN Gateway. Figure 3.17 displays the distribution of butterflies, the majority of which appear within the NIA boundary, with again a high distribution at Potteric Carr Nature Reserve. The distribution of bees is seen in Figure 3.18, which shows that there is a fairly even distribution across the HHL, with slightly more to the western side. Bats are displayed in Figure 3.19, with many more sighted on the eastern and southern sides. Finally, Figure 3.20 shows the distribution of dragonflies, with the majority sighted within the NIA and a proliferation at Potteric Carr Nature Reserve. Understanding the distribution of these groups of species will help with the restoration options, as they suggest where features might already occur that are conducive to wetlands. However, care will need to be taken with the bias discussed, especially around Potteric Carr. These maps could also be useful in wider decision making, but would need to be used by people with expertise as to what the presence or absence of each species means, and would be improved with more data.

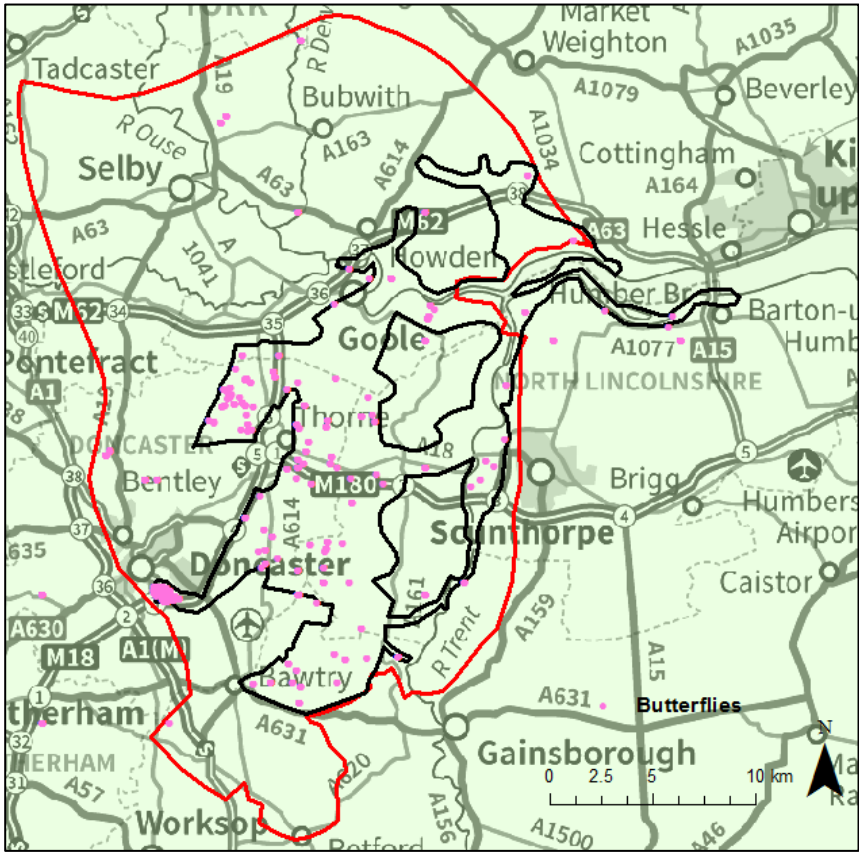


Figure 3.17: Butterfly observations in the HHL (data acquired from the NBN).

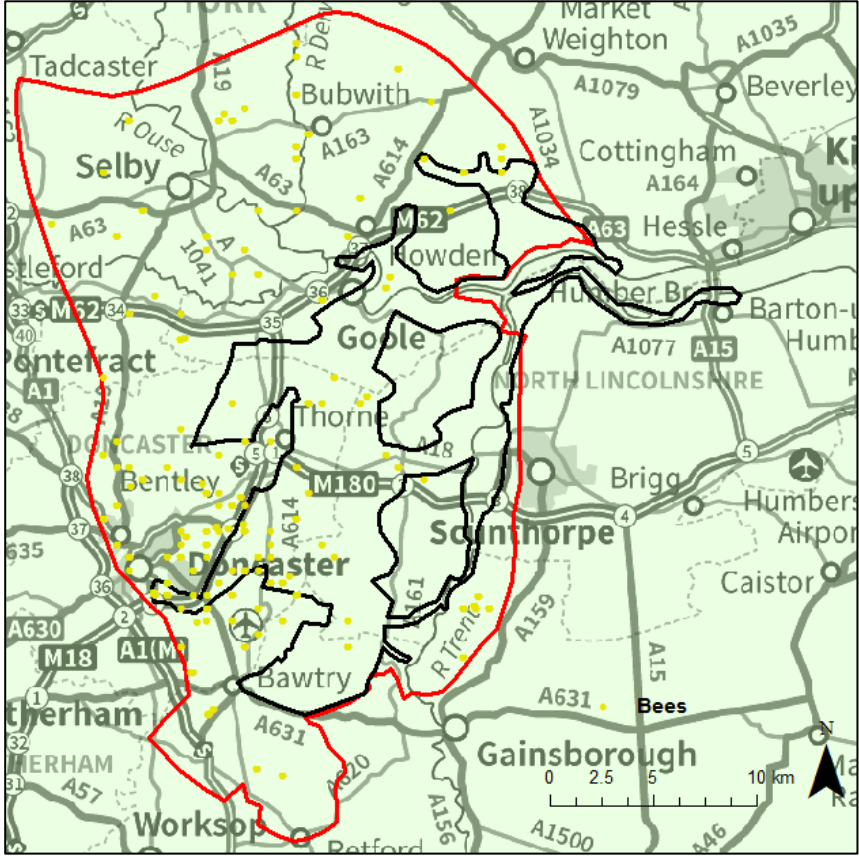


Figure 3.18: Bee observations in the HHL (data acquired from the NBN).

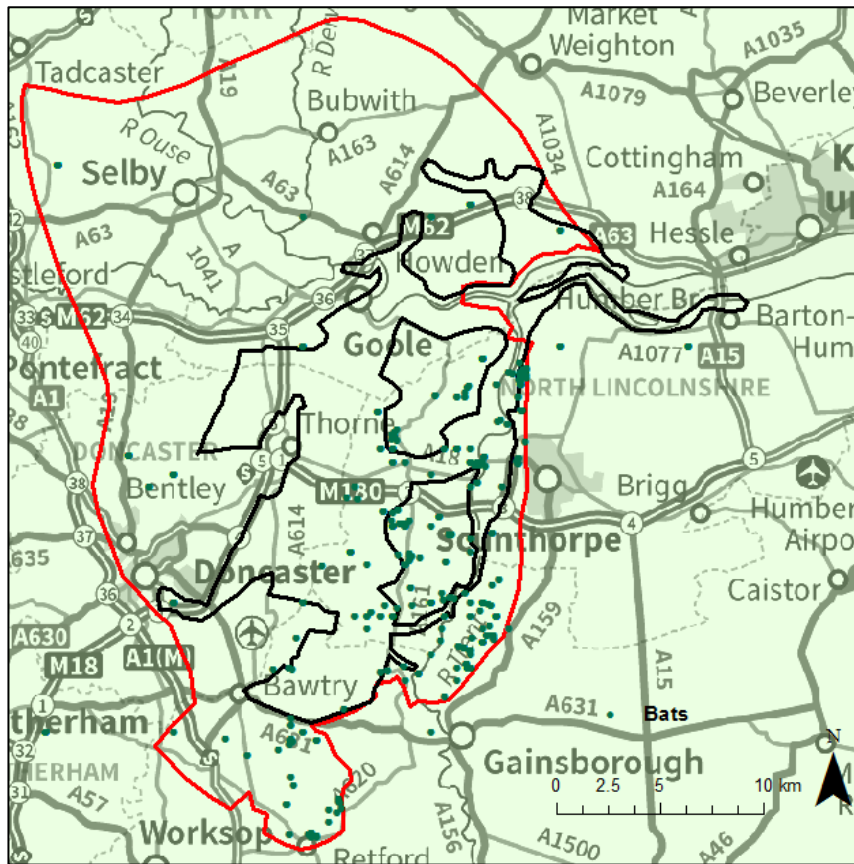


Figure 3.19: Bat observations in the HHL (data acquired from the NBN).

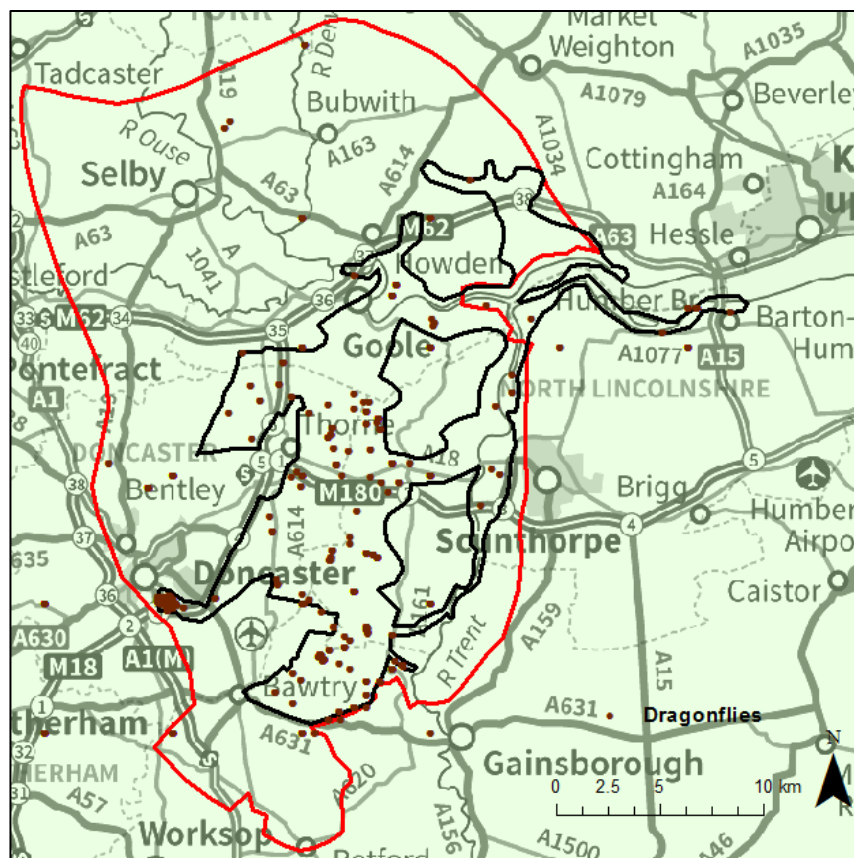


Figure 3.20: Dragonfly observations in the HHL (data acquired from the NBN).

3.4.5 Status of focal species

Again this indicator was chosen as it was also being used by the HHL Partnership, in order to understand the state of particular species in the landscape (Defra and Natural England, 2012). It was defined in the Monitoring and Evaluation regulations as the “status of species that are a focus for action or of drivers of change that are a specific concern within the NIA” (Defra and Natural England, 2012). From investigating the literature, policy documents and HHL Partnership information there were many species that could have been selected. Eleven were chosen using the Monitoring and Evaluation Framework suggestions (Defra and Natural England, 2012), the Biodiversity 2020 targets (Defra, 2012), the context of the HHL and trying to cover a wide range of species. Again, all the data was collected from the NBN Gateway. These eleven are: water voles, bittern, nightjars, cranes, newts, otters, sphagnum, sedge, plover, duckweed and banded demoiselle dragonfly. Figure 3.21 displays the distribution of water voles across the HHL and shows that most have been observed on the southern side of the landscape, but bias in the dataset may need to be considered, as with the previous section. The distribution of bittern across the HHL landscape is displayed in Figure 3.22 with nearly all of the sightings in or around Potteric Carr Nature Reserve, suggesting more locational/observational bias. Nightjar distribution is displayed in Figure 3.23, with very few sightings in the HHL.

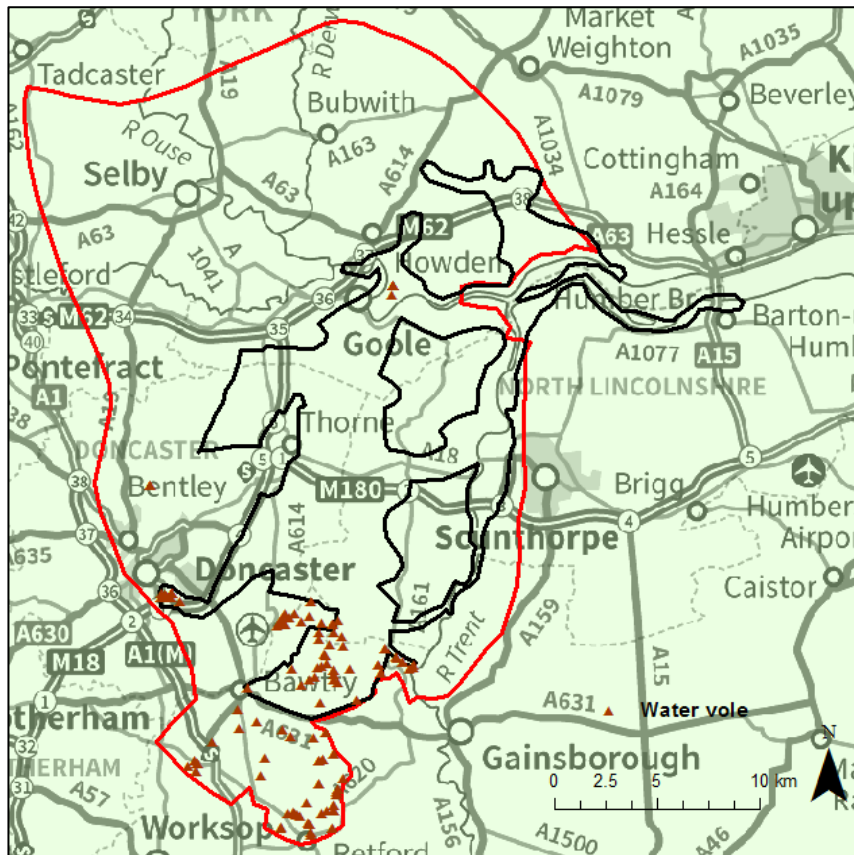


Figure 3.21: Water vole observations in the HHL (data acquired from the NBN).

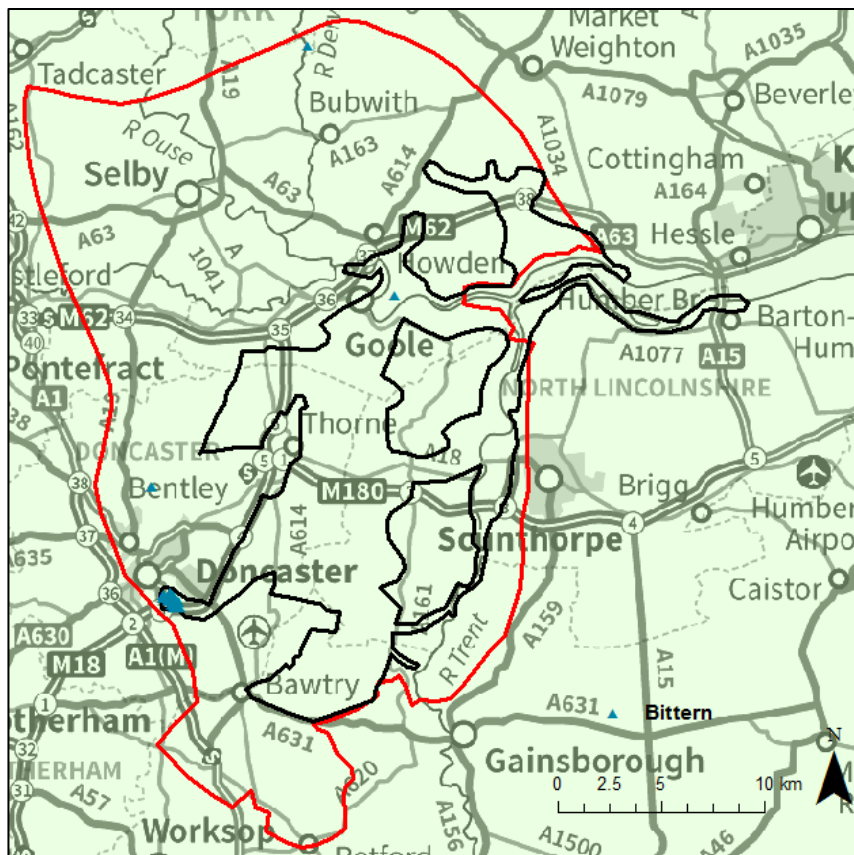


Figure 3.22: Bittern observations in the HHL (data acquired from the NBN).

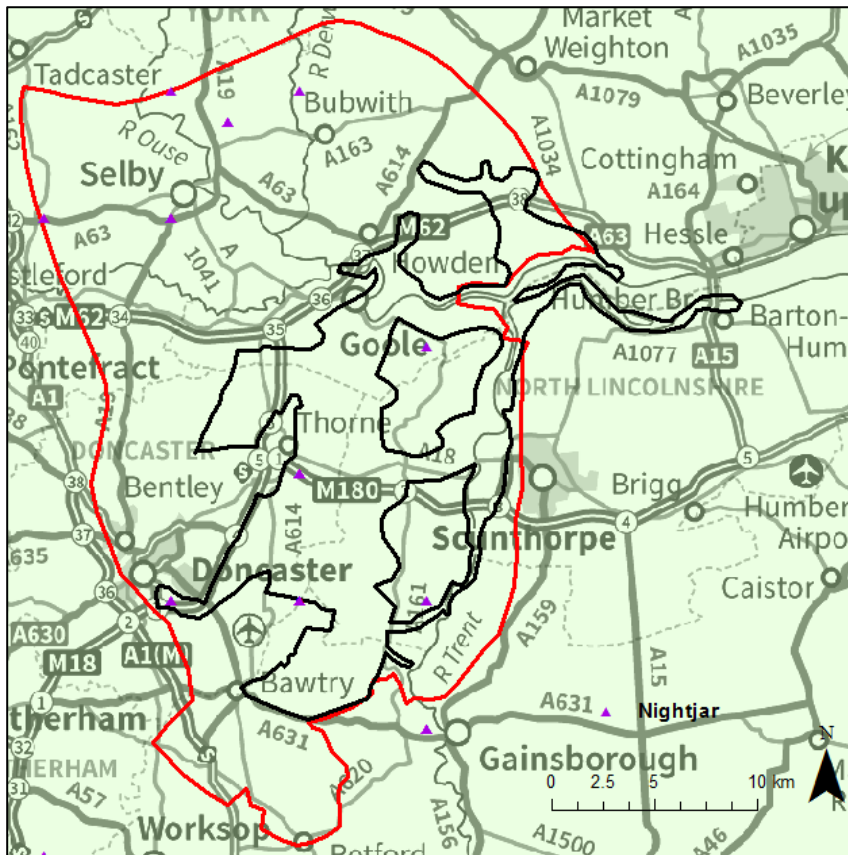


Figure 3.23: Nightjar observations in the HHL (data acquired from the NBN).

Figure 3.24 displays the distribution of cranes across the landscape, and the number of sightings is very low. The distribution of newts can be seen in Figure 3.25 and these are also very infrequent, with only five sightings across the whole HHL landscape according to this data source. In Figure 3.26 otters are seen to be more frequent, with a relatively even distribution across the landscape. Sphagnum, as displayed in Figure 3.27, has been infrequently recorded in the landscape, with most existing records found around the two NNRs.

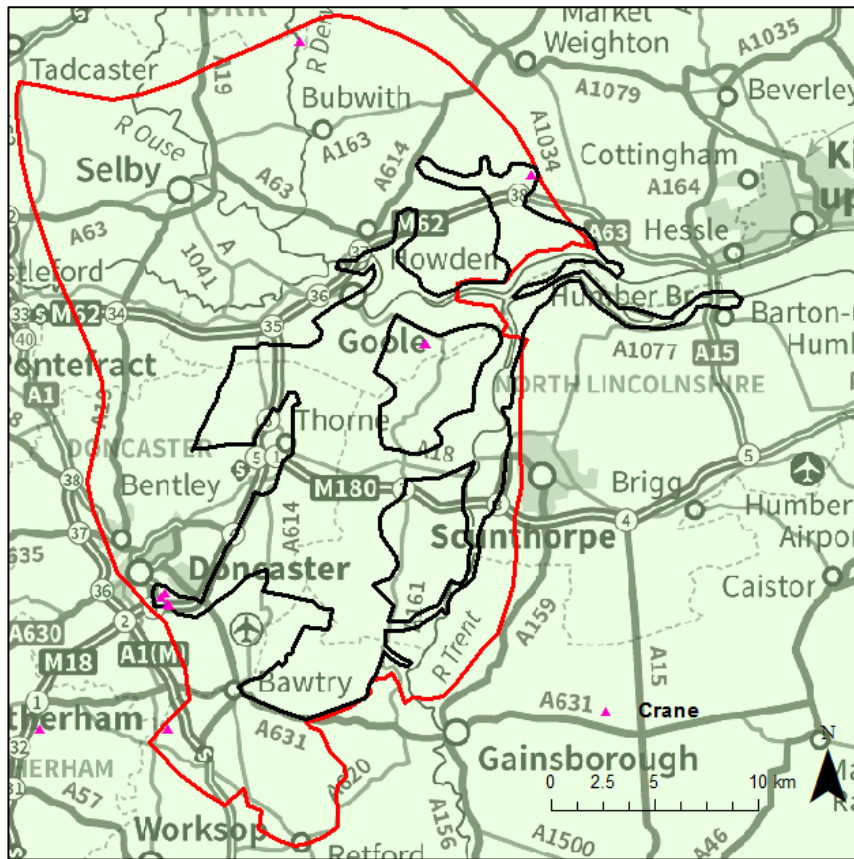


Figure 3.24: Crane observations in the HHL (data acquired from the NBN).

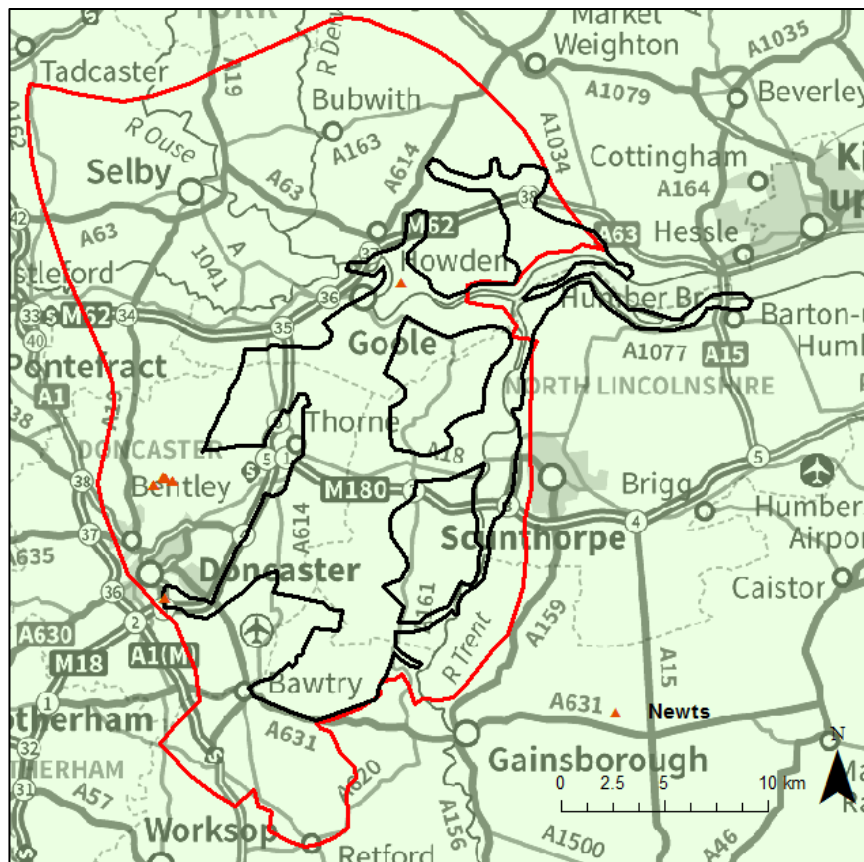


Figure 3.25: Newt observations in the HHL (data acquired from the NBN).

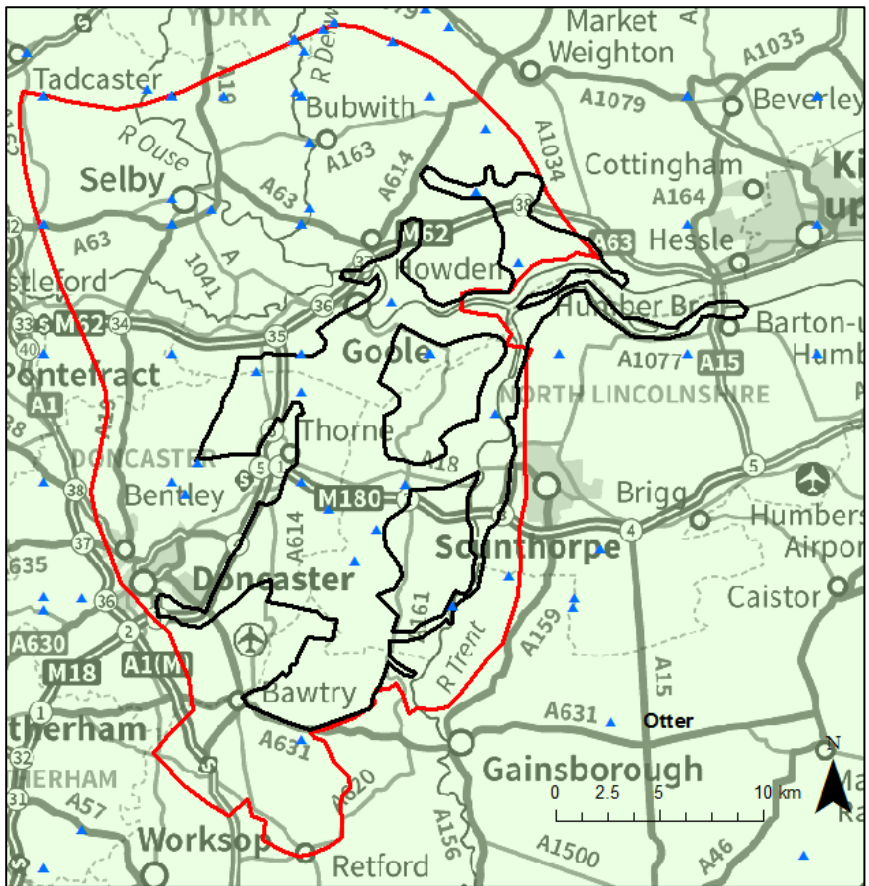


Figure 3.26: Otter observations in the HGL (data acquired from the NBN).

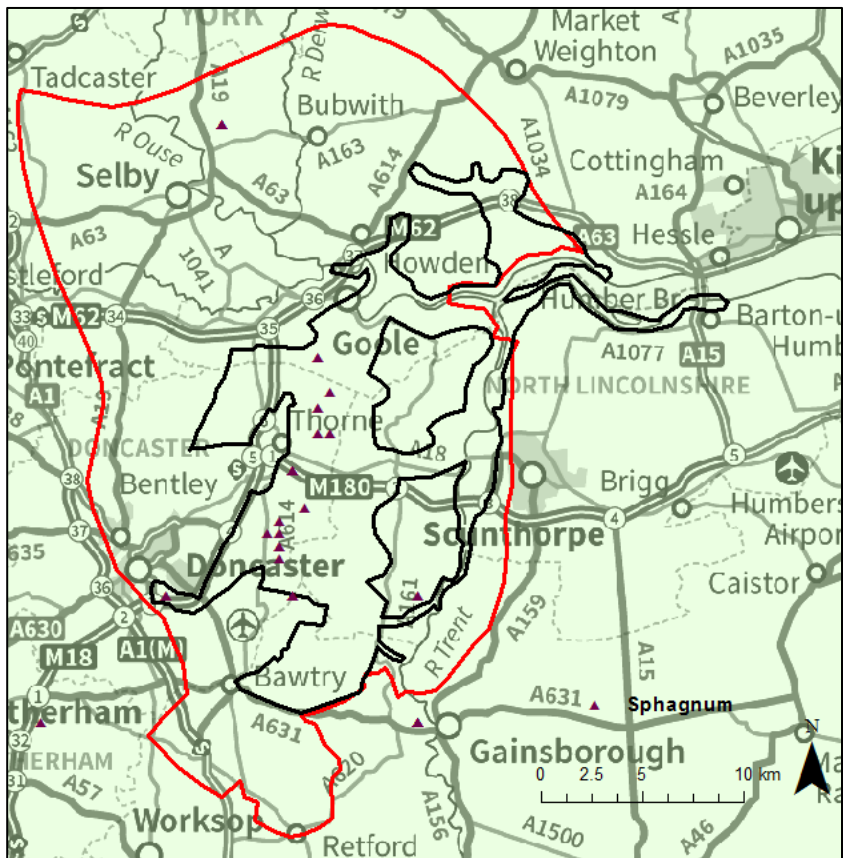


Figure 3.27: Sphagnum observations in the HGL (data acquired from the NBN).

Figure 3.28 shows the distribution of sedge across the HHL, with most records within the NIA boundary. The distribution of plover in Figure 3.29 shows that sightings are fairly infrequent, and most are at Potteric Carr. In Figure 3.30 the distribution of duckweed is displayed, which mostly occurs within the NIA boundary to the western side and in Hatfield Moors NNR. Duckweed is very common and often associated with eutrophication, and so it is more of a “dis-service” to the ecosystem. Finally, Figure 3.31 shows the distribution of banded demoiselle dragonflies, of which most are found along river corridors. Knowing the distribution of these particular species will help with developing restoration options, again by indicating the possible occurrence of certain characteristics in the landscape that are conducive to wetlands. Again, however the issue of recording bias, both locational and species preference, will need to be considered. As with the previous section, this information on focal species could be very useful for wider decision making, but would require an expert to understand what each species indicates for the landscape and its restoration, and to deal with the issues previously identified.

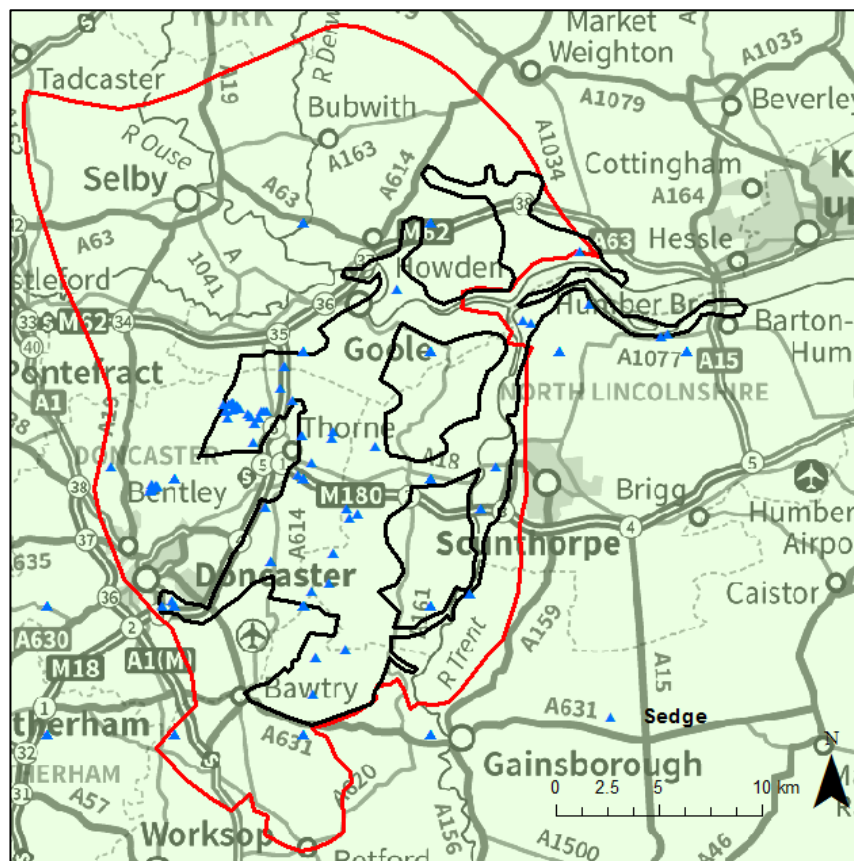


Figure 3.28: Sedge observations in the HHL (data acquired from the NBN).

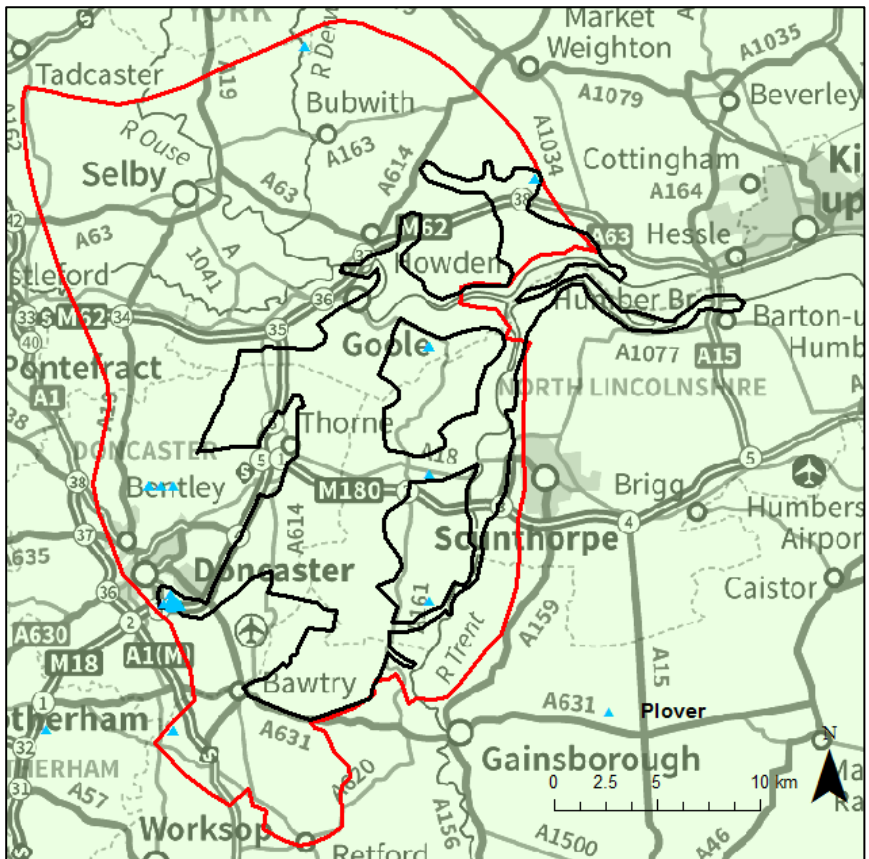


Figure 3.29: Plover observations in the HGL (data acquired from the NBN).

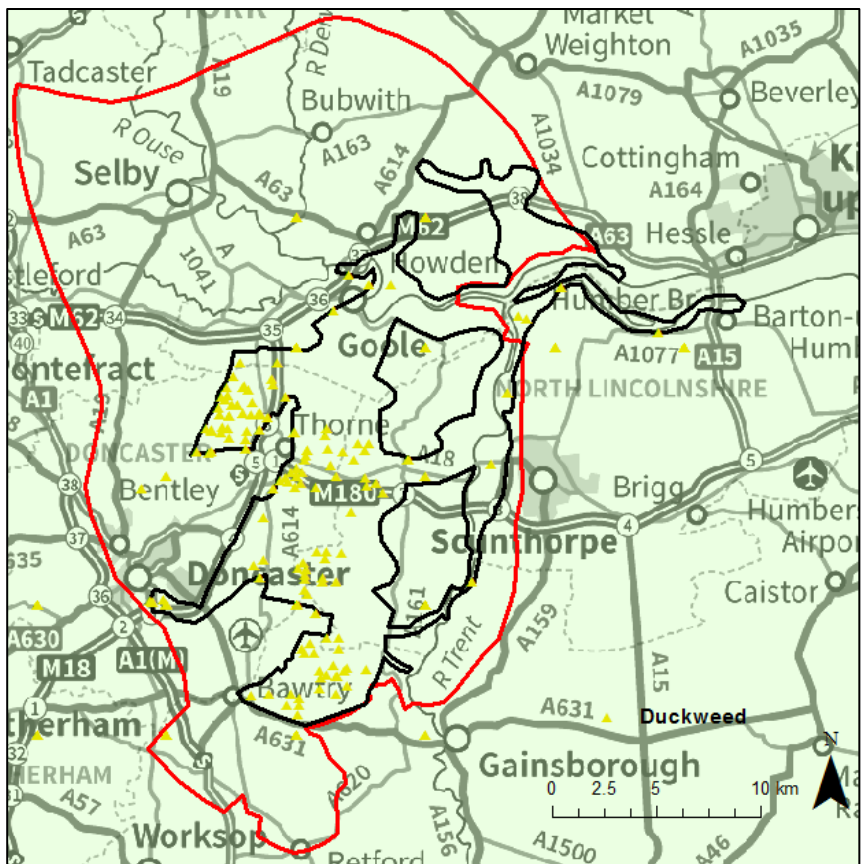


Figure 3.30: Duckweed observations in the HGL (data acquired from the NBN).

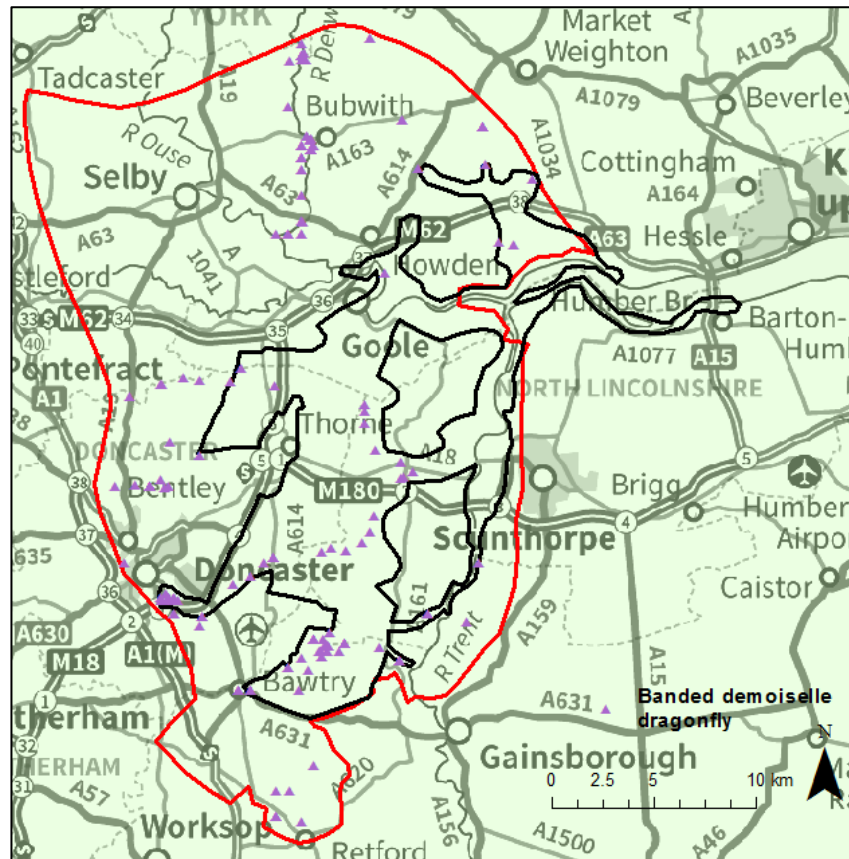


Figure 3.31: Banded demoiselle dragonfly observations in the HHL (data acquired from the NBN).

3.4.6 Species richness

In much of the literature, including the Biodiversity 2020 Strategy (Defra, 2012), species richness was an important biodiversity metric to consider and was proposed as a metric to be used across Europe (EEA, 2007). Species richness is important but should never be used alone (Jaunatre et al., 2013), as biodiversity is not just about richness (Balvanera et al., 2006). Comprehensive data of all species in the HHL landscape was extremely difficult to collect and so no map of suitable quality could be produced. Therefore, any map made would be incomplete and could introduce falsities into further analysis. Instead, more concentration will be given to the individual species and groups of species. In wider decision making, this information could be useful if a reliable data source existed in order to give an overall idea of biodiversity and wildlife distribution across the landscape.

3.4.7 Invasive non-native species

Invasive non-native species were included in order to understand how they might cause complications with biodiversity and restoration works, by being very difficult to eradicate. This metric was also suggested within the Biodiversity 2020 Strategy (Defra, 2012) and to be used across the whole of Europe (EEA, 2007). Himalayan Balsam and Japanese Knotweed were selected as these are the most prolific, with the data acquired from the NBN Gateway. Figure 3.32 shows the distribution of Himalayan Balsam across the landscape, with most records outside the NIA boundary on the western side of the landscape. Japanese Knotweed can be seen in Figure 3.33 and there is less in the HHL, with most occurrences on the southern side. These maps could provide useful information to wider decision making as they show areas where more work is required. However, as is the case with previous sections, it would require expertise to decipher the information in the correct way to make useful decisions. Therefore, these will be more difficult to work into restoration options and so will not be used in the forthcoming analysis.

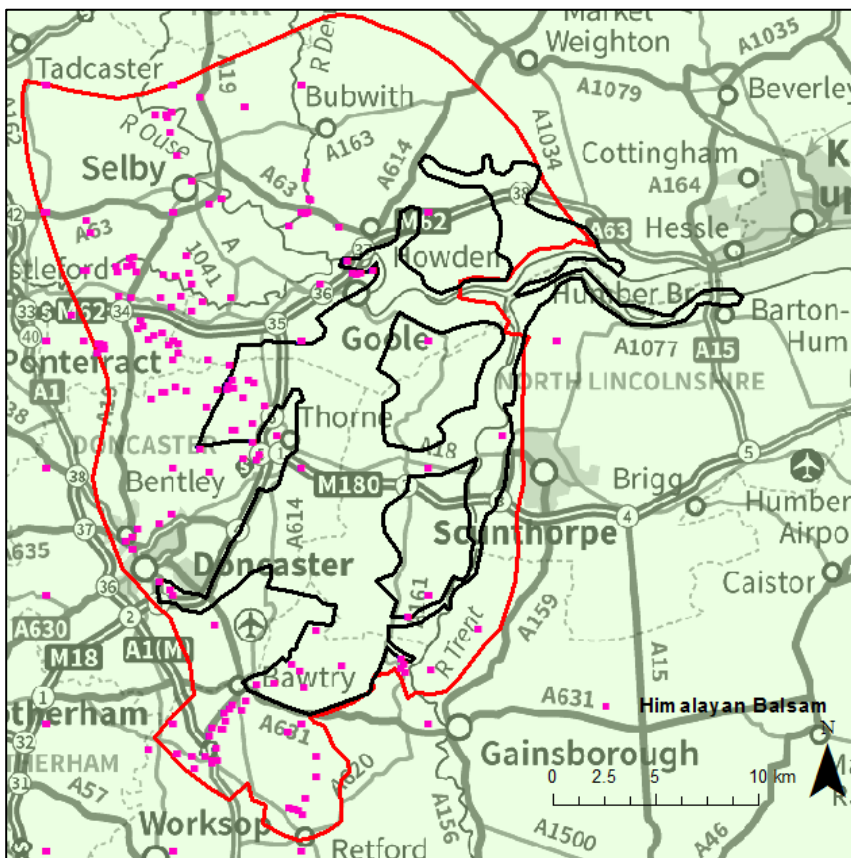


Figure 3.32: Himalayan balsam observations in the HHL (data acquired from the NBN).

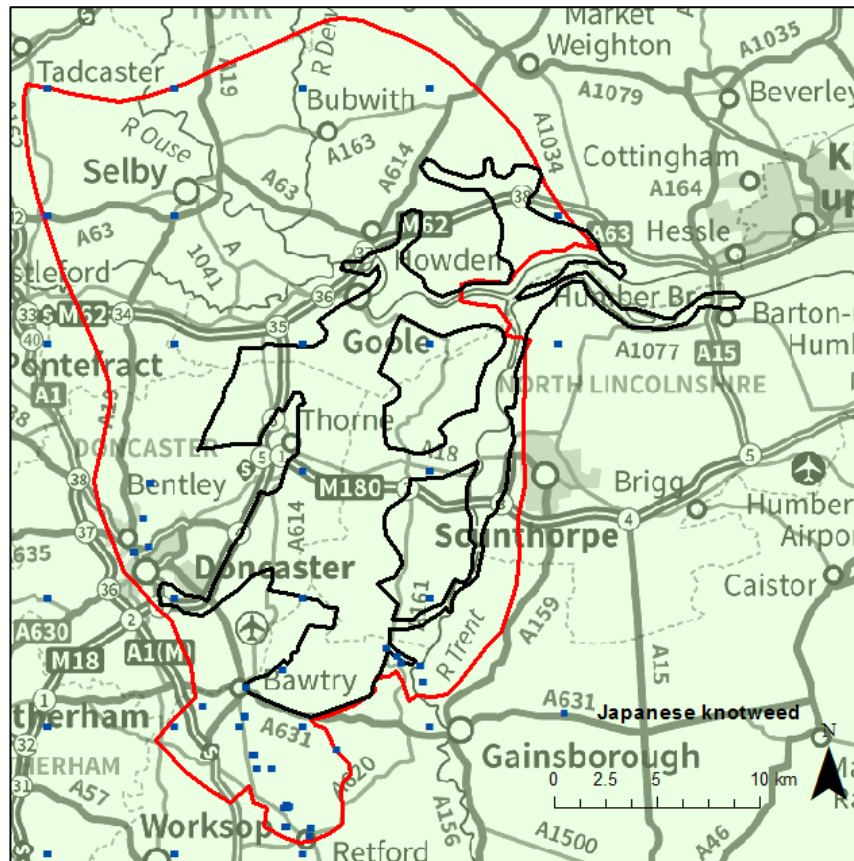


Figure 3.33: Japanese knotweed observations in the HHL (data acquired from the NBN).

3.4.8 Summary

By understanding the distribution of these indicators, a better understanding of biodiversity in the HHL landscape can be built up, which can then be fed into the development of options for wetland restoration. It is also important to consider whether the maps produced here are helpful to decision makers in their current format. Are they actually useful and what sort of decisions will/can be made from them? It will depend on the individual expertise of the people viewing them, their understanding of the data behind the maps and what each individual metric means for the landscape.

3.5 Ecosystem services mapping and modelling

As discussed previously, ecosystem services are much more difficult to select metrics for, as the research area is relatively new and there is a lack of scientific understanding, especially with cultural services (Rey Benayas et al., 2009;

Cardinale et al., 2012). There is also a huge range of ecosystem services and so a small number needed to be selected based on the particular ecosystem (Karjalainen et al., 2013). The HHL Monitoring and Evaluation Group identified four ecosystem services from the NIA options, which they thought were the most relevant to the HHL, and so these will be used in here too. By choosing services that are complimentary to those the HHL Partnership choose it may allow for better comparison between the different approaches in later analysis, as well as providing a useful basis for selecting relevant ecosystem services to map. Other appropriate services are found in the HHL NIA Business Plan (HHL Partnership, 2011), which are more context specific, and so will be perhaps more relevant. Many models have been developed to understand the spatial distribution of ecosystem services, which will be further investigated below. Then each of the services selected is reasoned, mapped and discussed.

3.5.1 Ecosystem services models

Two models will be used to help understand some of the ecosystem services selected: InVEST and LUCI. It would have been interesting to also use EcoServ (developed by The Wildlife Trusts), but the model was not available. Other models were considered, such as TESSA and ARIES, but the two chosen were the most appropriate for the HHL landscape and the results required because of their accessibility and range of services included. By using two models it hoped that more realistic information will be produced, as occurred with Vorstitis and Spray (2015) who used three different models (including InVEST) to map ecosystem services in a catchment in the Scottish Borders.

3.5.1.1 InVEST

The MEA states that ecosystem services need to be integrated into decision making and policy, but this has not occurred partly as the science still needs to advance (Daily et al., 2009). Daily et al. (2009) suggests that this is the reasoning behind the InVEST model. The InVEST acronym stands for the Integrated Valuation of Environmental Services and Trade-offs (The Natural Capital Project, no date b). It was developed by the Natural Capital Project, with natural capital

equating to ecosystem services. It is a freely available suite of 16 models, that maps and values goods and services (WBCSD, 2013; The Natural Capital Project, no date b). The toolkit can be used to answer a range of questions in terms of trade-offs, and balancing environmental and economic needs, to be used by a variety of organisations (The Natural Capital Project, no date b). The toolkit includes many services, such as: carbon, biodiversity, flooding, water quality, agricultural products and recreation (Nelson et al., 2009; The Natural Capital Project, no date a).

The process of the InVEST toolkit can be seen in Figure 3.34. Some InVEST projects start with stakeholder consultations that are used to identify management choices and policy options through scenarios of potential future land uses (WBCSD, 2013; The Natural Capital Project, no date b), although this is not necessary. The models can then estimate the current and future location, amount, delivery and value of ecosystem services (The Natural Capital Project, no date b), using both biophysical and economic input data (The Natural Capital Project, no date a). A major part of InVEST is the usage of maps as the inputs and outputs, meaning that the ecosystem services become spatially defined. The model is flexible as it can be used over a variety of scales (The Natural Capital Project, no date b). As many parts of the world experience data scarcity, there are simple models within the toolkit that require minimal data inputs (The Natural Capital Project, no date b). The models are run through ArcGIS, are a free and open source and there is a comprehensive User's Guide (The Natural Capital Project, no date a, b).

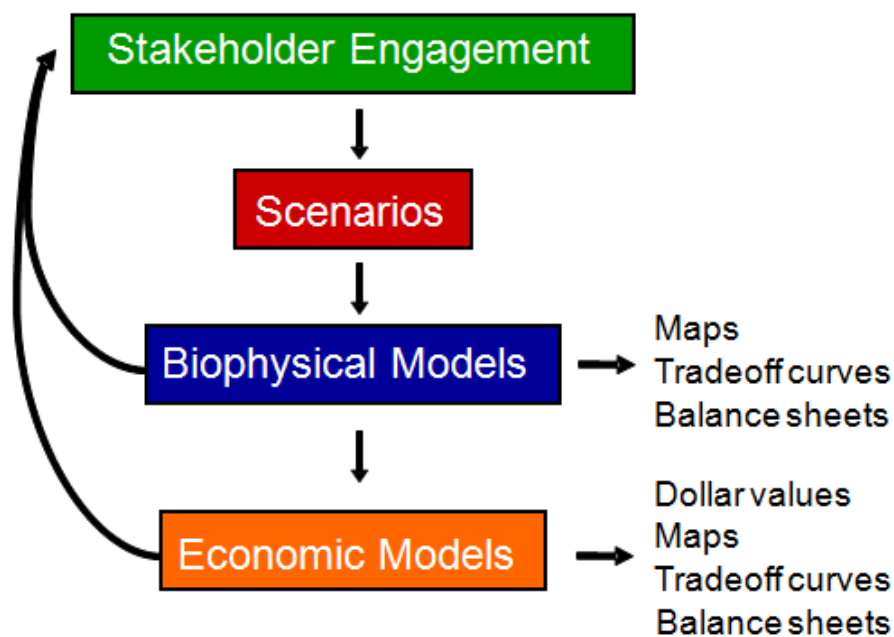


Figure 3.34: The process of the InVEST model (The Natural Capital Project, no date b).

InVEST has been used globally, including in Indonesia, the USA and China, for many case studies (Lautenbach et al., 2012; The Natural Capital Project, no date a). Van der Biest et al. (2014) felt that the toolkit misses out on the supporting services, mostly the biophysical potential for the delivery. BSR (2011) found that InVEST was weak for dealing with groundwater. However, they did also find strengths, including that it looked at a whole landscape, it was GIS-based and it was more focused on ecological data than many other models (BSR, 2011). InVEST appears to be stronger on the biophysical side than the valuation side, and Daily et al. (2009) stated that it is very flexible and can be used with a wide variety of data. Peh et al. (2013) also stated that the model can be run at a good resolution and is cheap in terms of time and man power, but required a high level of computing skills and technical knowledge. Overall, it appears that InVEST is a relatively well-developed tool that has been applied in many different areas and works with fairly minimal inputs.

3.5.1.2 LUCI

LUCI is a newer model and so is not well-established in the academic literature. It was formally called Polyscape and is put forward as a simple and transparent open-source tool, which can involve stakeholders (Bagstad et al., 2013b). It has

been developed using the Plynlimon catchment in Wales as a case study and has been run at a couple of other study sites, mostly agricultural based landscapes, but there is funding to test it further (Jackson et al., 2013a). The outputs mostly come in the form of maps, with more concentration on the biophysical models and less so on economic valuation. It was developed with farmers and scientists to better understand the provision and level of services, especially regulatory ones, to make them more spatially explicit, improve the economics and reduce environmental impacts under different scenarios (Jackson et al., 2013a, b; Bagstad et al., 2013b). However, it is meant more as a negotiation tool, rather than to feed directly into decision making (Jackson et al., 2013b). It is also meant as a tool for identifying areas where further scientific investigation would be valuable (Jackson et al., 2013b). A range of services are considered, including: food production, carbon, flooding, water quality and habitat, and there are plans to add cultural services (Jackson et al., 2013a, b).

LUCI has many advantages, including that it uses nationally available data, is good at the small scale, is fast-running, uses GIS and can use local knowledge (Jackson et al., 2013a, b; Bagstad et al., 2013b). There are only three national datasets that are absolutely necessary for input: soils, a land use/land cover (LULC) map and a Digital Elevation Model (DEM). Algorithms are then used to understand location and values to produce the results (Jackson et al., 2013a, b). The model does have quite a simple approach, as it is built for stakeholder use and it is stated that it should only take one day to run (Jackson et al., 2013b). Jackson et al. (2013b) admit that there are methodological assumptions and data deficiencies, but these are made transparent.

Before starting to run the individual models in LUCI there are two pre-processing steps to complete that provide the basis for all of the ecosystem service models. The first is called "Generate LUCI HydTopo Inputs" and produces hydrological and topographical information. The inputs required are a DEM (obtained from Edina) and a file of the rivers network (optional, obtained from Natural England). A set of intermediate files that feed into the second pre-processing step called "Generate LUCI Scenario with Landcover/Soil Only" are then produced. The resolution of the DEM defines the resolution of the outputs for the rest of the model (Jackson et al., 2013b). This second step produces the land management

information needed to run the individual service models, which will be explained in each following service section. The inputs required are a LULC map (used LCM 2007, produced by CEH and obtained from Edina) and Soilsclapes (soil information, obtained from Cranfield University).

3.5.2 Metrics of ecosystem services

3.5.2.1 Carbon storage

Carbon storage was selected as it is important for the HHL landscape through the peat bog habitats and the significant storage they provide. It is an important regulating service, which could be a big concern in the HHL as a change to intensive agriculture tends to mean less carbon can be stored (Jiang et al., 2013). The most common approach for looking at carbon storage is to use land cover maps, especially in terms of soils and vegetation, and estimate the storage for the different biomes, for both above and below ground (Naidoo et al., 2008; Egoh et al., 2008). Most of the carbon is stored in the soil, but this depends on vegetation type, climate, hydrology, topography and the nutrient environment (Ostle et al., 2009). Carbon in vegetation, both above and below ground, accounts for about five per cent of UK carbon, with heath and bog types containing the most, meaning wetlands can often provide a good storage system (Ostle et al., 2009; Kayranli et al., 2009). Also, it has been chosen by the HHL Partnership as part of their Monitoring and Evaluation programme (Defra and Natural England, 2012). By understanding carbon storage further the information can be built into the restoration options later on, in order to provide more storage opportunities in the landscape.

This service was investigated using both InVEST and LUCI. The inputs required for InVEST was a LULC map (LCM 2007: sourced from Edina, produced by CEH) and a spreadsheet of carbon pools. The spreadsheet included information on the amount of carbon stored in above ground biomass, below ground biomass, in soil and in dead organic matter for each land use. Most of the data was acquired from the 4th IPCC report (IPCC, 2007) and the InVEST example, but the bog habitat information came from a Natural England report (Natural England, 2009). Jackson et al. (2013b) discusses how difficult it is to collect accurate carbon data

and that they used estimates from multiple studies or from a similar soil or vegetation type, which is what occurred here. The model produces a map of the amount of carbon in megagrams in each grid cell (Figure 3.35), using the sum of the carbon pools. Due to the difficulty in acquiring accurate values for the carbon pools these values should just be taken as relative. Figure 3.35 shows that the amount of carbon stored varies across the landscape with different land uses. The two NNRs, mainly made up of bog habitats, store large amounts of carbon, but the woodlands also produce high values. The urban and freshwater land use types store the least carbon.

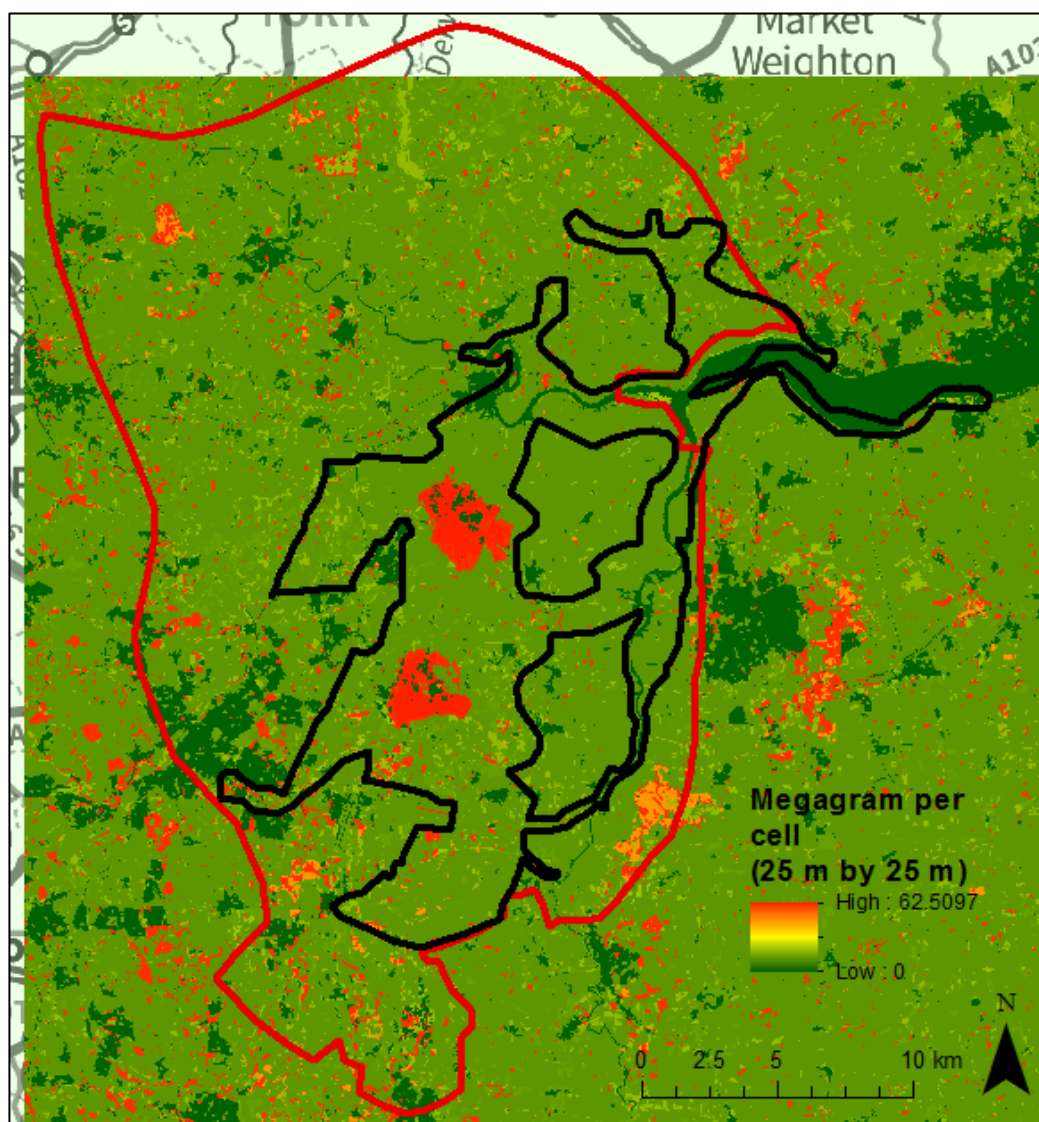


Figure 3.35: InVEST carbon storage results (megagram per cell (25 m by 25 m)).

The LUCI model for carbon stocks requires input pathways to the results of the two pre-processing steps and a series of thresholds for levels of carbon stocks. These values were not further researched as they were not required for the output map and difficult to find, so the default values were left in place. These carbon calculations also take into account above ground biomass, below ground biomass, deadwood, litter and soil carbon, and assume that carbon levels are at a “pseudo-steady state” (Jackson et al., 2013b). The model produces a series of outputs, including the stock of current carbon in the landscape in tonnes per hectare, as seen in Figure 3.36. In Figure 3.36 there are a few areas in white which indicates that there is no data, and these are areas the model has counted as water bodies during the pre-processing steps, believed to be because they are large areas of flat land that the model thinks that water can accumulate in. The majority of these areas are not actually water bodies. For most of the map the results appear to be as expected in relative terms, with the two NNRs producing the highest storage values and the urban areas being amongst the lowest.

In order to compare these results with the InVEST results, the InVEST map was converted to the same units as LUCI, tonnes per hectare, using the raster calculator in ArcGIS, as seen in Figure 3.37. The results are then compared in Figure 3.38. In general, the LUCI results are higher, but it is not systematic. However, for the majority of land uses the two models do show a similar pattern. For wider decision making these results will be useful as they will allow decision makers to understand relatively how different land uses will lead to different amounts of carbon being stored (Ostle et al., 2009), but should not really be used for absolute values. Visually these models are very clear and easy to interpret, and so will be useful, but may require someone with a level of expertise to fully understand what is occurring in the landscape.

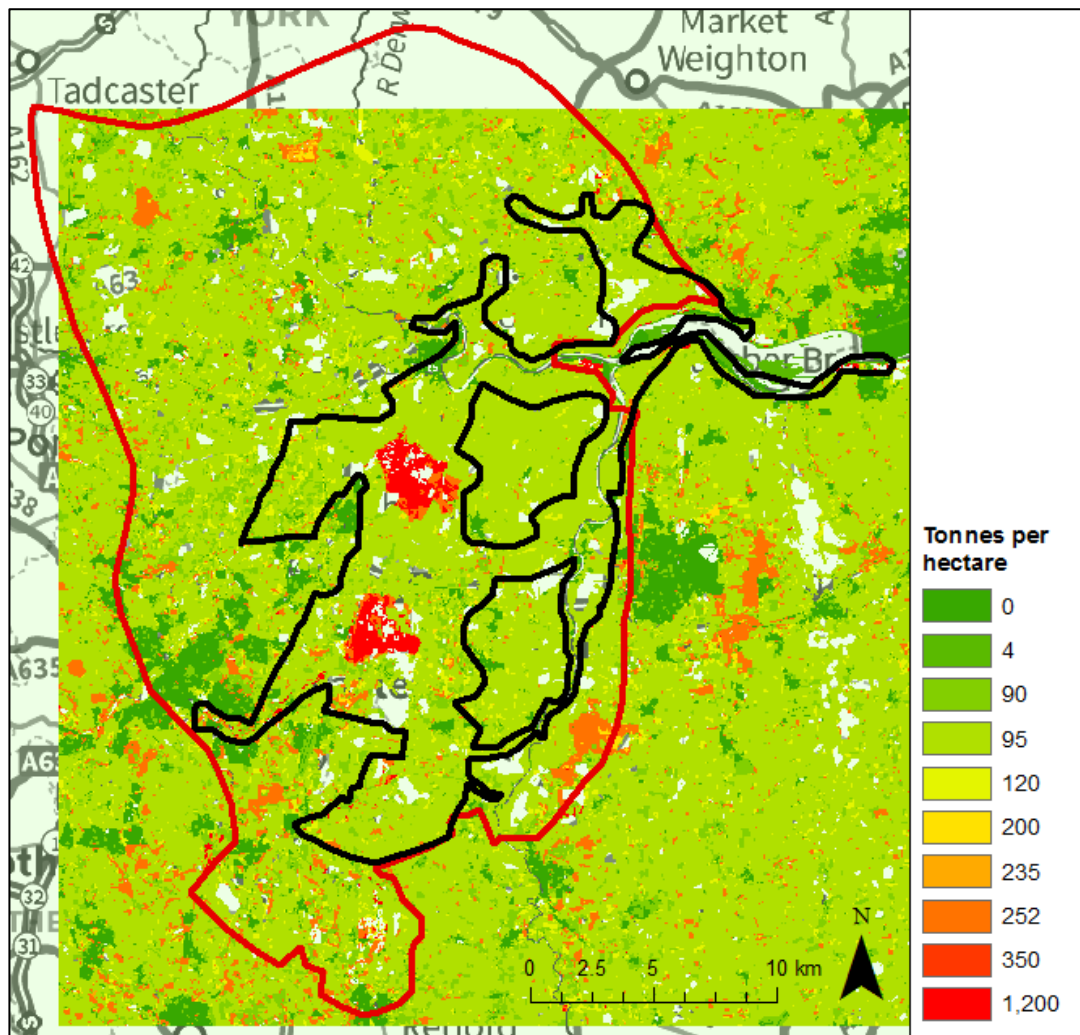


Figure 3.36: LUCI carbon storage results (tonnes per hectare).

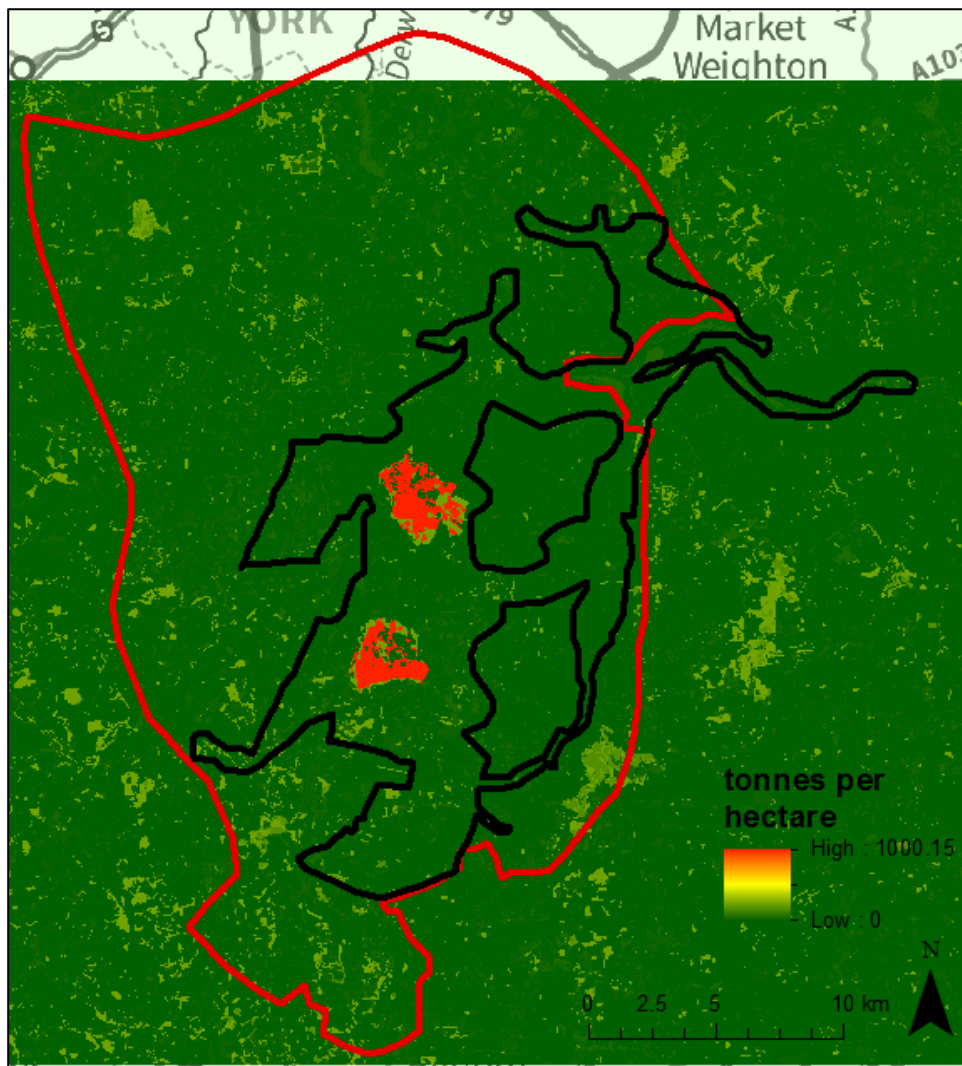


Figure 3.37: InVEST carbon storage results converted to tonnes per hectare (for comparison to LUCI results).

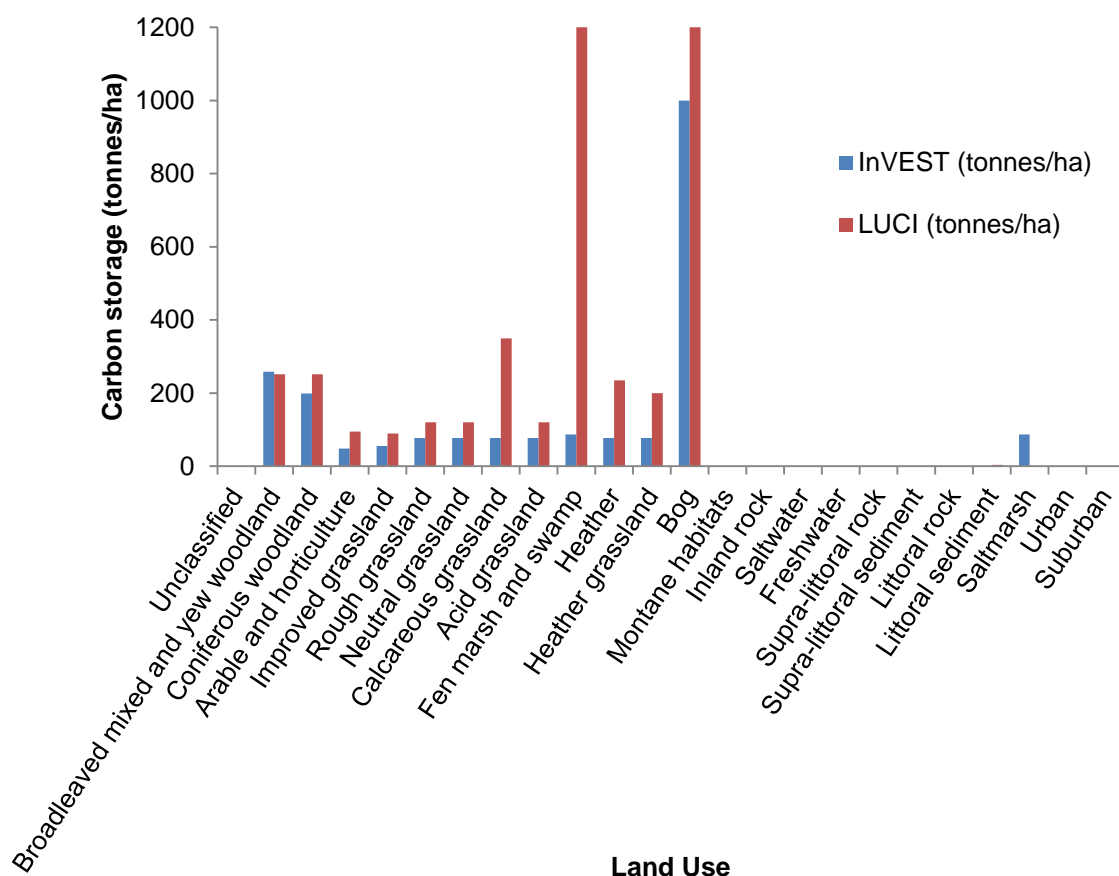


Figure 3.38: Comparison of the InVEST and LUCI carbon storage results (tonnes per hectare).

3.5.2.2 Areas of more sustainable agricultural production

Sustainable agricultural production was selected as it was used by the HHL NIA Monitoring and Evaluation group. As well as providing parallels with the HHL NIA project, it is an important service to consider, as the landscape is farmed so intensively over the greatest area of the HHL. By investigating this service it will help understand which areas may be difficult to restore and which areas are already assisting with habitat creation because of a scheme in being place. Within the ecosystem services framework this is a supporting service that TEEB (2010) defines as a habitat service. As Figure 3.39 shows, there are five different environmental stewardship schemes, with the “Entry Level Scheme” covering large amount of the HHL, but with large gaps within the NIA. The “Entry Level Scheme with Higher Level Stewardship” also covers large patches, with the other three schemes covering much smaller areas. The large coverage of the schemes suggests that some farmers are actually assisting with habitat creation on their land, but it is difficult to understand how much they have done without further investigation. This information will be useful in wider decision making to

understand which areas are more or less likely to be available for restoration, as the information is clear, but further detail is really needed.

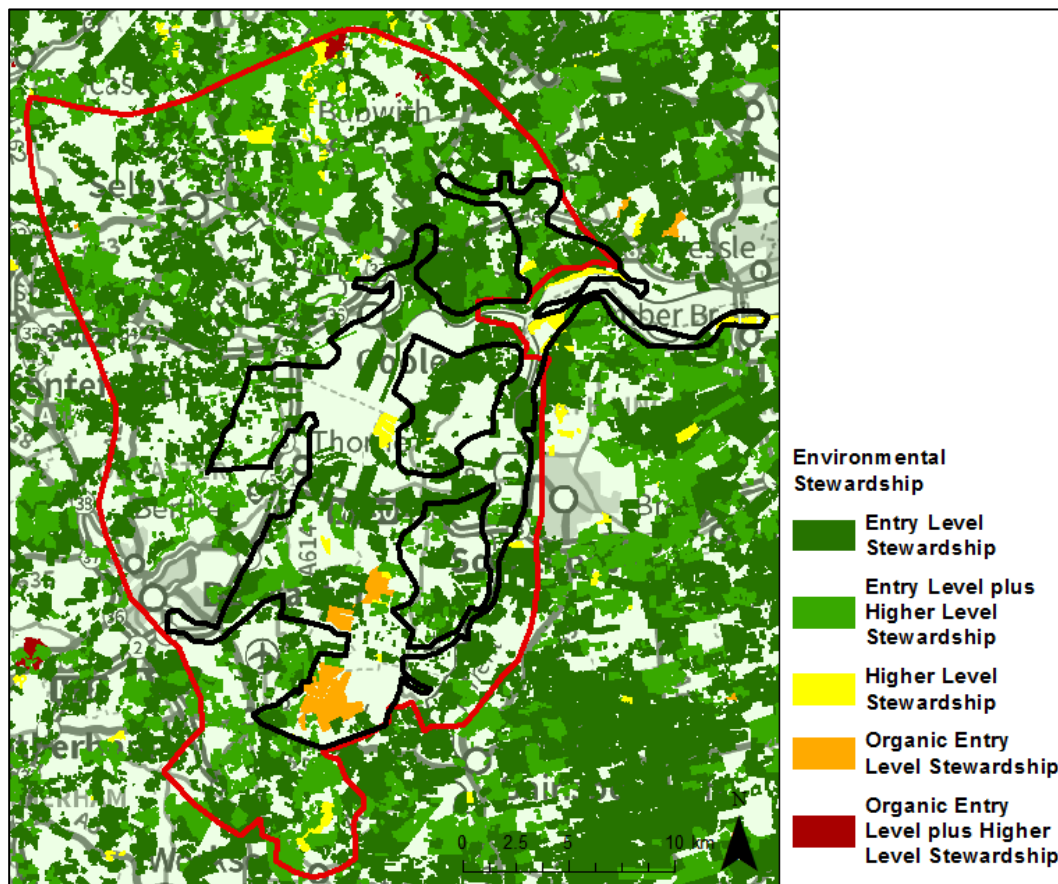


Figure 3.39: Environmental Stewardship map for the HHL (acquired from Natural England).

3.5.2.3 Areas of woodland in active management

Again, this was another service chosen by the HHL NIA Monitoring and Evaluation Group (Defra and Natural England, 2012), and as stated previously by selecting some of the same services as them it may allow for better comparison in later analysis. There are many small patches of woodland across the HHL, and this ecosystem service specifically looks at the areas in active management, so those under sustainable management schemes, that protect the environment as well as providing timber. By investigating woodland in active management it gives an insight into the status of a provisioning service in the HHL landscape and how policies such as the UK Biodiversity Framework and England Biodiversity 2020 can be met, which could be important when developing restoration options. Figure 3.40 displays the GIS files that were acquired from the

Forestry Commission (no date). There are many woodland patches across the landscape, both managed and unmanaged. Within the NIA boundary most of the woodland patches are round the two NNRs and are mostly unmanaged, which could have an influence on the nearby wetland habitats. Again, the map is fairly easy to interpret and so should lead to appropriate decision making.

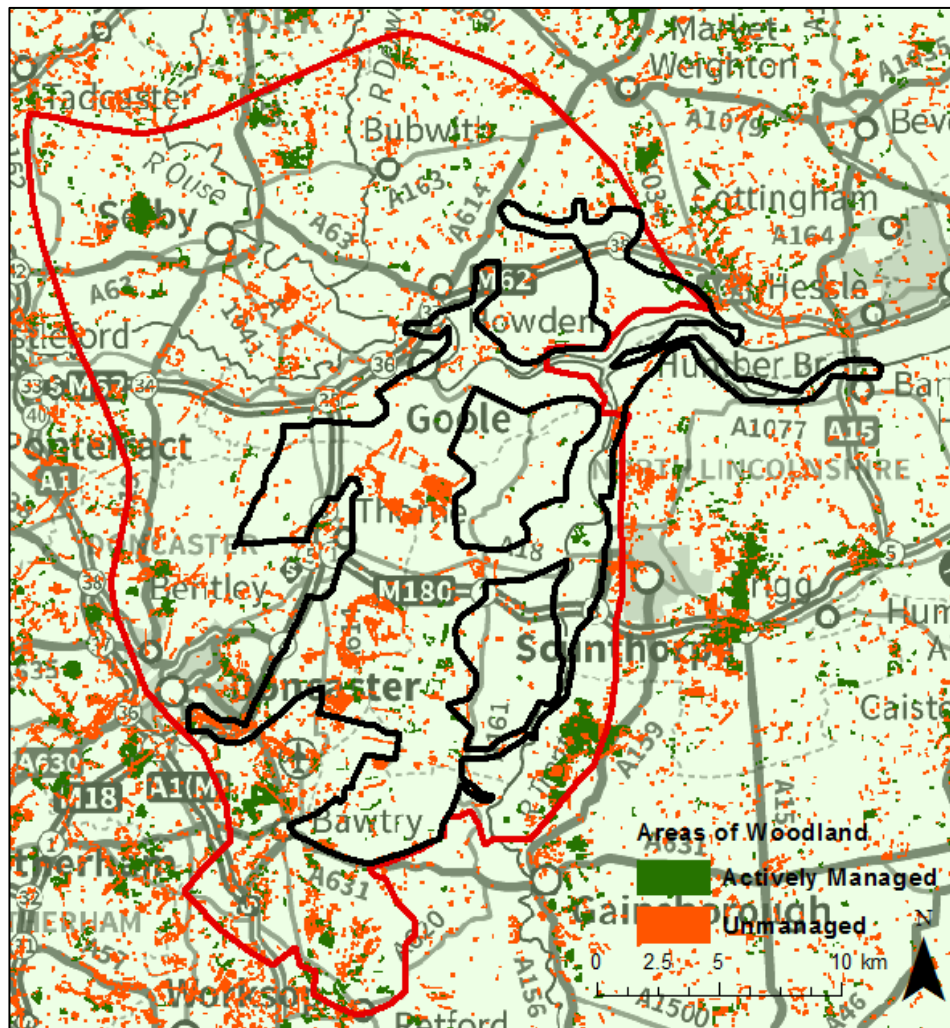


Figure 3.40: Forestry Commission map of actively managed or unmanaged woodland.

3.5.2.4 Public rights of way (PROW) and Cycle rights of way (CROW)

Another service chosen by the HHL NIA Monitoring and Evaluation group under their funding obligations was the cultural service of recreation, measured using the proxies of PROW and CROW, as this service has been acknowledged as difficult to measure (Defra and Natural England, 2012). This service was selected to be complimentary to the work of the HHL Partnership, to include a cultural

service and it is also a very important service for the HHL landscape, especially for decisions around wetland restoration. Not only is restoration about creating habitats for species, but also about providing people with opportunities to experience nature. Figure 3.41 displays PROW sourced from Lincolnshire County Council and CROW sourced from Natural England. From Figure 3.41 it can be seen that there are many public footpaths across the HHL Landscape, but very little within the NIA boundary. Figure 3.41 does not include the paths within the NNRs, as these are not established, but there are still large gaps in the landscape, caused mostly by agricultural land use. There are very few CROW and these are mostly outside the NIA boundary. The distribution of PROW and CROW suggest that currently it is difficult for the public to experience nature within the NIA boundary, which needs to be considered when developing restoration options. The map is useful to understand which areas require footpath expansion as a proxy for recreational ecosystem services, but understanding recreational ecosystem services better is much more complex and requires further investigation.

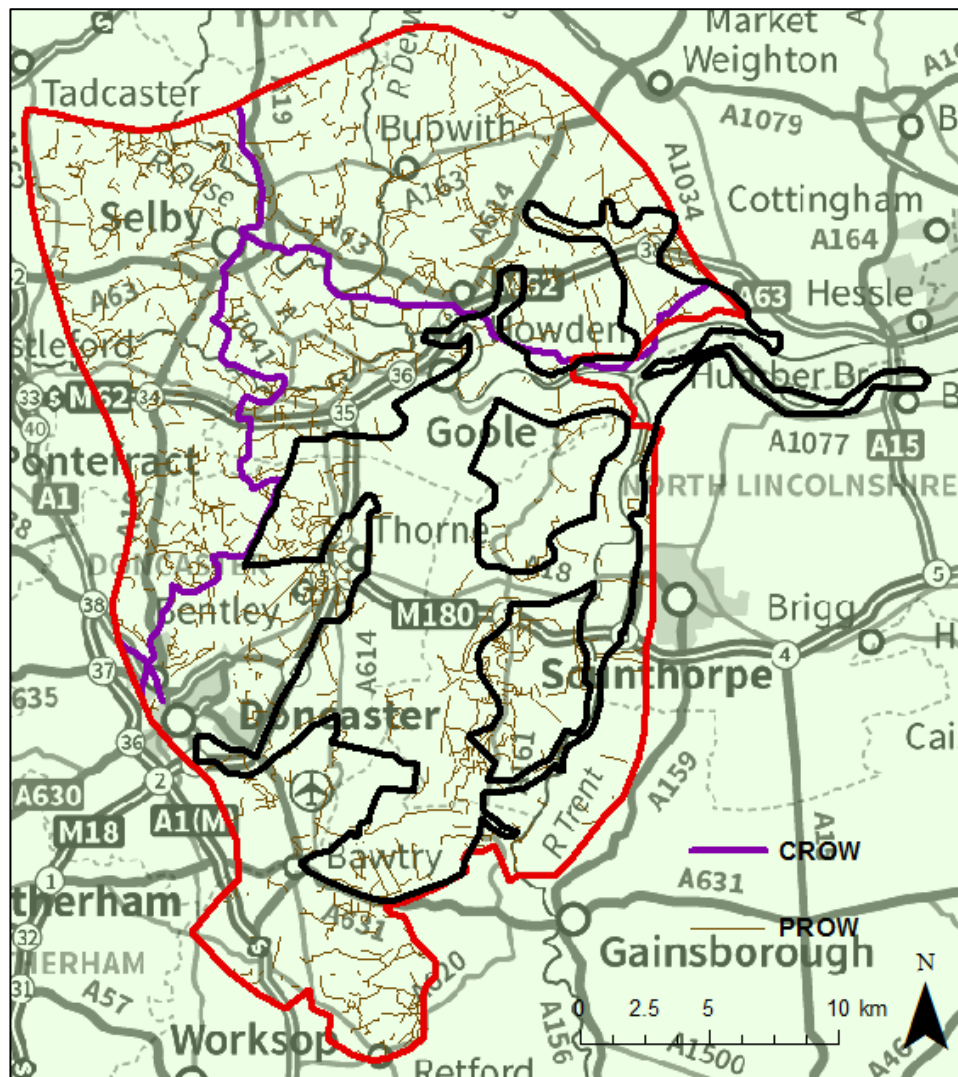
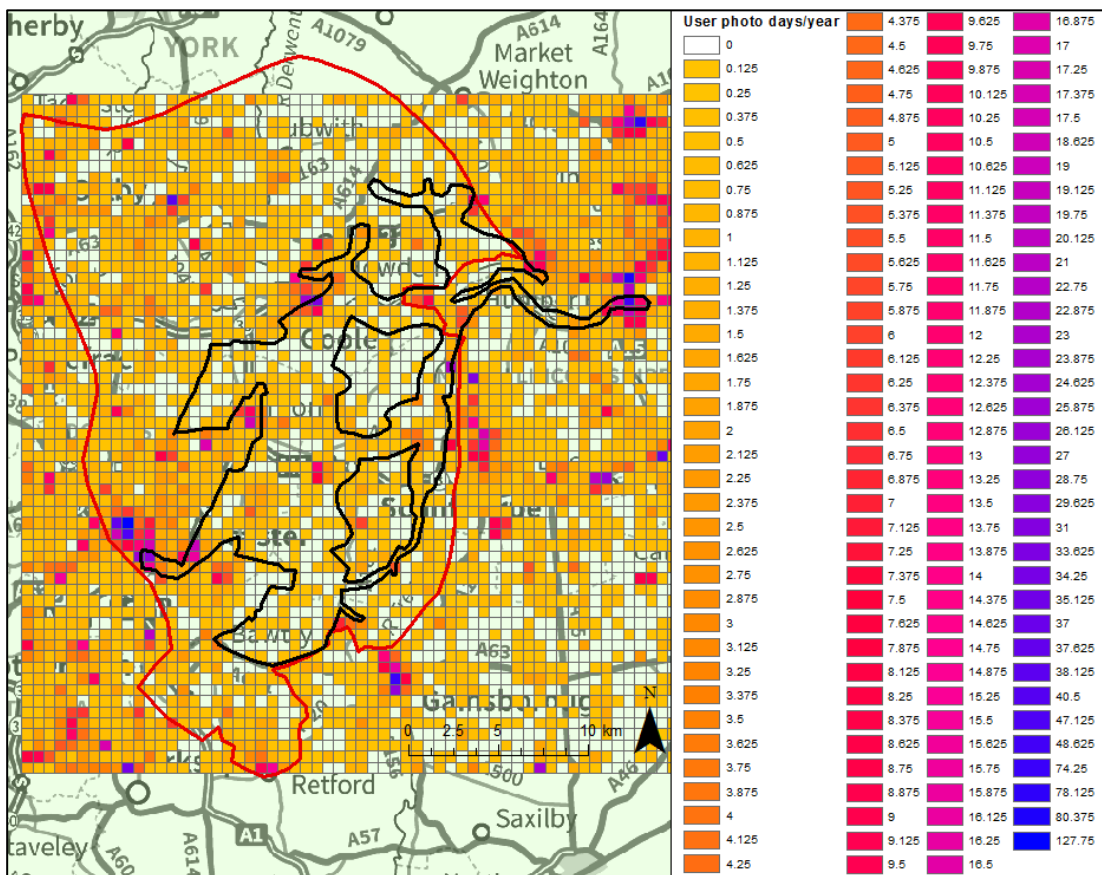


Figure 3.41: PROW and CROW map for the HHL.

3.5.2.5 Recreational use

Within the InVEST model, recreation as an ecosystem service is explicitly used and so was selected here to act as a comparison to the recreational proxies in the previous section and look at another way of measuring this service. The InVEST model accesses photos uploaded onto the website Flickr in order to understand people's usage of a landscape. By using the location stamp on the photo, it is used as a proxy for indicating where people visit in the landscape, which is mapped using algorithms built into the model (Sharp et al., 2014). A range of landscape parameters could be included in the model, but were not, as weightings could not be changed between parameters, and negative weightings could not be enforced. The model could also be run on different scales and so was run at three different ones to see what difference scale made. These are

displayed in Figure 3.42 (1000 m grid), Figure 3.43 (500 m grid) and Figure 3.44 (250 m grid, which was the lowest scale possible). As the scale changes, more detail is seen and a better understanding of this proxy can be understood. Figure 3.42 at the 1000 m grid scale shows that areas around the Humber and urban areas tend to have the highest values. At the 500 m scale (Figure 3.43) as well as the urban areas, there is also some evidence of visitation around the two NNRs, which is further highlighted by the 250 m scale results (Figure 3.44).



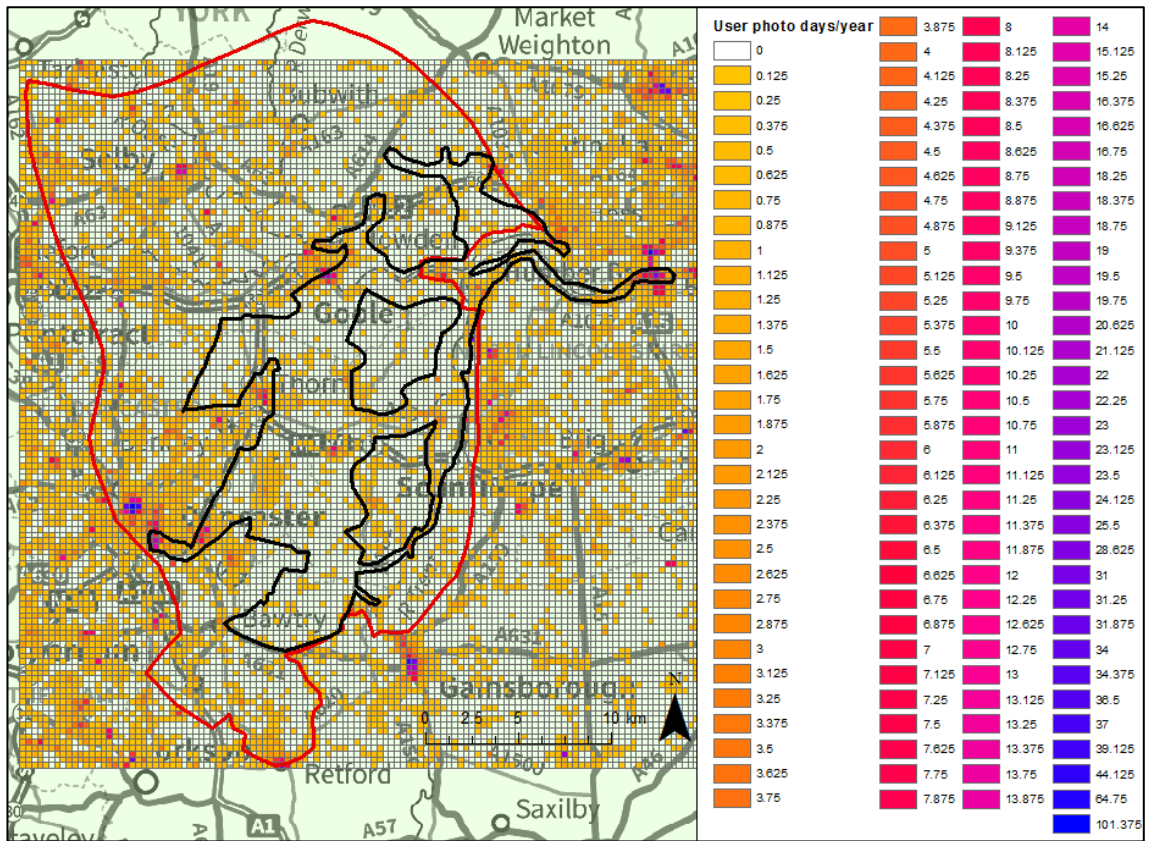


Figure 3.43: InVEST recreation results at the 500 m scale.

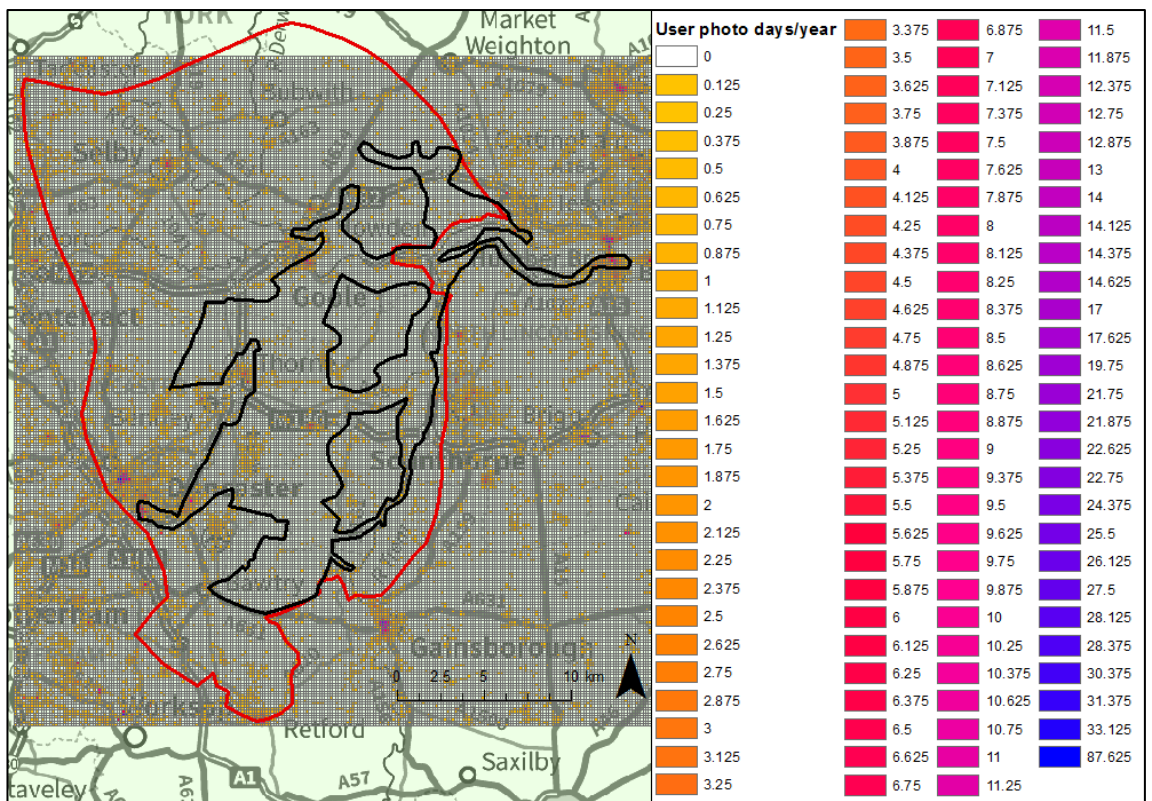


Figure 3.44: InVEST recreation results at the 250 m scale.

In order to fully understand these results it is worth investigating the photos used within the model and classifying them. Figure 3.45 shows the distribution of Flickr photos that were collected from the HHL landscape. These were collected from the Flickr website using the software R and a code developed by Dr. Daniel Richards. Unfortunately, the code only retrieves the most recent photos from the landscape (which were all from early 2015) and the InVEST model uses photos from between 2005 and 2011, but it can still be used to potentially understand patterns. In Figure 3.45 the photos are in six categories (chosen based on the specific context), looking at different activities across the landscape and if these fit into recreational ecosystem services (Richards and Friess, 2015). There appears to be some pattern between the InVEST results and the Flickr photos, which backs up the way the model works, and suggests that activity across the landscape has not changed much in recent years. A large amount of the photos taken come under the vehicles category; there were many photos of trains and lorries, suggesting spotters of these often use the Flickr website. Vehicle spotting can count as recreation, but was not included in this case.

Figure 3.46 looks at just the three categories which were classed as recreation: nature appreciation, landscape and physical activity, which partly fit into the pattern produced by InVEST. Figure 3.47 looks in more detail at the photos taken around the two NNRs, which are supposed to be focal points for recreation in the landscape, but there are very few photos and they come from a variety of different categories. Further investigation with many more Flickr photos going back in time is required, but is not possible using the R code method. It would be useful to try and compare the PROW map and these results in some way, but due to the differences in data format it is not possible. The model results for recreational services will be useful for the restoration options. However, these maps and photos are actually quite difficult to interpret, partly because it is difficult to understand which areas the high patches represent, how these results were produced and how people use the Flickr website.

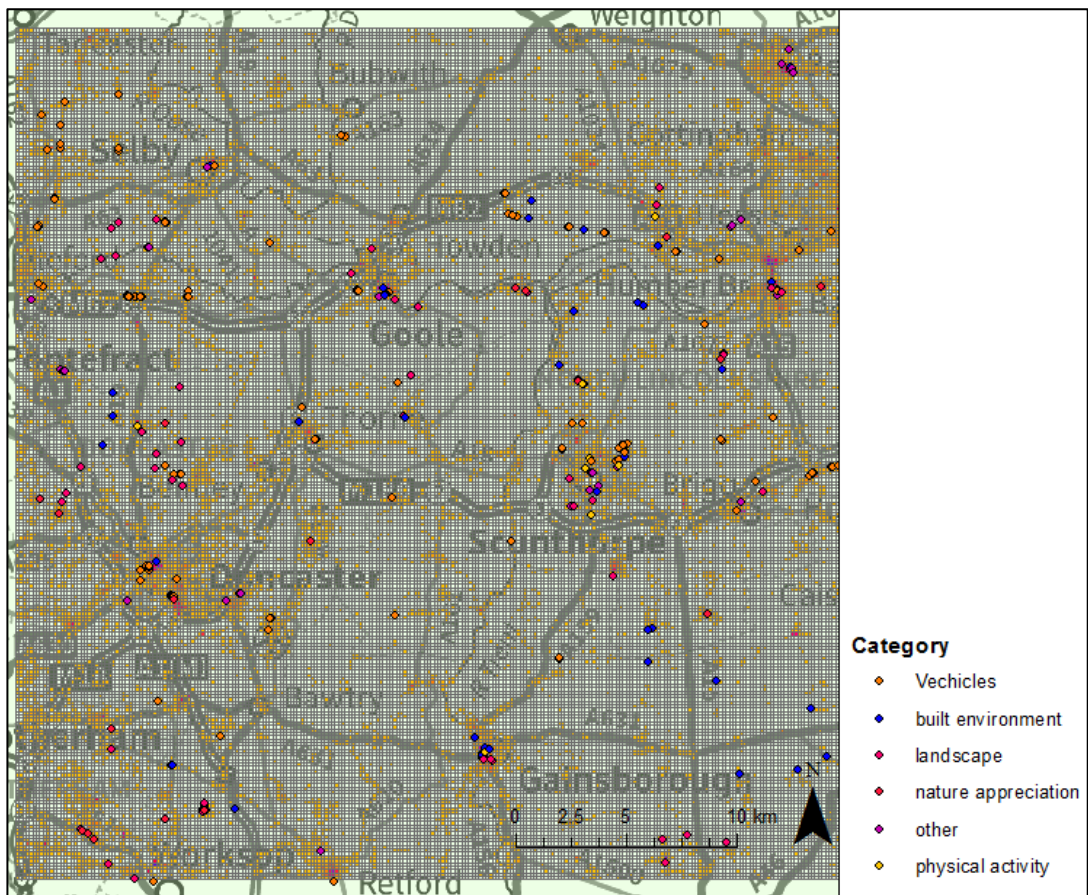


Figure 3.45: Flickr photos over the InVEST recreation results at the 250 m scale.

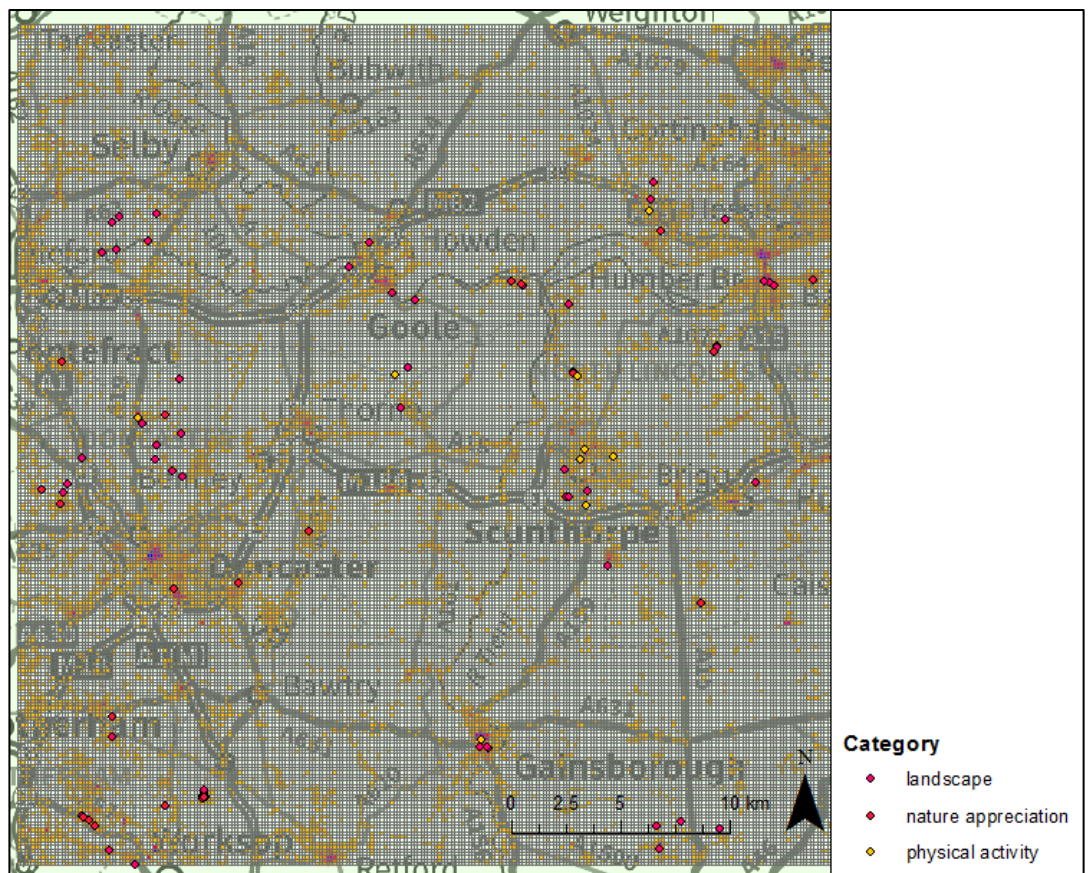


Figure 3.46: Nature appreciation, landscape and recreation Flickr photos with the InVEST recreation results at the 250 m scale.

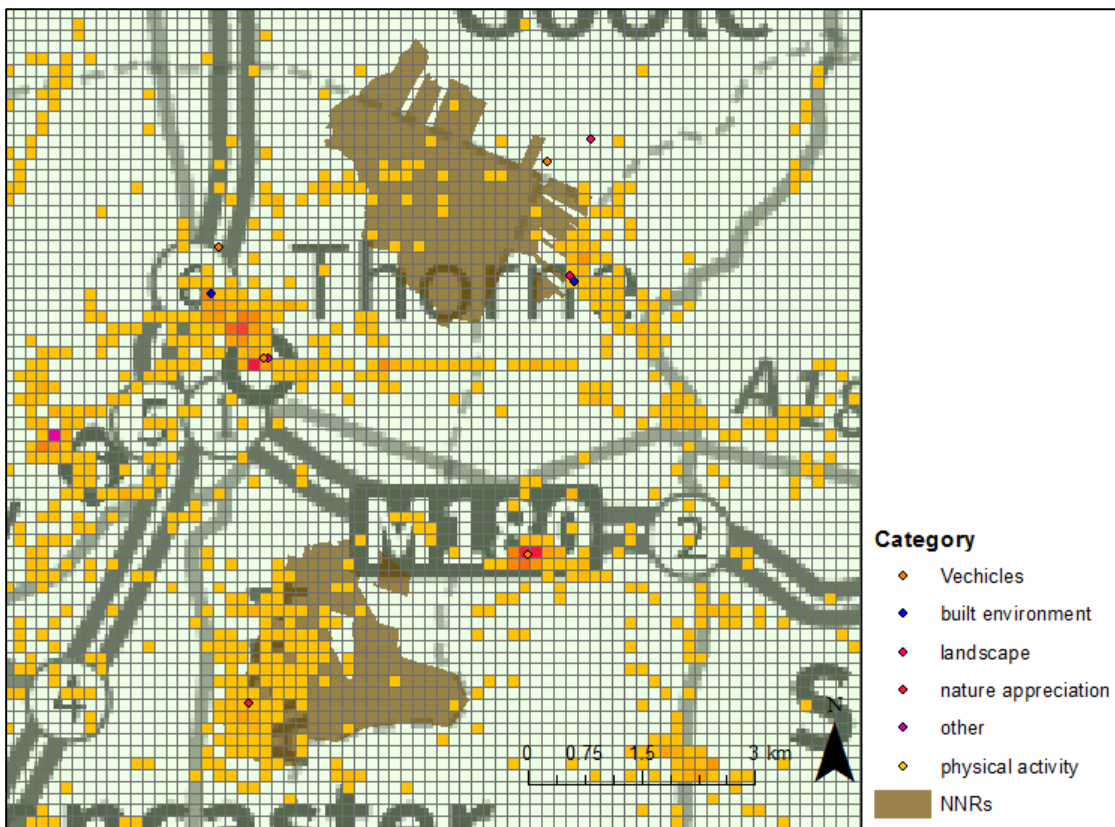


Figure 3.47: Flickr photos around the NNRs with InVEST recreation results at the 250 m scale.

3.5.2.6 Flood storage (Flood mitigation in LUCI)

Flooding is an extremely important ecosystem service to consider in the HHL, especially due to the large areas of low lying land and in the face of climate change. It is also considered in the literature, including the HHL Business Plan (Peacock, 2003; Cook and Hauer, 2007; Maltby, 2010; HHL Partnership, 2011). Under the ecosystem services framework it is a regulating service defined as flood mitigation (as in the LUCI model), although some of the data collected provides more of a supporting service. As well as running the LUCI model, information was gained from the Environment Agency in the form of the maps in Figures 3.48, 3.49 and 3.50. Figure 3.48 shows the flood alert map for the landscape, displaying that much of the area has the potential to flood, due in part to the large areas of low lying land. Figure 3.49 displays the historic flood map for the region, showing that the areas along the rivers have been flooded before, but around the two NNRs have not. Figure 3.50 identifies areas that the Environment Agency believe are potential places for storing water to prevent floods (a similar concept as LUCI flood mitigation), which are not particularly large or found in downstream areas, but could still be very useful in the landscape.

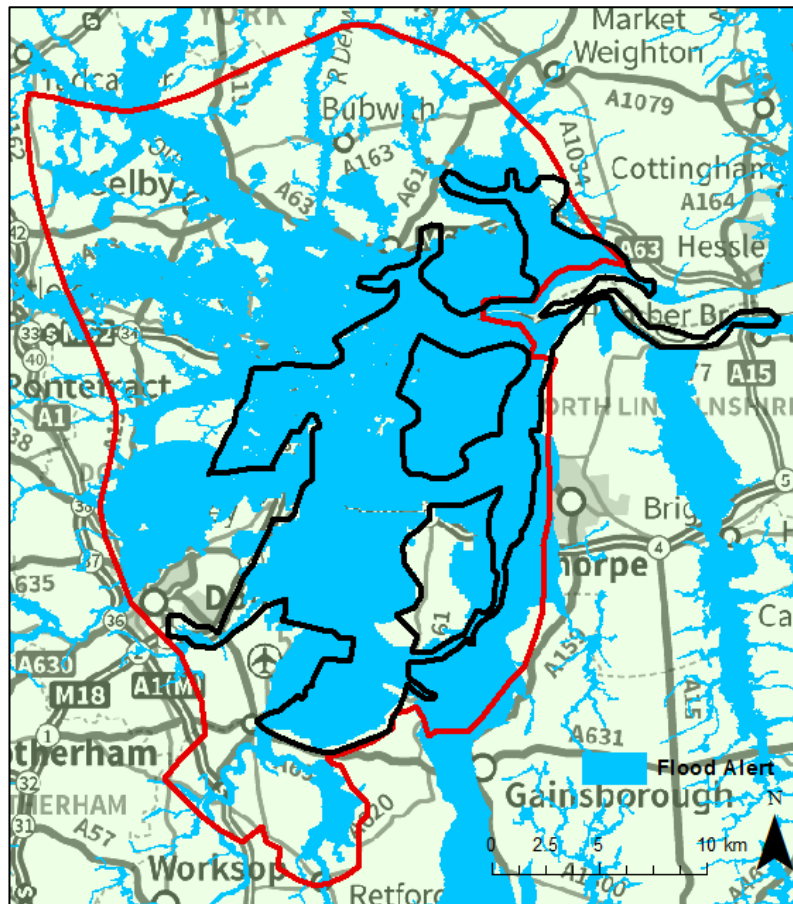


Figure 3.48: Flood alert areas in the HHL (data acquired from the Environment Agency).

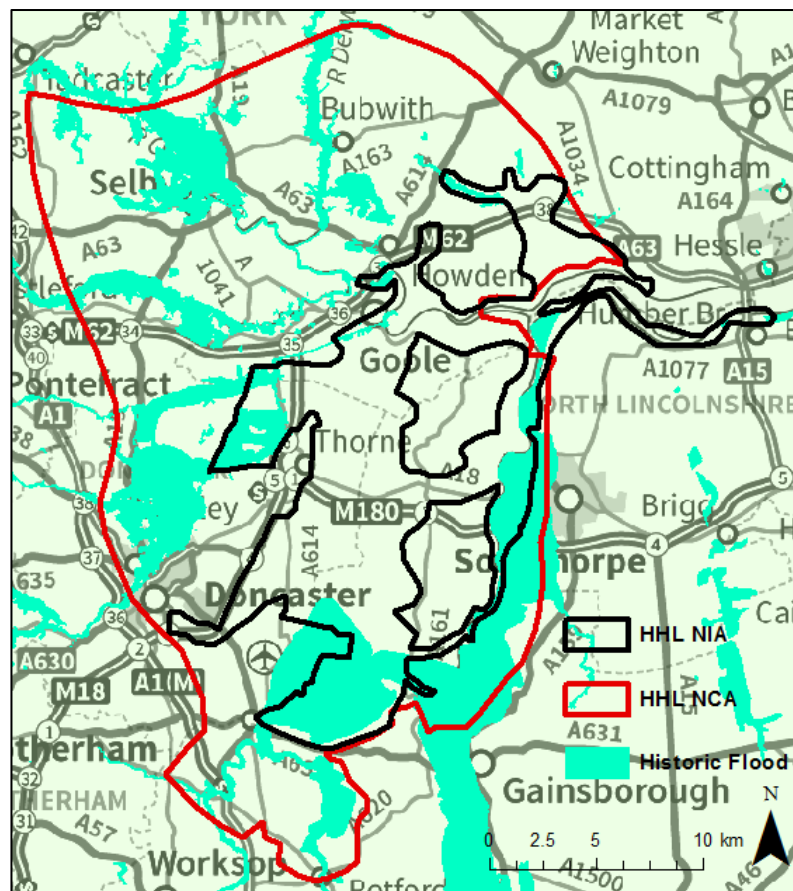


Figure 3.49: Historic flood map of the HHL (data acquired from the Environment Agency).

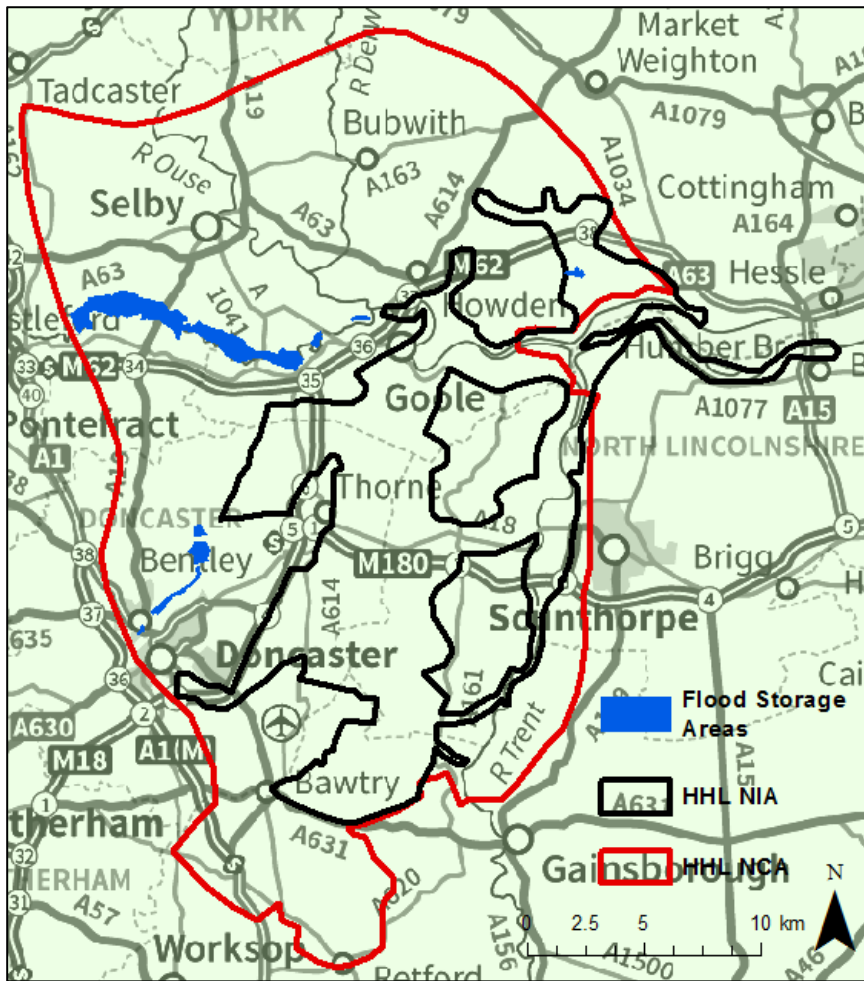


Figure 3.50: Flood storage areas in the HHL (data acquired from the Environment Agency).

The flooding based ecosystem service in LUCI is defined as the regulating service “flood mitigation”. The inputs required for LUCI are: input folders for the two pre-processing steps, a lower threshold value for flood mitigation opportunity and a lower threshold value for very high flood mitigation opportunity (used default values for both as data did not exist for the landscape). The model then produces a series of outputs, including Figure 3.51. The tool works out the areas in the landscape that have the ability to help mitigate against flood waters by either storing the water or directing it slowly through the subsurface, taking into account soil and land use type (Jackson et al., 2013b). In Figure 3.51 there are many large water bodies that do not actually exist, as with the carbon model. It is thought that these are caused by the model not being able to cope with the large areas of very flat land, and perhaps the map is picking up historic remnants of what the landscape may have looked like before much of the historic drainage work. The rest of the map is extremely difficult to interpret, partly as the category names are not clear and partly as the map is so visually complex, and there also

appears to have been no other studies that directly refer to the LUCI output. However, even though there are no values behind these categories they can still be used in relative terms to understand this ecosystem service.

It appears that land use and distance from the river network have an influence, as ditches and streams are visible in the results. The further away from these features the more flooding is seen, suggesting that once these areas become flooded it is harder for the excess water to then flow away. Little comparison can be drawn between Figure 3.51 and the previous flood maps, as they are on different scales and take into account different scales of water networks. The results from LUCI and the previous maps will be useful for restoration options by considering the areas most susceptible to flooding and what that could mean for restoring wetlands. However, the map produced by LUCI is difficult to interpret, due to the names of the categories and the complex patterns produced, and so would be difficult to use within decision making.

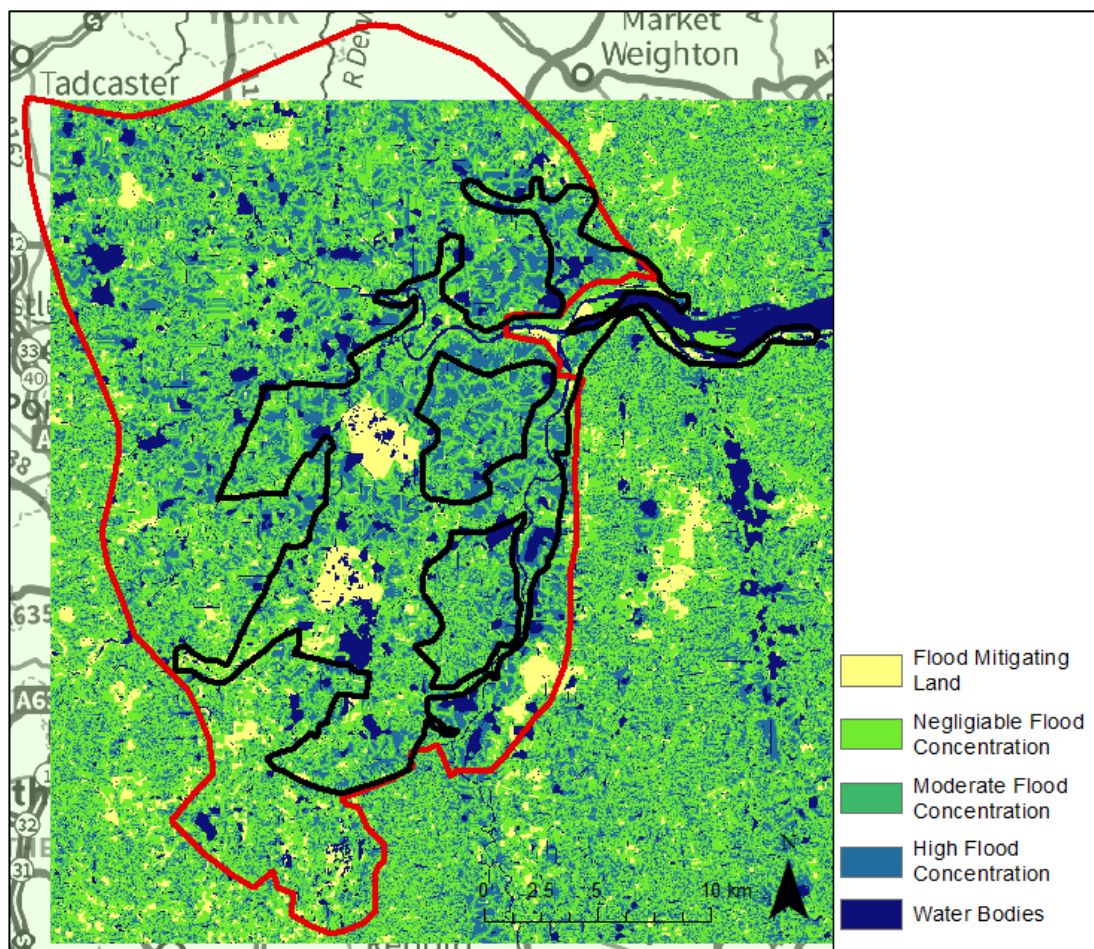


Figure 3.51: LUCI flood mitigation results.

3.5.2.7 Food production (Agriculture in LUCI)

Food production was selected because it is an extremely important provisioning service in the HHL, as it is some of the most intensively farmed land in the country and is extremely fertile. It was also a service that the HHL Partnership had chosen to further investigate (HHL Partnership, 2011). Food production can be investigated using LUCI, but also by looking at Figure 3.52, which shows the agricultural land classes for the HHL (obtained from Natural England). These land classes give a good indication of the best areas for farming, which are probably not the best areas to consider for wetland restoration. Looking at the coverage of “Grade 1” and “Grade 2” (the best grades for agriculture); these broadly cover large parts of the HHL.

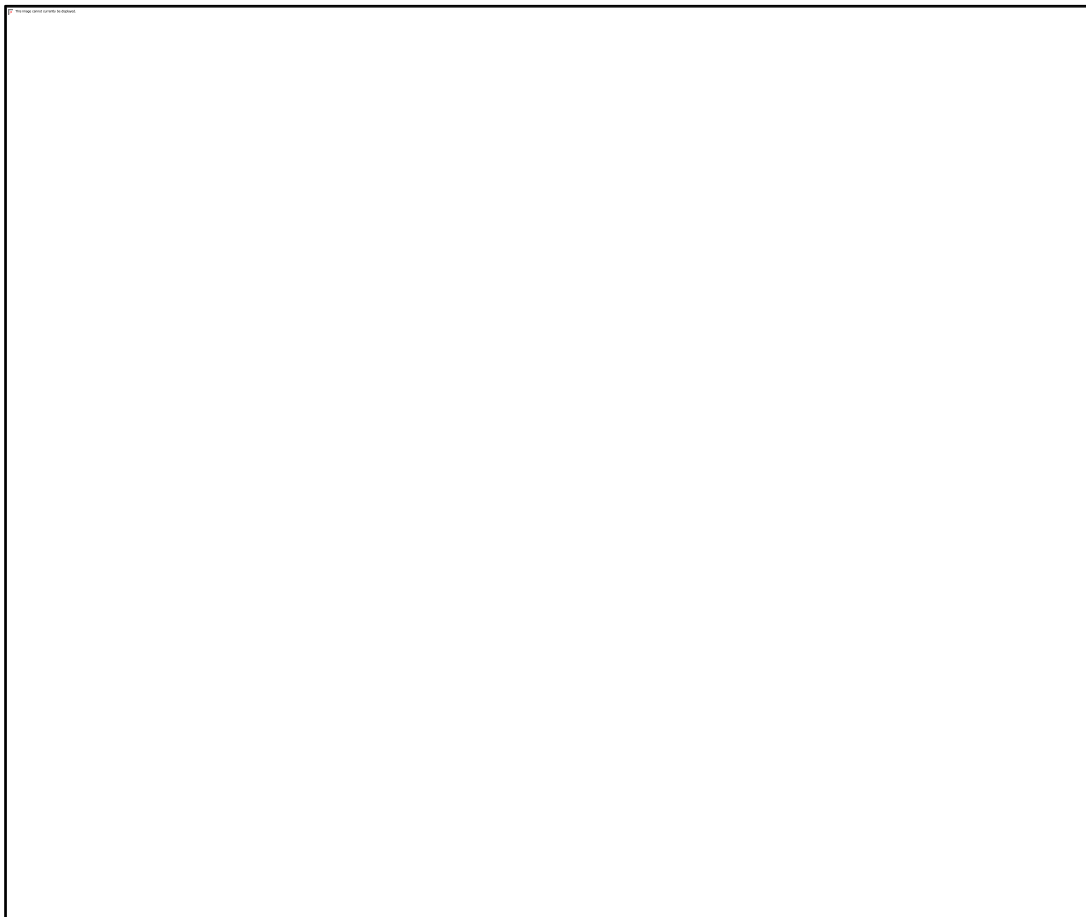


Figure 3.52: Agricultural land classes in the HHL (data sourced from Natural England), where Grade One is the best agricultural and Grade Six is the worst.

With LUCI, it is perceived agricultural value that is calculated. The inputs required are: input folders from the two pre-processing steps, slope threshold value (in degrees) for very productive land, slope threshold value (in degrees) for somewhat productive land, elevation threshold value (metres above sea level) for improved agriculture and fertility relative to national standard. The fertility value for the HHL was given as the highest (classed as standard fertility) due to the acknowledged high fertility of the land, and the other values were kept as the default as these were not known for the HHL. Therefore, the model takes into account slope, aspect, soil drainage and soil fertility (Jackson et al., 2013b). The model then produces a series of outputs, including Figure 3.53, which shows the different levels of productivity across the HHL, although it is not clear what the definitions of each category are.

This model also produces the large phantom water bodies that are not actually in the landscape, and so will have to be considered when interpreting the results. In relative terms the rest of the categories could be useful. Table 3.1 shows which land uses fit into each category and these are mostly as expected, with “arable and horticulture” being of high productivity, and for example, “saltmarsh” being of low productivity. When comparing Figure 3.53 to Figure 3.53 there is little pattern between the two because the LUCI results are governed by land use outlines, but there is some general agreement in the locations of the highest productivity areas compared to the highest grades, and the same with the lowest areas. The pre-processing steps in LUCI take into account a range of data, but it appears that these results are mostly governed by land use. This ecosystem service will be extremely important to consider when developing restoration options, because it will give a good indication of areas that will be very difficult to restore in terms of land availability and work required. In terms of wider decision making the map is fairly easy to interpret for understanding food production in the area, however care will have to be taken with the large phantom water bodies, suggesting therefore the map should only be used by someone who has an understanding of the landscape.

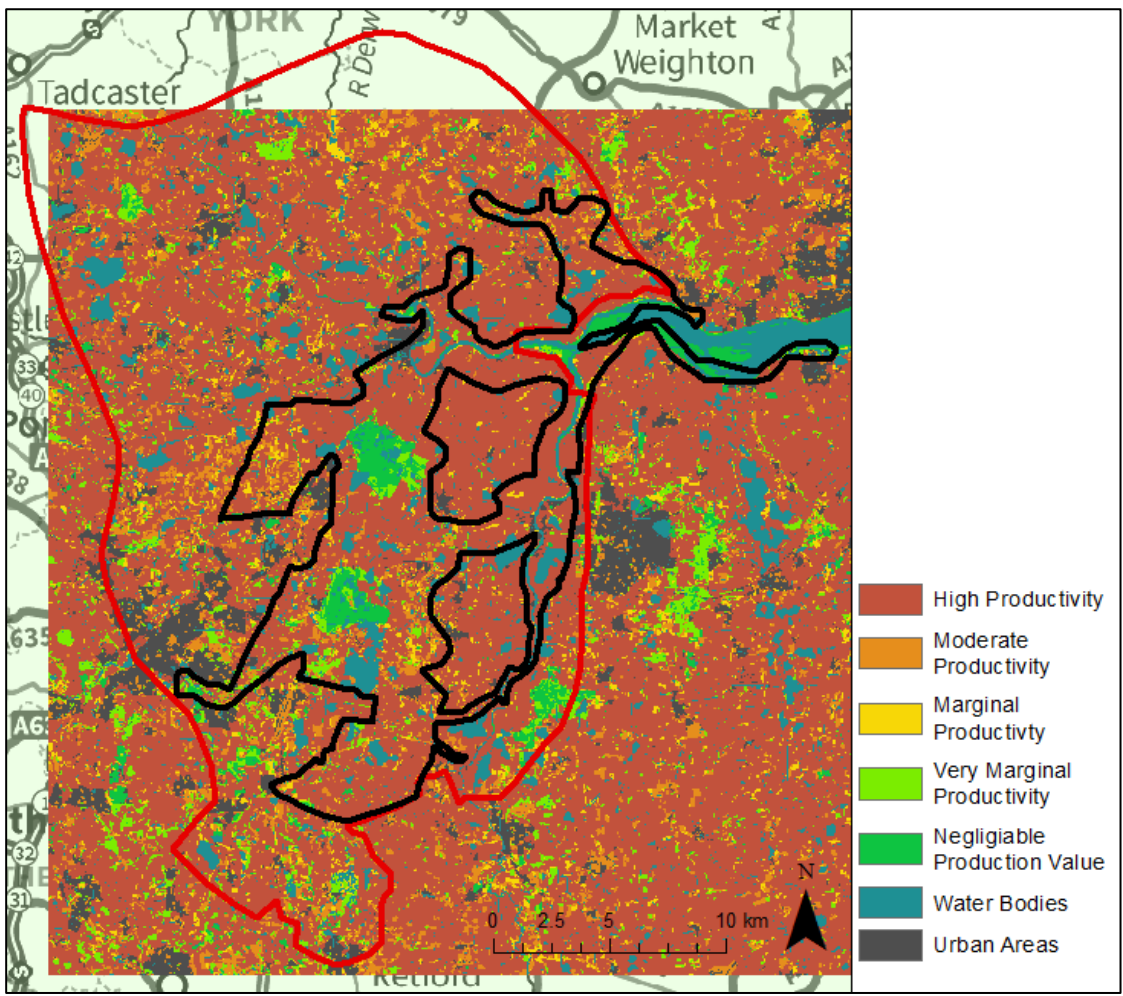


Figure 3.53: LUCI perceived agricultural productivity results.

Table 3.1: Land use types in each LUCI agricultural productivity category.

Value	Category	Land Use
1	High Productivity	Arable and horticulture
2	Moderate Productivity	Improved grassland
3	Marginal Productivity	Rough grassland
		Neutral grassland
		Calcareous grassland
		Acid grassland
4	Very Marginal Productivity	Broadleaved, mixed and yew woodland
		Fen, marsh and swamp
		Heather
		Heather grassland
5	Negligible Production Value	Coniferous woodland
		Bog
		Inland rock
		Supra-littoral rock
		Supra-littoral sediment
		Littoral sediment
		Saltmarsh

3.5.2.8 Habitat quality/suitability

Habitat quality was selected as it is an important aim for the HHL to provide good quality habitat (a supporting service) for increasing biodiversity. It is also a service covered by both models, which could provide an interesting comparison, with InVEST looking at habitat quality and LUCI at habitat suitability. For InVEST there were four inputs required; the CEH LULC 2007 map (acquired from Edina), threats to the landscape (GIS files of urban areas, agricultural areas and major roads), threat information (maximum distance of effect (comes from InVEST examples) and weightings (all set equally, as unknown)), and the relative sensitivity of each land cover to each threat (all set to 0.5, as unknown). The main output of habitat quality is shown in Figure 3.54, but the score is unitless and should be taken as relative. The influences of the urban and road threats are obvious, as these create paler patches around the threats. The areas with the highest scores are those furthest from the roads and urban areas, on particular land cover types. It is hard to see the influence of the agricultural threat because it covers such a large proportion of the landscape.

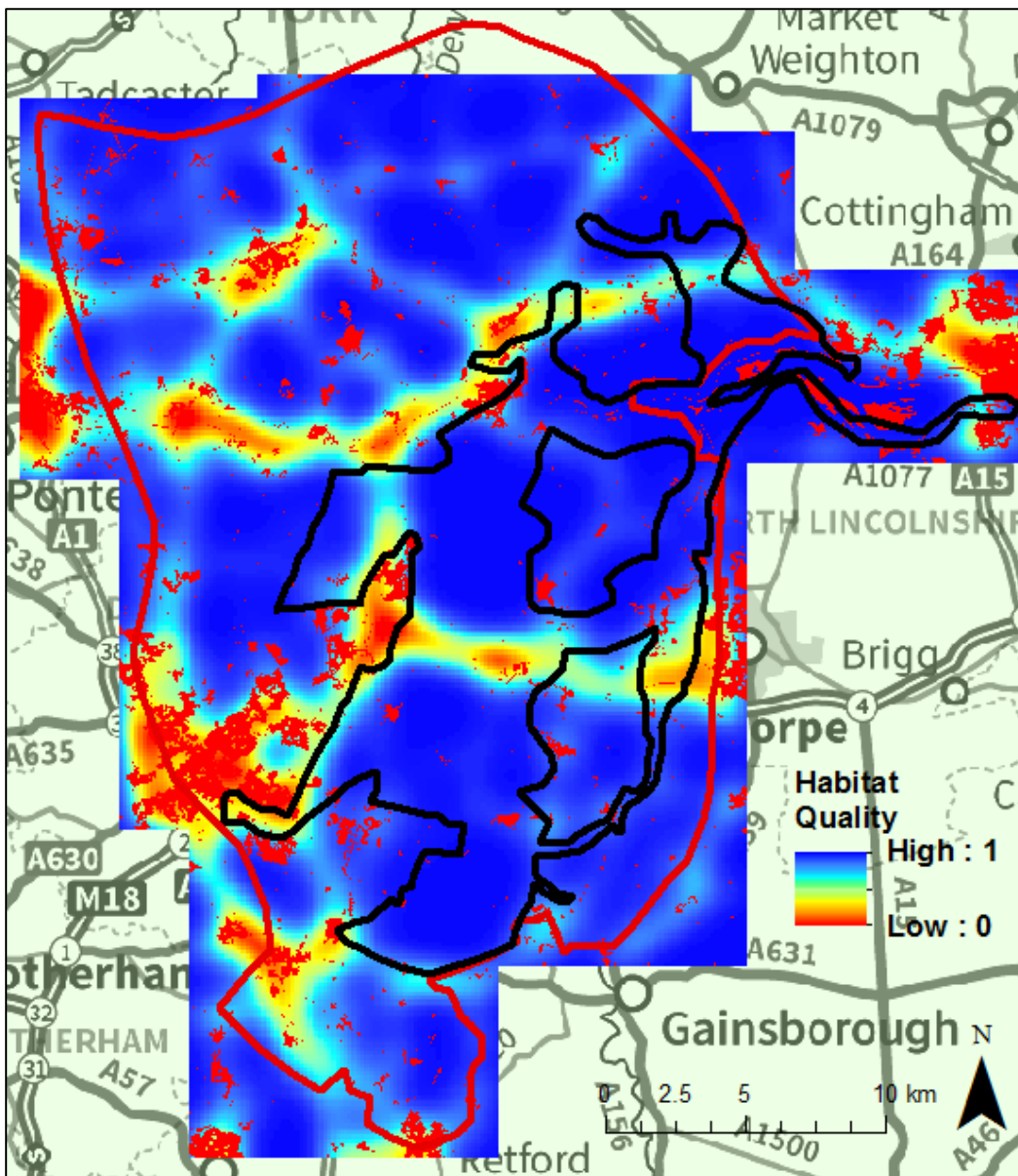


Figure 3.54: InVEST habitat quality results.

The inputs required for the LUCI model are: input folders from the two pre-processing steps and the habitat of interest (wetlands in this case). The model then produces a map of areas suitable for habitat or habitat extension (Figure 3.55). Unlike some of the other LUCI models, it does not produce any large phantom water bodies. The results are difficult to interpret, with the categories not clearly defined. The two NNRs and areas around the rivers are where most of the wetland habitats are found. “Other priority habitats”, are important non-wetland habitats, but should not be included in restoration planning as they are important areas already. The area that is of interest for wetland restoration is “Opportunity to expand existing habitats”, which mostly buffers aquatic environments. Overall, the results confirm the importance of the NNRs in the HHL

and show some potential for smaller scale habitat restoration in the wider landscape. These results will be useful for developing restoration options as it gives a good understanding of what areas are currently in good condition, which could be core parts of ecological networks to expand out from. As with the other ecosystem services, the model seems to be mostly defined by land use.

It was hoped that these two models would be comparable, but due to the different types of output it is not the case. In terms of wider decision making, both are difficult to interpret, which may lead to inappropriate decisions or then being left out of the process altogether. The InVEST model visually is very complicated, and it is difficult to see what influences each threat is having. For the LUCI model the individual categories and the meaning behind them are unclear, and so both models would cause issues within decision making, and require further work or guidance before they could be used.

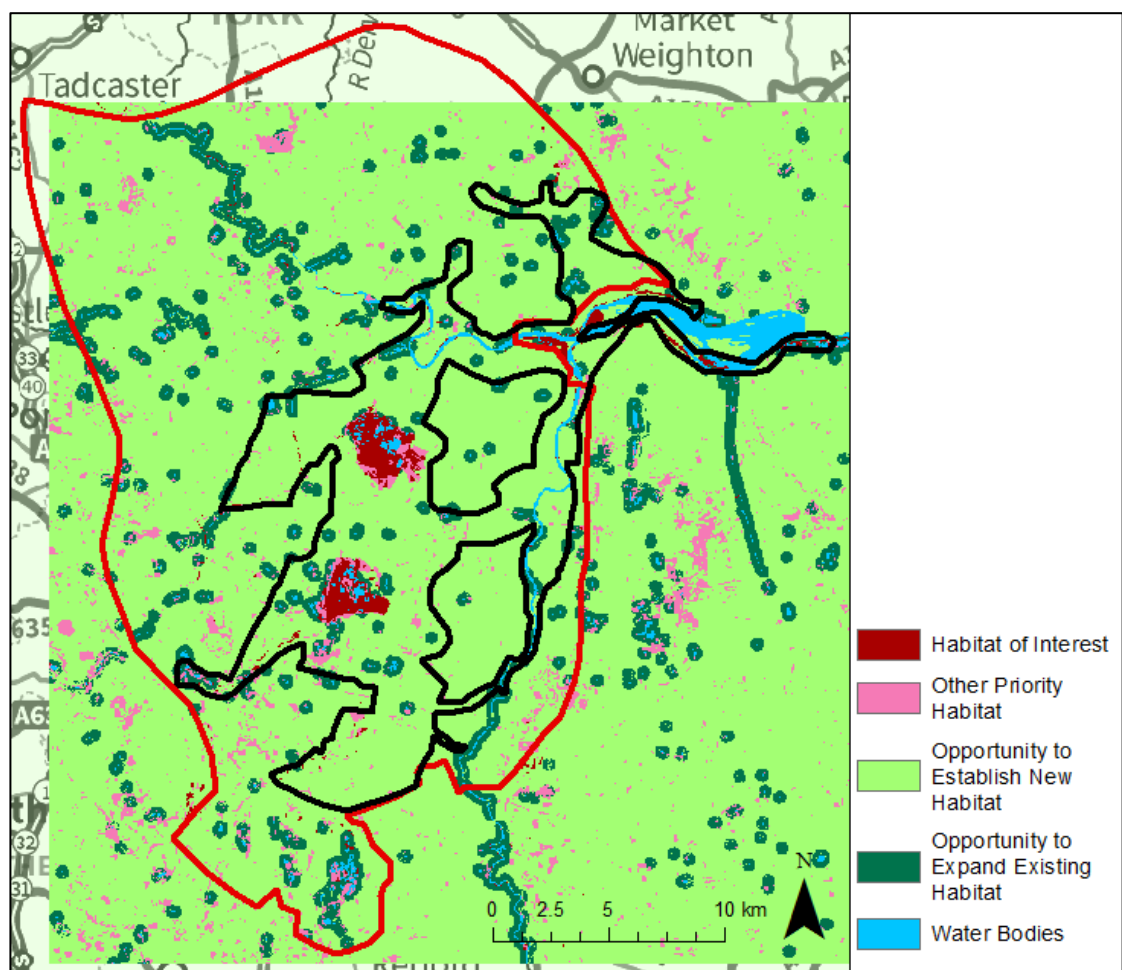


Figure 3.55: LUCI habitat suitability results.

3.5.2.9 Pollination

Pollination was selected as it is an important consideration for the HHL landscape where the aim is to simultaneously increase plant biodiversity and the amount of crop growing that occurs in the landscape, and it was also included in the InVEST package. By further understanding this regulating service it will help with the development of restoration options. The inputs required for the InVEST model are the LULC map, a table of pollinator species and a table of land cover attributes. The table of pollinator species contained information on the name of species present in the landscape, their nesting type (cavity or ground or both), activity by floral season (spring, summer), and average distance species travels to forage flowers. Data was sourced from NBN Gateway, the InVEST model example and other literature, but was very difficult to find. The table of land cover attributes contains information on the relative availability of each nesting type and the relative abundance of flowers in each floral season for each land cover type. Data was acquired from the InVEST example, as the handbook recommends survey data (which does not exist) or expert opinion (which is not accessible) (Sharp et al., 2014). Therefore, the results produced from the model will need to be handled carefully and it indicates that this model is not easy to use or acquire data for, as was the case with the other InVEST models.

Several outputs were produced, but the one of interest is displayed in Figure 3.56 and shows “an index of the likely abundance of pollinator species resting on each cell in the landscape, given the availability of nesting sites and flower resources nearby” (Sharp et al., 2014). These results are also displayed in Figure 3.57, which shows the range of values for each individual land use type. The highest occurrences are mostly in woodland areas, with agriculture mid-range, and urban areas and the NNRs having the lowest values. It is not surprising that the NNRs have lower values, they will not be the best nectar sources as they are mostly cotton grass and sphagnum. It is important to think about whether the urban areas are really that low, due to urban parks and gardens, but it is difficult to further investigate without better data. If there was better data then these results could be useful when developing restoration options, but in its current form it is not. In wider decision making the map is visually clear, but collecting the data needed

and what the values stand for is more complex, and so would require someone with expertise on pollination in order to produce and interpret the information.

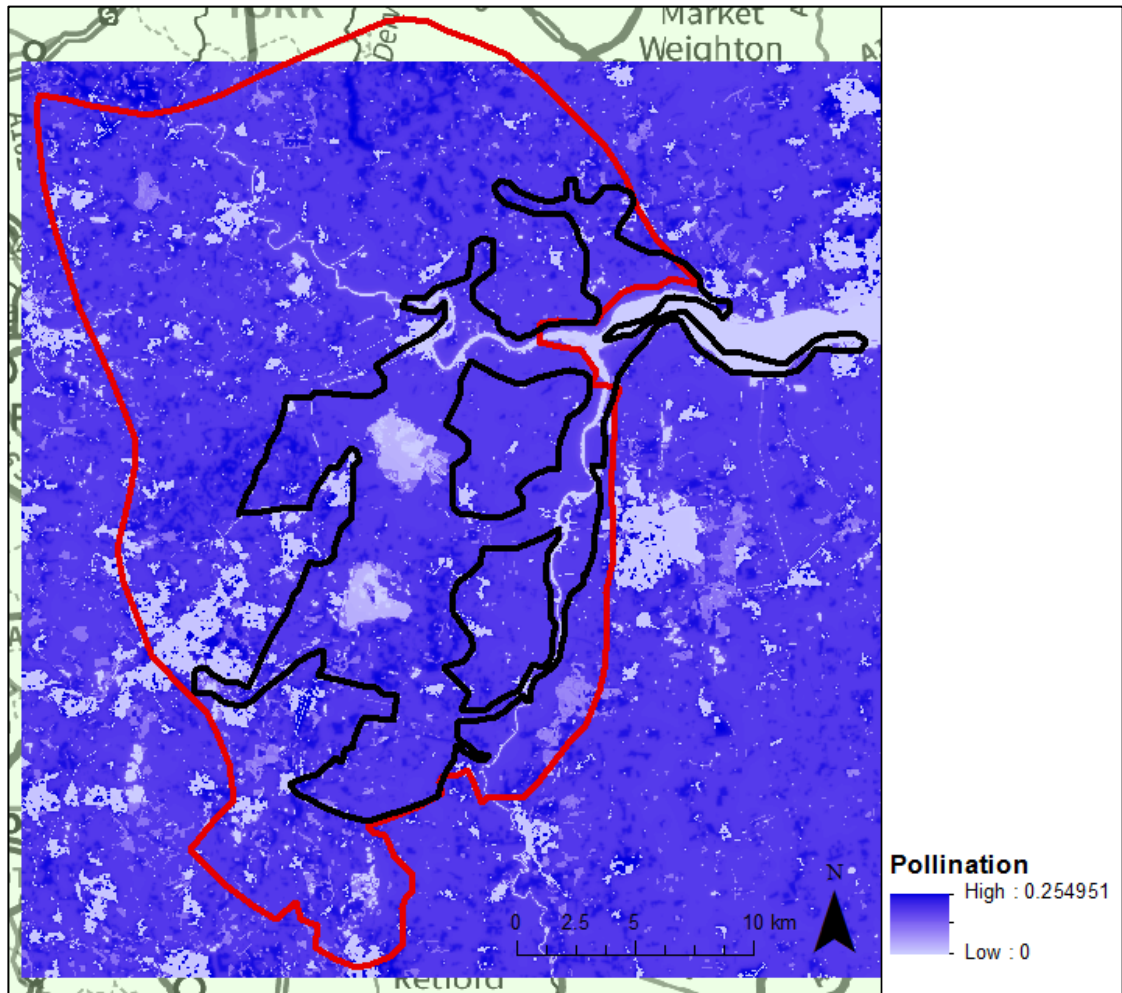


Figure 3.56: InVEST pollination results (values to be taken as relative).

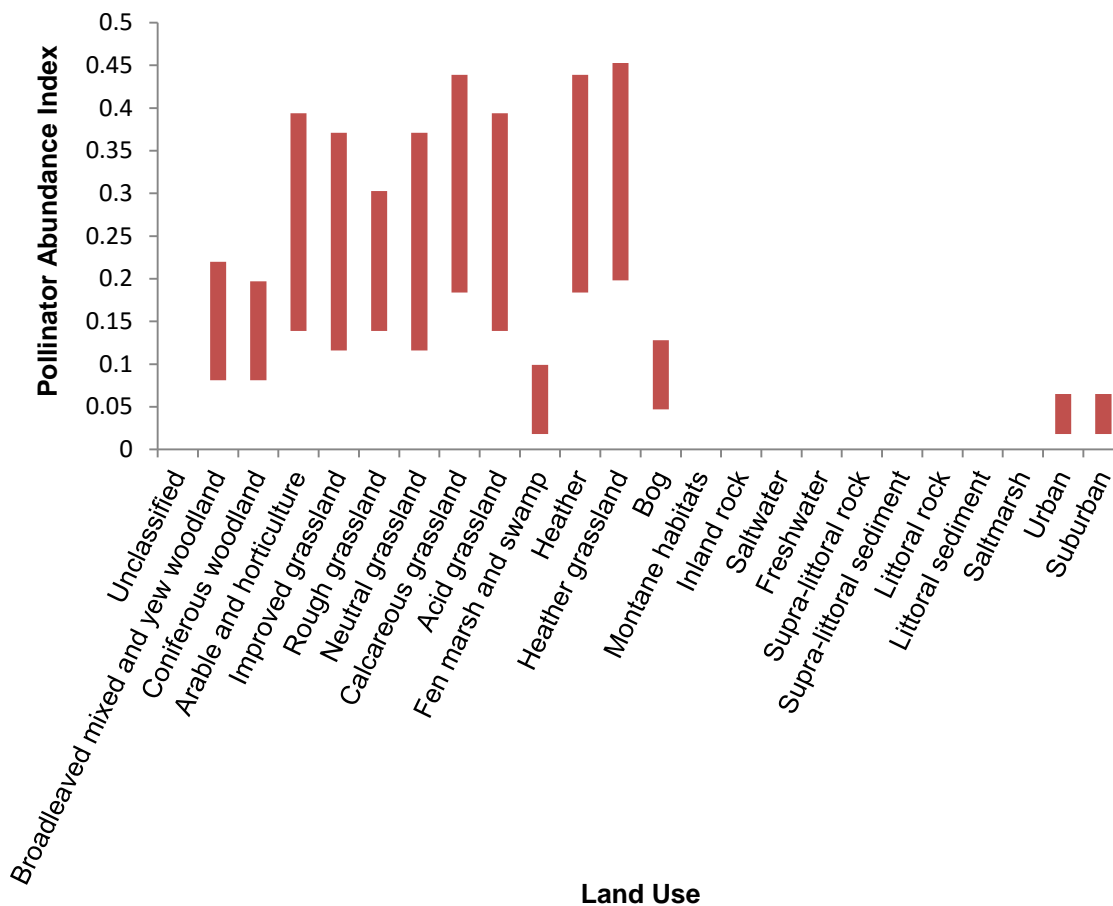


Figure 3.57: InVEST pollination results for each land use type (values to be taken as relative).

3.5.2.10 Habitat connectivity

Habitat connectivity was chosen because it is very important supporting service for ecological networks and therefore for the restoration options, and is discussed in the literature (e.g. Peacock, 2003; Chisholm et al., 2010; Lawton et al., 2010; Mortelli et al., 2012). The inputs required for LUCI are: input folders from the two pre-processing steps results, species of interest (marshy grassland species was selected), minimum area for focal network (ha) and maximum cost distance through hostile terrain (km). Marshy grassland species were chosen as it was thought to be the most appropriate out of the limited number of options available, although what species made up this group was not made clear, which is why the other values were just left at default. LUCI produces a map of habitat connectivity, seen in Figure 3.58. The algorithm takes a cost-distance approach looking at the distances between habitat patches and the species' ability to move over certain distances (Jackson et al., 2013b).

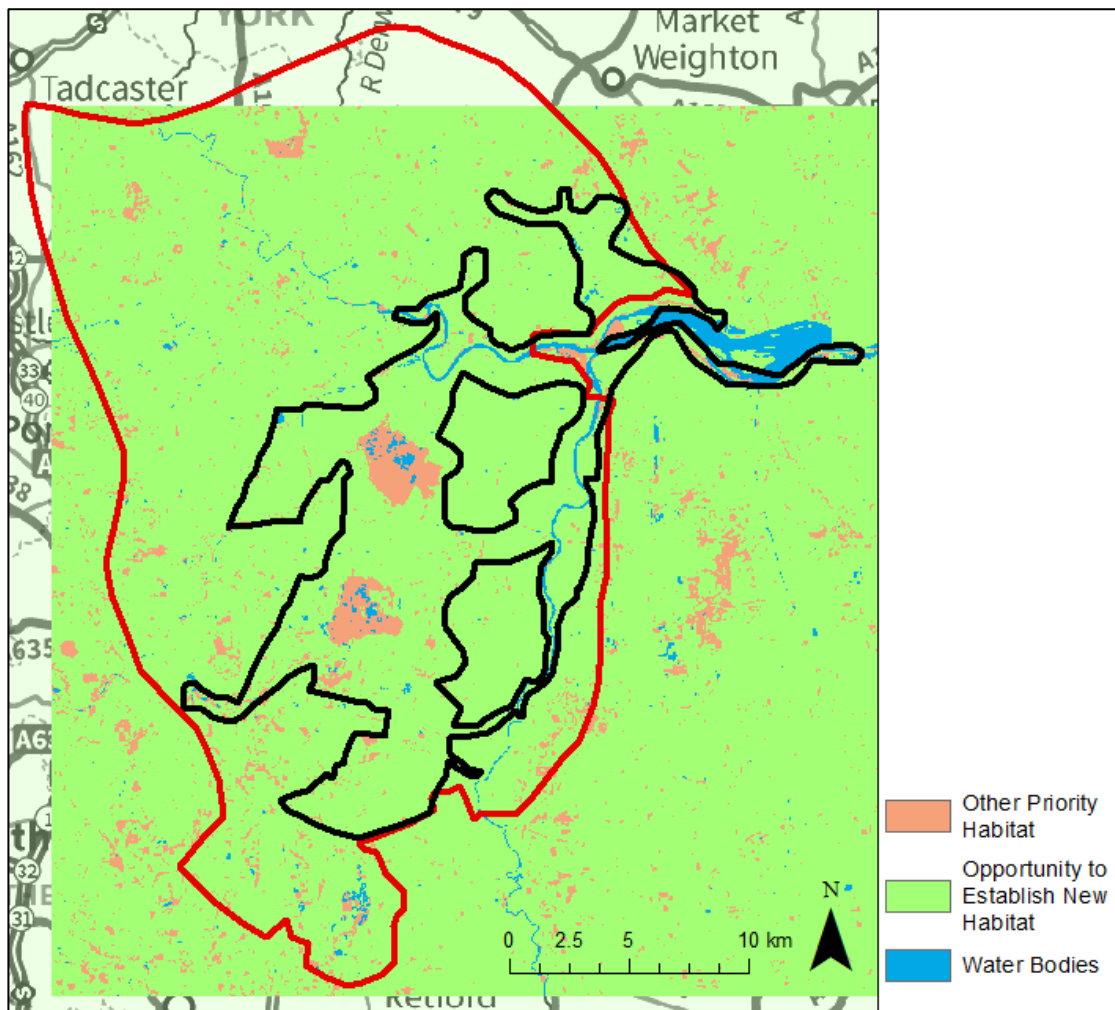


Figure 3.58: LUCI habitat connectivity results.

Firstly, as with habitat suitability, the model does not produce the large phantom water bodies seen on other models. Also, as with habitat suitability the categories are difficult to interpret. Figure 3.58 suggests there are currently no suitable habitats for the group of species chosen, but that there are some useful areas to establish new habitat. However, because there is no information on what marshy grassland species includes it is difficult to see the benefit of this model for this work, and so it will not be used further in this work. Again the model appears to be defined more by land use than any of the other major datasets used in the pre-processing steps. In wider decision making the map of habitat connectivity would be difficult to use as the categories are not clear and it does not produce enough detail for the landscape even if the number of species choices did increase.

3.5.2.11 Summary

Ecosystem services are extremely popular in current literature and policy, but still little is known about how to manage them in a landscape and how they should feed into decision making. By producing these maps it is hoped that it will provide clearer information on ecosystem service delivery within the HHL in order to feed into decision making, as well as contributing to restoration options in later chapters. However, the actual worth of these maps is questionable as some of them are extremely difficult to interpret and locate data for.

3.6 Discussion and conclusion

The main aim of the chapter was to choose a range of metrics, and map these across the HHL in preparation for the second objective. However, there are many points and questions that can be brought out of the discussions concerning mapping various metrics, and their usefulness to decision making.

3.6.1 Mapping

The selection of ecosystem services metrics for this study was based on recommendations from the extensive academic and policy literature in this field (e.g. EEA, 2007; Biodiversity 2020 Strategy (Defra, 2012); HHL Partnership, 2011; Maltby, 2010) with consideration of the context of the landscape, the end point of the work, data availability and using the models available. Data availability, as well the ecosystem services models, is what defined the actual mapping process, as has been the case with other studies (Kandziora et al, 2013). Many studies discussed how useful it is for decision making for information to be spatially defined in maps (Kandziora et al., 2013), as has been attempted here, but there are some issues. For example, the HHL is very fragmented at a fine scale, which makes mapping at the landscape scale more complex. Further to this, the maps represent ecosystem service supply, rather than flows or demand, which would need to be reflected upon if used to inform decision making (Kandziora et al., 2013).

In terms of the models used, there are a few issues that have come through in the analysis, as these are very general models. The HHL landscape has been highly modified by humans, which could have implications for using both of these models. The InVEST model has been tested worldwide but was originally designed for North America, which may have some implications. For example, this geological background could be the reason that the recreational services with Flickr has not worked as well as expected, due to the large open natural spaces found in North America. The LUCI model was designed around rural, terrestrial landscapes, but more testing has been carried out on upland catchments and Jackson et al. (2013b) has suggested it needs more work on flat landscapes. It was suggested that perhaps LUCI has picked up some historic aspects of the HHL landscape, for example the large phantom water bodies seen in several models.

3.6.2 Decision making

There are many issues to consider with how useful these maps are for decision making, with questions such as what are the implications for management and do these maps just add confusion or useful information. Also, there is little trust in the outputs, what do they actually mean and if they were to be used what sort of decisions would it lead to. Currently, biodiversity and ecosystem services do get fed into policy, but not at a large scale, and much more information and work at the appropriate scale and format is needed before this occurs (Daily et al., 2009; Goldstein et al., 2012; Sitas et al., 2013; Albert et al., 2014a). Very few studies use scientific methods to predict or test restoration (Holl et al., 2003), and so by collecting this data together it provides a big resource for decision making in the HHL, as well as a case study for other areas to compare with, even though there are some major reservations.

Policy needs to have clear objectives for the environment, rather than just being a side interest (Piper et al., 2006) or something just done on just economic valuation (Tallis and Polasky, 2009; Frank et al., 2012; Tanner et al., 2013). However, there is a challenge as to how to communicate scientific findings into decision making (Hatton MacDonald et al., 2014) and an interdisciplinary

approach is needed (Egoh et al., 2008; Cotter et al., 2014), along with methods that are easy to use, transparent and robust (Albert et al., 2014b). Maps can act as powerful communication tools and provide information to policy makers about trade-offs and synergies (Jiang et al., 2013; Buckhard et al., 2013; Vorstits and Spray, 2015), and so the information in this chapter will be extremely useful in the following chapters.

3.6.3 Summary

Much of the information and discussion from this chapter will feed into the following chapters looking at Objective Two. When creating restoration options in Chapter Five there will be many issues to consider, such as that of trade-offs, and whether different services and biodiversity indicators can be delivered simultaneously in the landscape.

Chapter Four – Biodiversity and ecosystem services in the current wetland patches of the HHL Landscape

4.1 Introduction

The purpose of this chapter is to address the first part of Objective Two, which is as follows;

To use the information from Objective One to address:

(a) What biodiversity and ecosystem services are delivered by the current wetland patches in the HHL landscape?

In the previous chapter, various metrics of biodiversity and ecosystem services were identified and mapped across the landscape in order to increase knowledge of the HHL and to inform the process of the identification of optimal locations for wetland restoration. An extremely large data set has been amassed here to represent biodiversity and ecosystem services in the landscape, something which has not been done before, and is a big step forward for understanding the landscape. The next step is to examine the distribution of biodiversity and ecosystem services across the HHL landscape, to see if there are differences between the wetland patches and the rest of the landscape, to attempt to understand what the current wetland patches deliver. Using GIS and basic statistical techniques applied to a grid system, the levels of agreement between each of the mapped sets of data and the current wetland patches will be calculated. This analysis will also generate useful information for developing a range of restoration options in the following chapter. The analysis will be conducted at two spatial scales (500 m and 1000 m) with the purpose of investigating which is optimal and the influence of different scales on the interpretation of the landscape, as undertaken by Holt et al. (2015).

The restoration of large areas of wetlands is not feasible, and so we need to enhance what already exists and integrate patches of restored wetlands into larger landscapes. There does appear to be an increasing awareness of the need for action at the landscape scale (Bennett and Mulongoy, 2006; Maltby and Acreman, 2011), but there is still uncertainty as to how this should be delivered.

Information on the spatial distribution of biodiversity and ecosystem services, such as the metrics from the previous chapter, can guide decisions on restoration. However, there is uncertainty over how biodiversity and ecosystem services fit in with current wetland patches in the landscape, which would be important information to use as part of restoration decisions, especially at the landscape scale. The uncertainty is over which metrics are delivered in current wetland patches, and at what level or amount. This uncertainty is partly driven by the fact that there is still much unknown about the production of ecosystem services, including basic questions such as “where are ecosystems producing benefits?” (Bagstad et al., 2013a) and “what is the spatial distribution (of ecosystem services)?” (Wu, 2013). Also, ecosystem services have not yet been properly integrated into decision making (Koschke et al., 2012), and they are difficult to observe, with a lack of monitoring data (English Nature, 2006).

There is much discussion in the literature about how ecosystem services are delivered in wetlands, as it has been identified that wetlands provide services from all four categories (Tanner et al., 2013). In the literature, carbon storage is a service often associated with wetlands and they are also renowned for providing little food or timber (Tanner et al., 2013; Pan et al., 2013; Van der Biest et al., 2014). It is important to identify the most important ecosystem services for each particular landscape (Peh et al., 2013), as it is impossible to study everything, which will be partly understood by looking at what the current wetland patches in the HHL deliver. Understanding what current patches deliver will also aid with understanding ecosystem service trade-offs and synergies in the landscape, which is important (Tallis and Polasky, 2009; HHL Partnership, 2011; Rey Benayas and Bullock, 2012; Pan et al., 2013), as well as the delivery of multiple ecosystem services (Pan et al., 2013).

Biodiversity in wetland environments has been more extensively studied, but there are still unanswered questions and more work is urgently needed, because biodiversity is in increasing decline (Feld et al., 2009; Butchart et al., 2010). Globally, up to 50% of wetlands have now been lost (Feld et al., 2009; Cui et al., 2012), but it is currently not clear how to halt this loss. Also, relatively little is known about the biological attributes of floodplains (Chipps et al., 2006), an important part of many wetland environments in the UK. Wetlands are important

for biodiversity, both of which are in need of protecting, but more research is needed to fully understand how and why. By understanding what current wetland patches are delivering in terms of biodiversity, it will be useful to answer these key questions and develop restoration options in the next chapter.

Few studies have explored the spatial relationships between areas that are optimal for ecosystem service delivery and those for high biodiversity (Anderson et al., 2009). Whilst some studies have identified strong links between biodiversity and ecosystem services (Loreau et al., 2001; Norris, 2011; Van Oudenhaven et al., 2012), this research is still very much in its infancy, with many questions remaining unanswered (Luck et al., 2009; Norris, 2011; UK National Ecosystem Assessment, 2011). Reyers et al. (2012a) have suggested that the links that exist between biodiversity and ecosystems services could have a lot of potential in terms of trying to face the sustainability challenges of the future, and potentially inform landscape scale restoration. The landscape context information from Chapter Two is also useful, as these are important for biodiversity and ecosystem services (Botequilha Leitao and Ahern, 2002; Nelson et al., 2009; Jones et al., 2013).

A common, and often effective approach to mapping and understanding spatial distributions is to use presence/absence grids, which have the advantage of reducing the size of the data set, and simplifying further analysis, but can involve changes of scale and resolution that can mean some information is lost (Fehmi and Bartolome, 2001; Gibson et al., 2004; Conlisk et al., 2009; Muller et al., 2011). Grid systems also provide an accessible tool for decision makers, such as the HHL Partnership, as they require less time, less data and are simpler to use than other approaches. Engler et al. (2004) states that the size of the grid cell should be selected by taking into consideration the spatial accuracy of the observed information, and that a larger cell size will make data more manageable but might not represent the landscape as well. Examples of the methodology include Pereira and Itami (1991) and Van Horssen et al. (1999). Pereira and Itami (1991) created habitat suitability maps for red squirrels using various metrics on a 0.5 hectare grid size, and defined the presence (1) and absence (0) of the squirrels and each metric in each cell to understand the spatial distributions. Using 1 km by 1 km grid squares Van Horssen et al. (1999) mapped the

distribution of land use, soil type, infiltration and water level to better understand vegetation restoration in wetland environments.

In order to address these issues raised, and therefore the objective, a method is developed to understand which metrics have a good association with the current wetland patches, and therefore what the current wetland patches deliver in terms of ecosystem services and biodiversity, which can then be fed into the restoration options in the following chapter. This information is important for a range of reasons, as discussed previously. The second objective of scale will be investigated by choosing two different scales and using the second of these as a comparison to look how landscape interpretation could differ. The structure of the chapter is as follows: methods, investigation of the metrics of landscape context, biodiversity and ecosystem services, discussion and conclusion.

4.2 Methods

This chapter uses the data collected in Chapter Three to further explore the metrics in the HHL, specifically looking at the delivery of biodiversity and ecosystem services in the current wetland patches and to further investigate restoration potential for this landscape. As well as using biodiversity and ecosystem services information from the previous chapter, landscape context information from Chapter Two will also be used. All of the metrics mapped in the previous two chapters will be used here, except for the habitat connectivity ecosystem service, as was explained in Chapter Three Section 3.5.2.10. The methods for this chapter use some of the same ideas as other work, such as Pereira and Itami (1991) and Van Horssen et al. (1999). The landscape is converted into a 500m and 1000m scale grid, and each metric is overlaid at each resolution. Converting each data set to a standard grid overcomes some of the issues with different sources of information and different scales, and using two grid sizes allows the effect of scale on landscape interpretation to be considered. Grid squares of 1000 m were chosen as it is a common scale used in the literature (e.g. Van Horssen et al., 1999) and 500 m was selected in an attempt to make use of some of the data that are at a better resolution. Using this method does mean that some more detailed information is lost, which could be overcome by

using smaller grid squares, but this would not work for all of the data because of the range of scales that it was found in.

Metrics that were purely presence or absence, for example SSSIs, is transformed to a grid representation by giving the squares in which the metric had more than 50% coverage a value of 1, and the squares that were less than 50% present a value of 0 (Figure 4.1). For the species data records there is a lot of variation in the resolution of the secondary data and so when converting to grids the species points were sometimes found in the centre of four grids squares on the grid lines, and so were recorded as present in all four grid squares. This could be a problem with interpreting the location of species in the landscape, but is the best way of dealing with the data. For metrics that were categorical, such as LUCI flood mitigation, the grid squares were assigned a number that represented the category that covered the largest area of each square. Metrics, such as InVEST pollination, that are continuous data, were put into equally sized bins, and treated in the same way as the categorical data. The dataset was amassed in ArcGIS (10.1) at both scales, but exported into Excel (2010) for analysis.

At both scales the grid squares that contain current wetland patches were collected together (94 squares at 1000 m, 472 at 500 m), using the 50% threshold to define a current wetland patch, as previously. The percentage agreement between each metric and these current wetlands was then calculated by counting up the occurrences of current wetland squares with each metric in ("Wetland (%)", for example the amount of wetland squares that SSSIs are found in. Next the percentage agreement between each metric and the non-wetland patches is also calculated ("Non-Wetlands (%)"). This was done by randomly selecting the same number of non-wetland grid squares as wetland grid squares and calculating the percentage of each metric in these new selections. Every random selection was carried out 100 times and the results averaged to give an estimate of the distribution of the metrics in the non-wetland areas of the landscape, with the standard deviation also calculated. Each square is taken independently, not considering the status of the squares around it. In each case, it is the percentage of current wetlands that contain these metrics, and not the percentage of each metric that is found in current wetlands. One issue is that associations may be

discussed between wetlands and metrics that might have been used to designate these areas as wetlands.

The results are presented in the following section, with the graphs containing binary data presented with the percentage agreement between each metric and the current wetlands patches, and the percentage agreement between each metric and the non-wetland patches. The categorical data are explained in each section and each result is rounded to the nearest integer. Each graph includes error bars for the non-wetland patches results, which is the standard deviation of the 100 random selections. The y axis of each is at different scales for visual clarity. More discussion will be given to the 1000 m results, but these will be compared to 500 m in order to interpret the secondary objective of what effect scale may have on landscape interpretation.

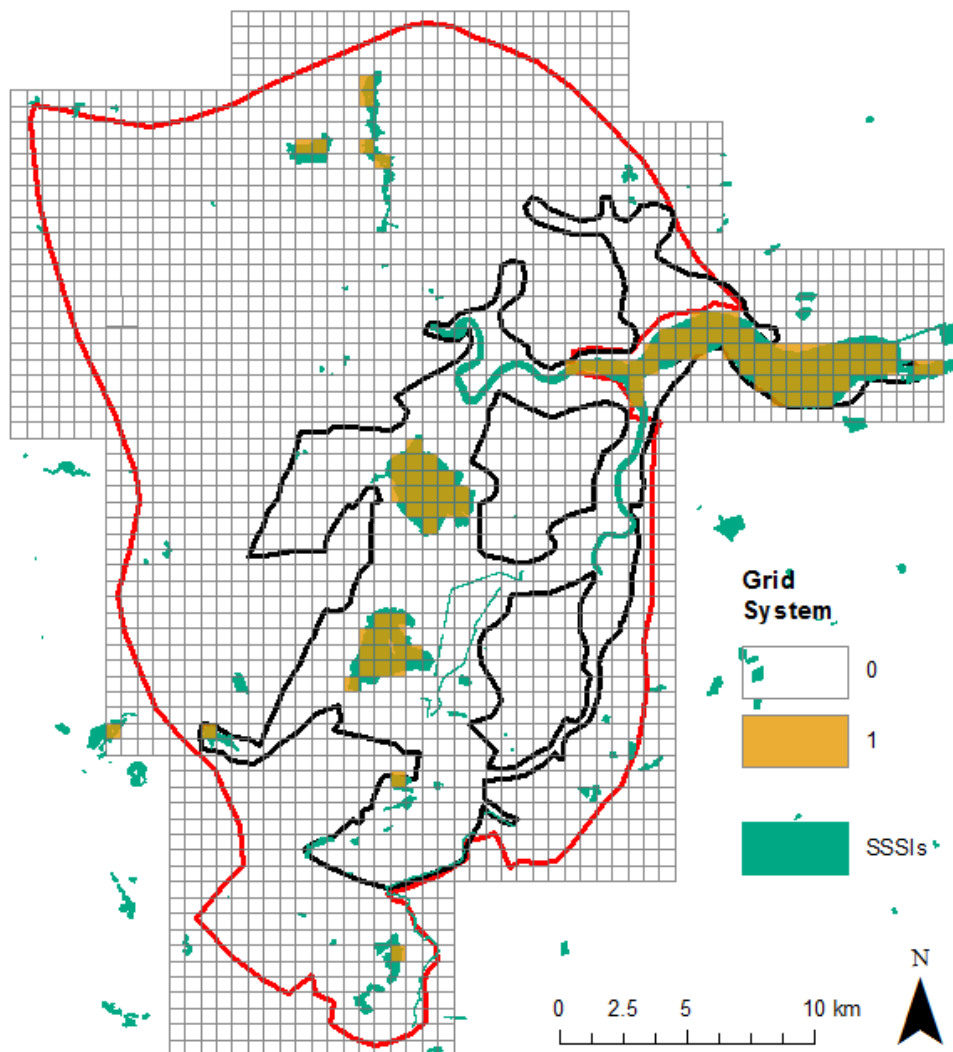


Figure 4.1: SSSIs shapefile and the conversion of this into the 1000 m grid system.

4.3 Results

4.3.1 Landscape context

4.3.1.1 1000 m

The results for the binary metrics are presented in Figure 4.2, which include land use, soil type, nutrient levels, topography, historic wetlands and areas where the NIA Partnership carried out their work (see Chapter Two Figure 2.12). There is a 70% agreement between the current wetland patches and the historic wetlands, compared to the non-wetland result of 34%. Therefore, approximately two thirds of the current wetland patches were recorded as being wetlands in the past, but perhaps actually near a 100% agreement should be expected. However, there have been wetlands created since then, this could also be affected by the definition of historic wetlands (see Chapter Three Section 3.3) and the 50% threshold method used in this chapter. With “NIA work” there is a 22% agreement with the current wetland patches, which is almost double that of the non-wetland patches, and suggests that the NIA programme created new wetlands and restored current ones. Out of the land use metrics, the one that has the strongest association with current wetland patches is “arable and horticulture”, which is found in 44%. However, the non-wetland patches have a very high value of 83%, suggesting that the whole landscape is dominated by “arable and horticulture” and the land use is not specific to wetlands. The “bog” land use has a relatively high association (found in 29% of wetland patches), as does “broadleaved, yew and mixed woodland” and “saltwater”, in relation to the non-wetland values.

From the soil metrics “raised bog peat” has a strong association, with a value almost 62 times higher than for non-wetland patches. “Slowly permeable seasonally wet slightly acid but base-rich loamy and clayey soils” and “loamy and clayey floodplain soils with naturally high groundwater” also have strong associations, but not as strong as “raised bog peat”. It is unsurprising that “raised peat bog” is the soil type that the current wetland patches contain, as the HHL is a wetland landscape with bogs integrated.

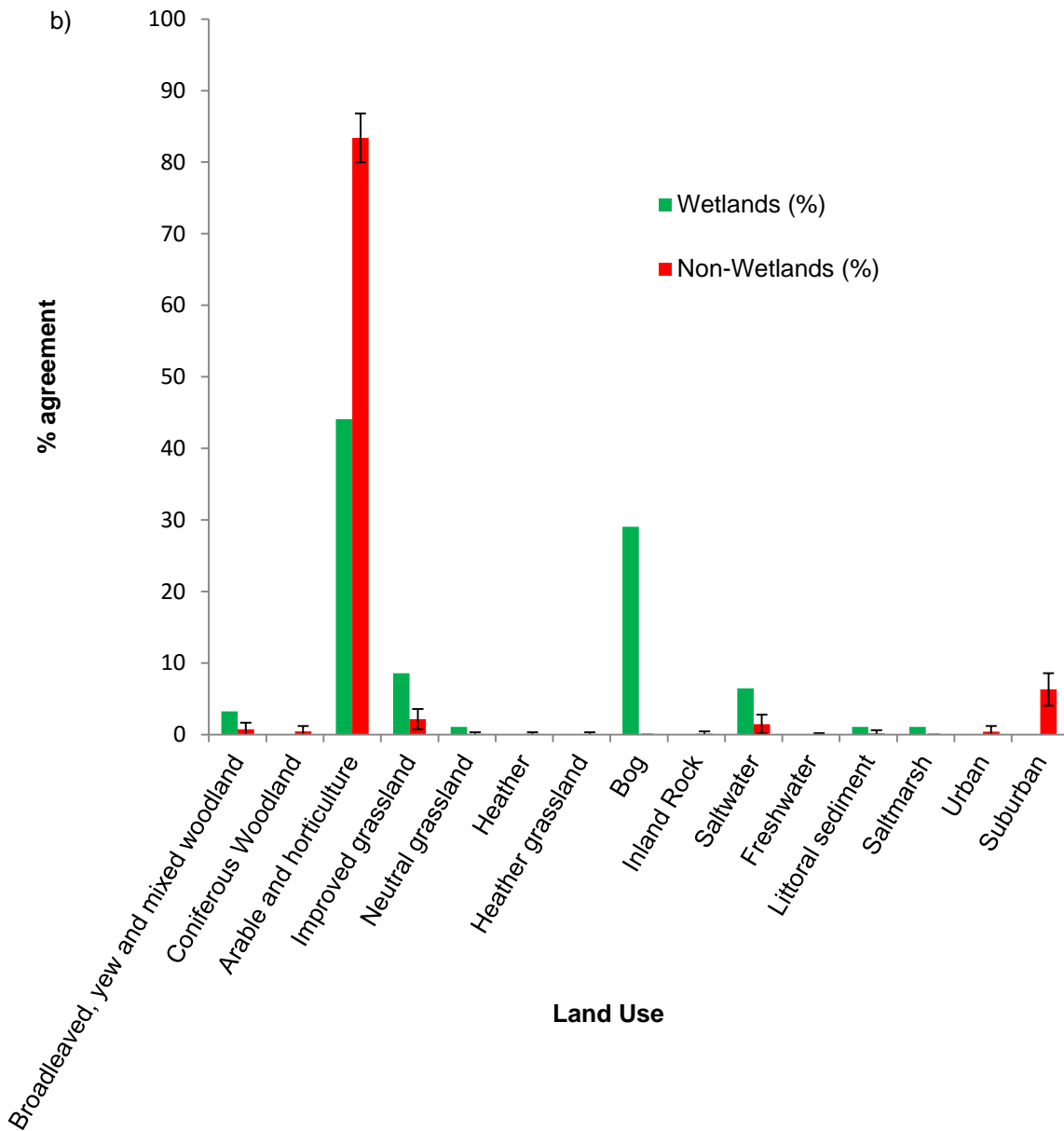
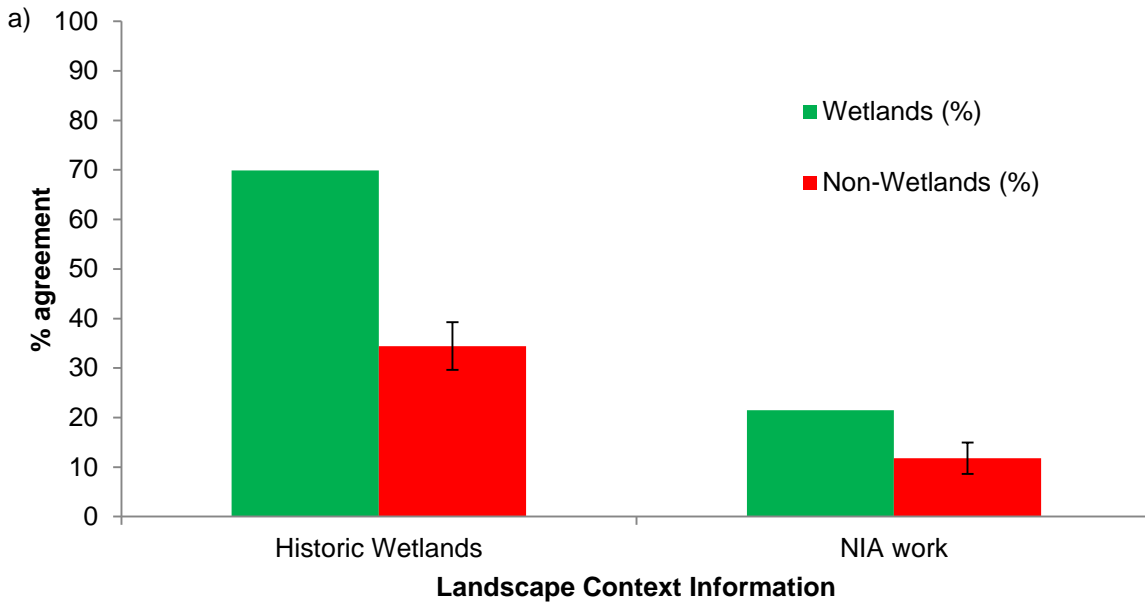


Figure 4.3 displays the results for the categorical nutrient metrics, which is split into the grades set by the Environment Agency on these individual chemicals. For ammonia and DO, “A” is the grade with the lowest amounts of each chemical, whereas it is “Grade 1” for nitrates and phosphates. There is a high percentage of “no data” across the landscape because the data only covers certain water bodies and so care needs to be taken with interpretation. In the case of ammonia and DO, where these are recorded, there appears to be lower levels of these in wetland areas. Figure 4.3(a) shows that 11% of the current wetland patches have “Grade B” levels of ammonia in the water bodies, whereas in the non-wetland patches it is 4%. For DO, the results in Figure 4.3(b) show that “Grade B” covers 10% of current wetland patches, whereas the non-wetland patches have a value of 6%. Overall, the current wetlands in the HHL tend to have lower levels of ammonia and DO than the non-wetland patches.

For nitrates and phosphates there is no clear pattern between wetland and non-wetland areas, both show fairly high levels, which could be an issue for the landscape as a whole, but is not something that the current wetland patches particularly contain. The final results for this section are for the topography metric (Figure 4.4), which is presented in metres above sea level. It shows that all of the wetland patches are found in the lowest height category (“0- 16.7” metres above sea level), whereas some of the non-wetland patches are found at higher levels (up to 133 metres above sea level). Therefore, the current wetland patches occur on low level land, which is unsurprising considering the characteristics of wetlands. Some of the information discussed here, such as soils and topography, will be useful for restoration options in the next chapter.

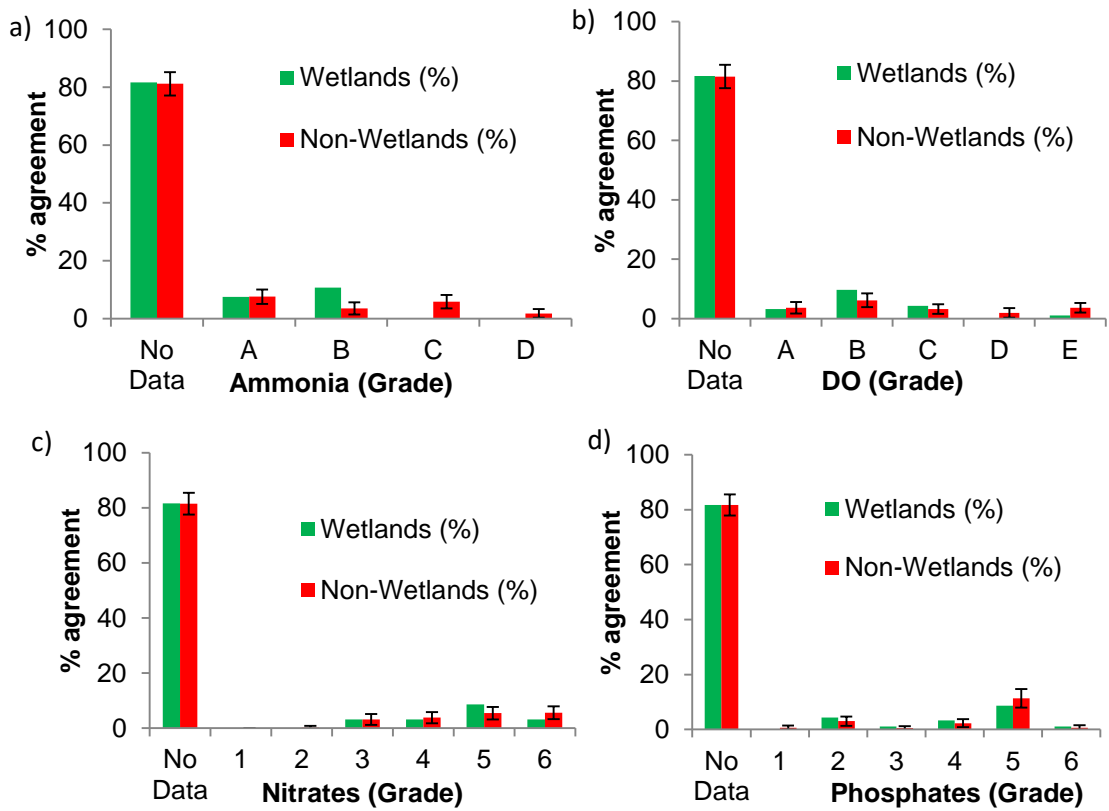


Figure 4.3: (a) Ammonia, (b) DO, (c) Nitrates and (d) Phosphates levels at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

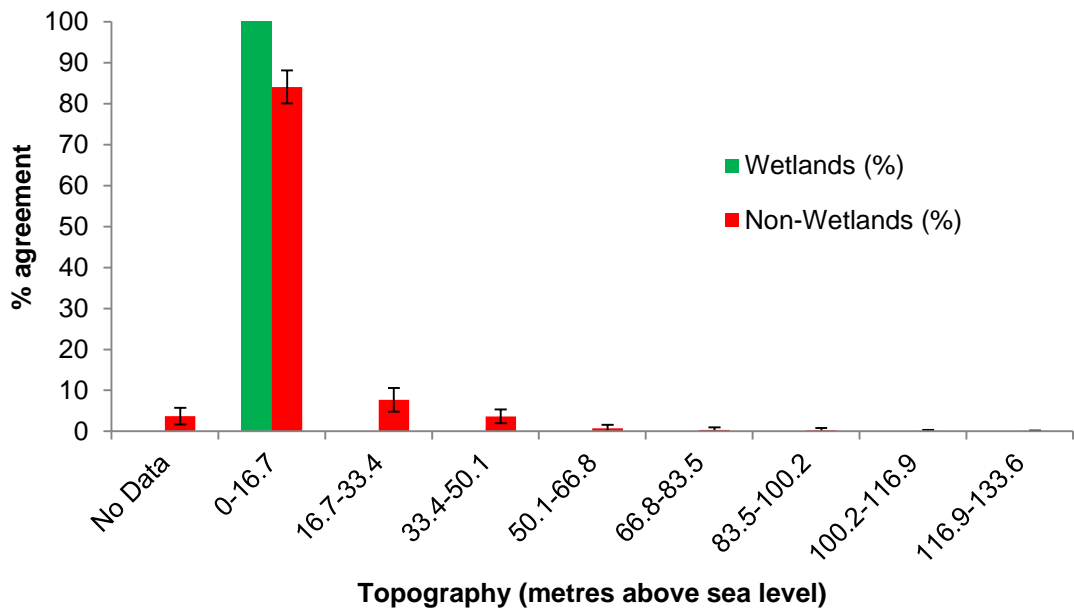
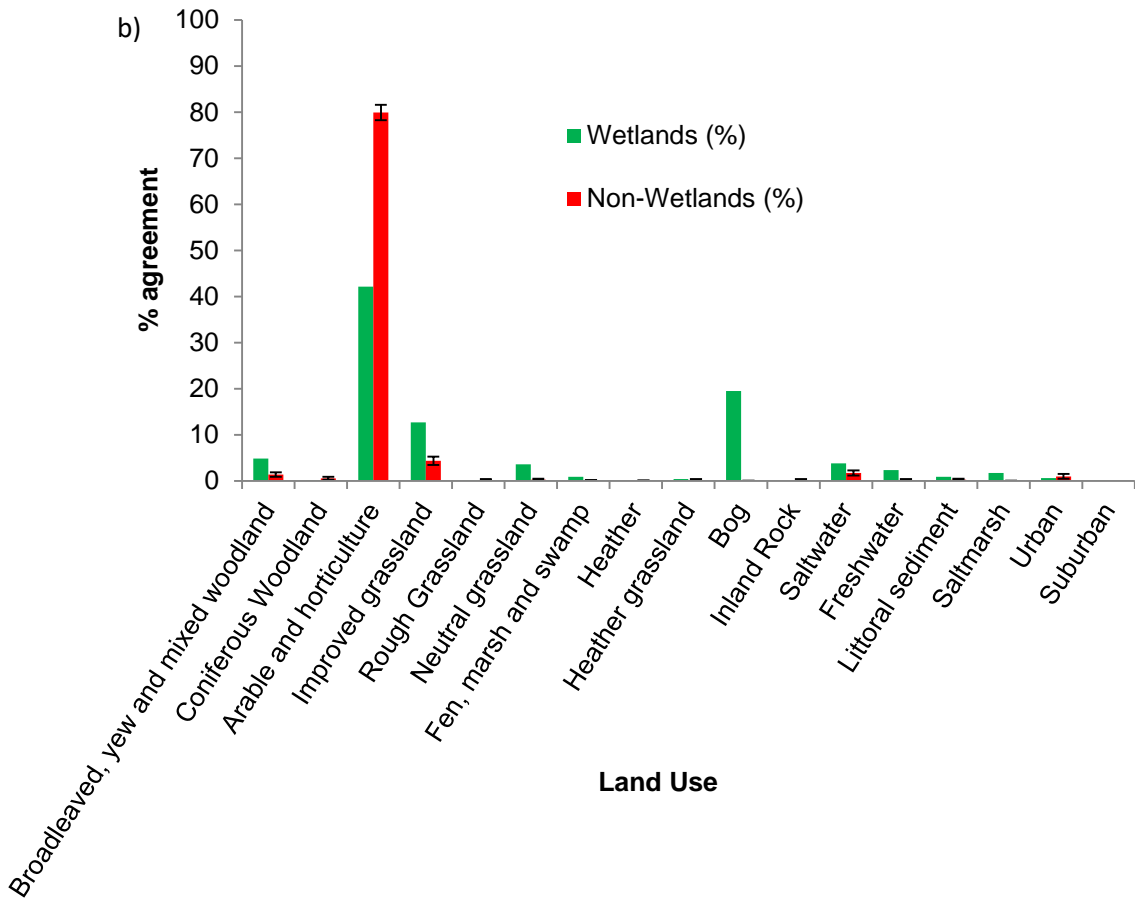
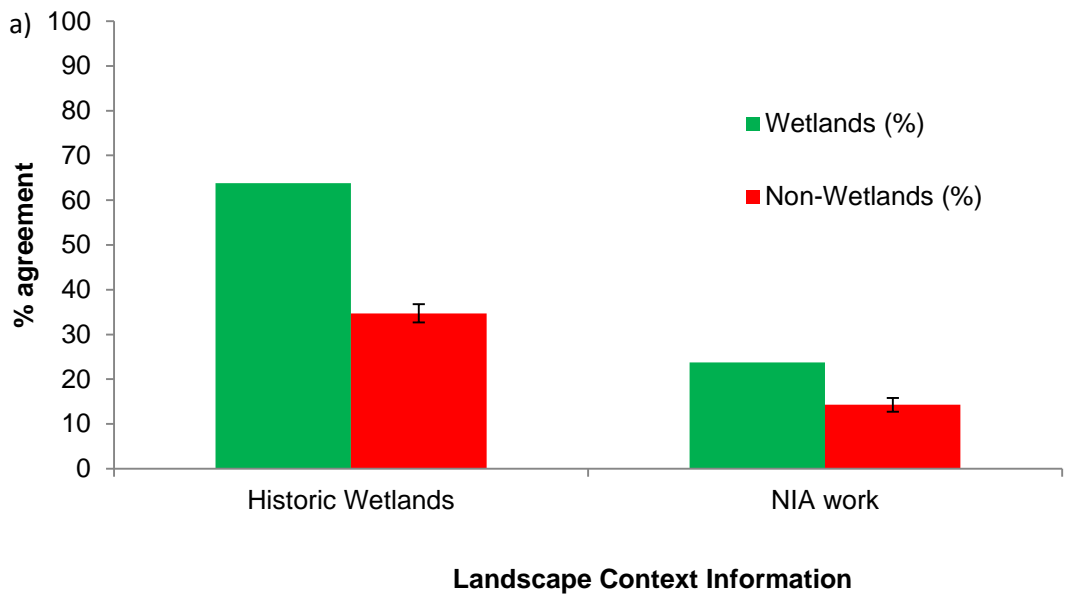


Figure 4.4: Topography at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

4.3.1.2 500 m

The first set of results are in binary format and are seen in Figure 4.5. Again, there is a high agreement between the historic wetlands and the current wetland patches, with nearly double the amount of squares in the current wetlands containing historic wetlands compared to the non-wetland patches. As discussed at the 1000 m scale, this is not surprising considering the landscape history. There is also a relatively high level of agreement between the “NIA work” and the current wetland patches (24%) compared to the non-wetland patches (14%), as with the 1000 m results. At the 500 m scale the “arable and horticulture” land use still shows a strong potential association with wetland patches in the HHL of 42%, with a very high occurrence in non-wetland areas (80%). Therefore, again the landscape is dominated by “arable and horticulture” but it is not something that the wetland patches particularly contain. “Bog” and “improved grassland” show stronger associations, especially “bog”, as 19% of current wetland patches contain this land use, opposed to approximately 0% for non-wetland patches, which is not a surprise for the HHL landscape.

Out of the soil types category, there are three that show some potential association with wetlands, which can be seen in Figure 4.5, with wetland results of 22%, 27% and 25%. However, the “slowly permeable seasonally wet slightly acid but base-rich loamy and clayey soils class” also has a high value for the non-wetland patches. “Raised bog peat soils” and “loamy and clayey floodplain soils with naturally high groundwater” both show strong associations with current wetland patches compared to the non-wetland patches (over 200 times more), which is the same as at the 1000 m scale.



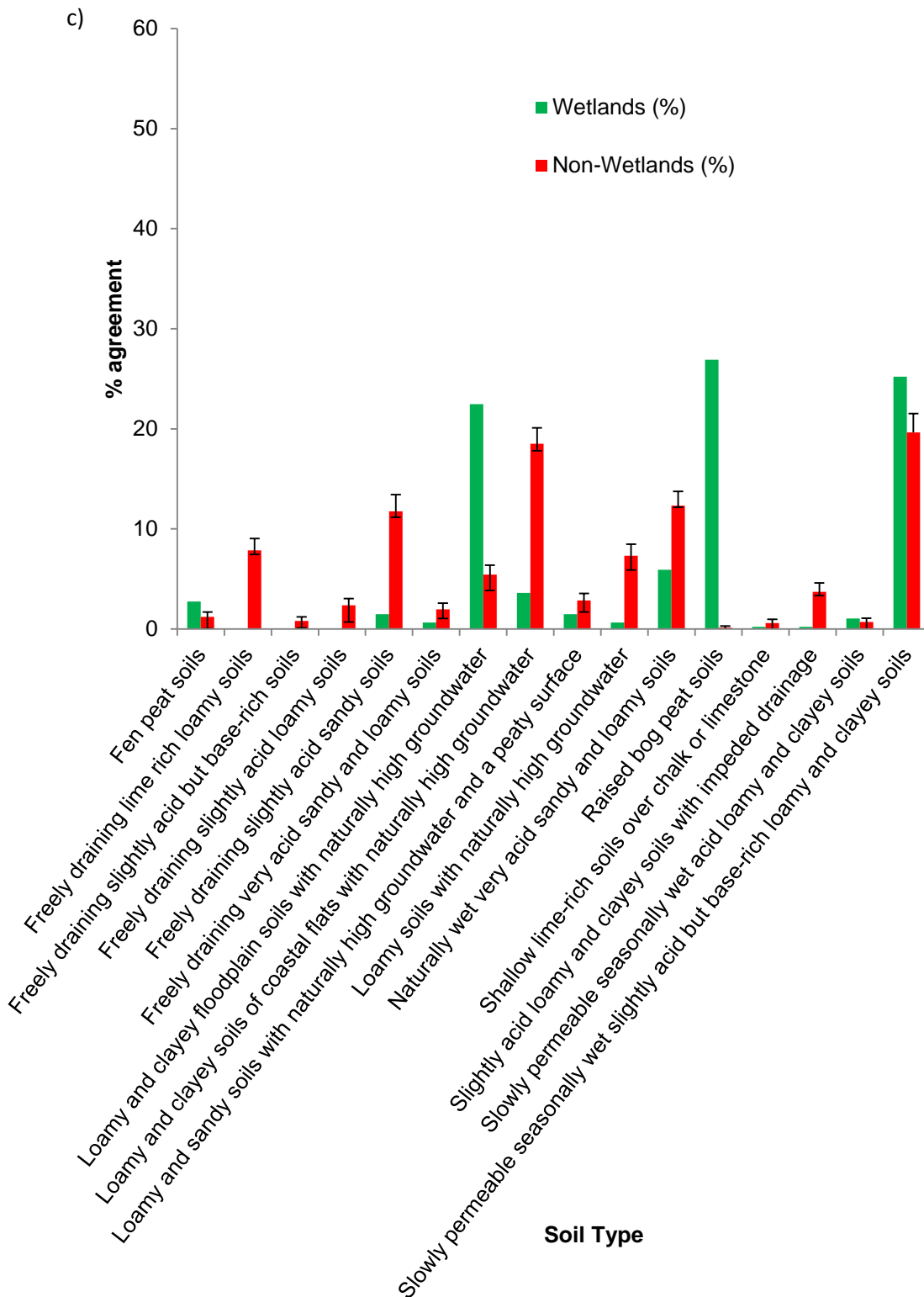


Figure 4.5: (a) Binary landscape context information at the 500 m scale (b) Land use at the 500 m scale (c) Soil types at the 500 m scale. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric. It is the percentage of grid squares that contain these metrics, and not the percentage of each metric that is found in the grid squares.

The nutrient data at the 500 m scale is presented in Figure 4.6. As with the 1000 m scale data, categories “A” and “1” are the lowest levels for each chemical. Both ammonia and DO show a slight pattern towards lower levels within current wetland patches, although it is difficult to say because the wetland areas have more data than the non-wetland areas. In Figure 4.6(a) the current wetland patches show much higher results for “A” and “B” (up to three times). For DO Figure 4.6(b) shows over three times higher results for current wetland patches in “B” and over two times lower for “E”. Therefore, the current wetland patches in the HHL landscape have lower levels of ammonia and DO than the non-wetland patches, as occurred at the 1000 m scale. Nitrates and phosphates again show no clear pattern between wetland and non-wetland patches, possibly the current wetlands have a slightly higher level of these chemicals, but again there is more data in these areas than the non-wetland patches, and so it is difficult to identify any pattern.

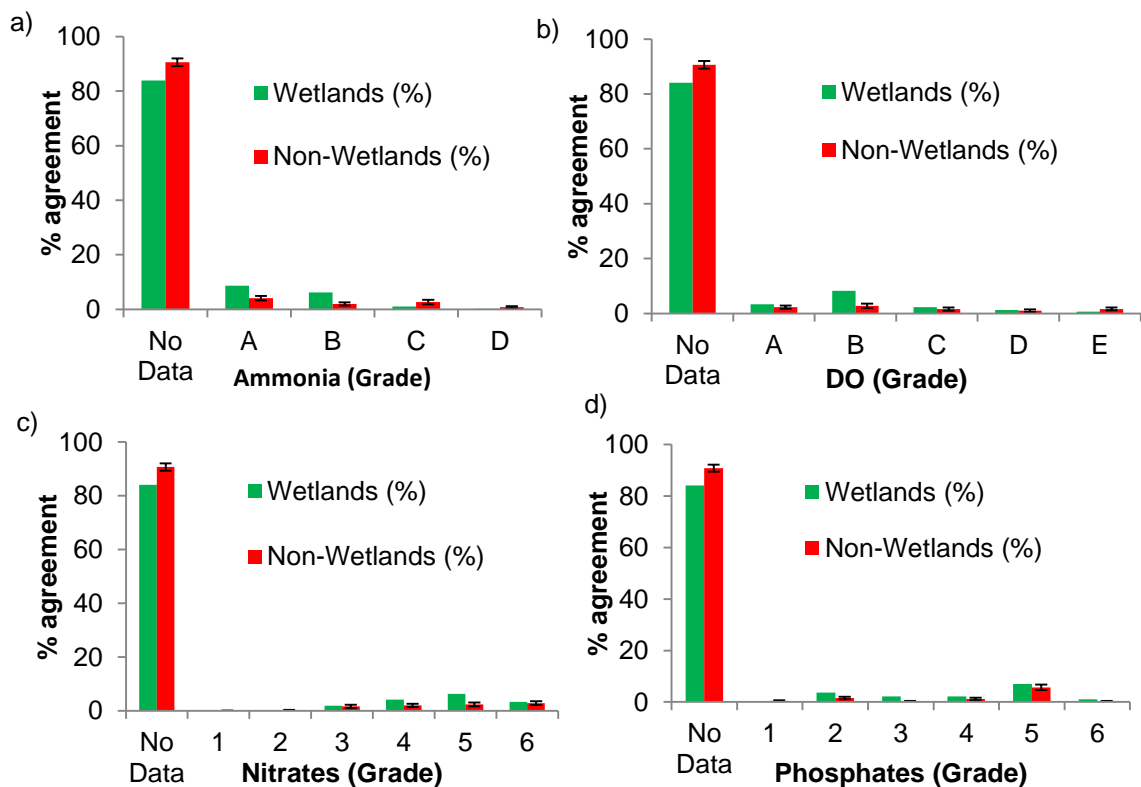


Figure 4.6: (a) Ammonia, (b) DO, (c) Nitrates and (d) Phosphates levels at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

Again the topography metric (Figure 4.7) shows that the vast majority of wetland patches are found at the lowest level (“0-16.7” metres above sea level), whereas some of the non-wetland patches are found at higher levels of between 116.9 and 133.6 metres above sea level. Therefore, again the current wetland patches are found at the low levels of the HHL landscape in relation to the non-wetland patches. At the two scales, similar results are produced for landscape context, some of which will be useful for the restoration options in the following chapter. There are also slight differences in results between the two scales, such as the “improved grassland” land use, but the pattern of associations is similar, and so in this case the choice of scale does not make a huge difference to the interpretation.

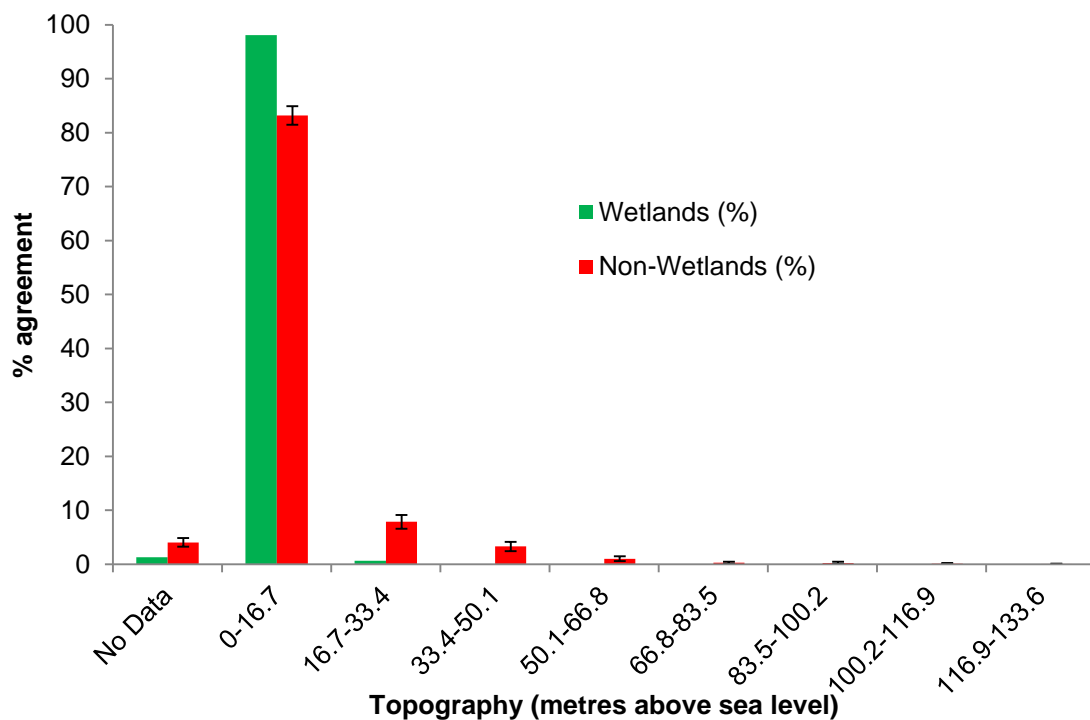


Figure 4.7: Topography at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

4.3.2 Biodiversity

4.3.2.1 1000 m

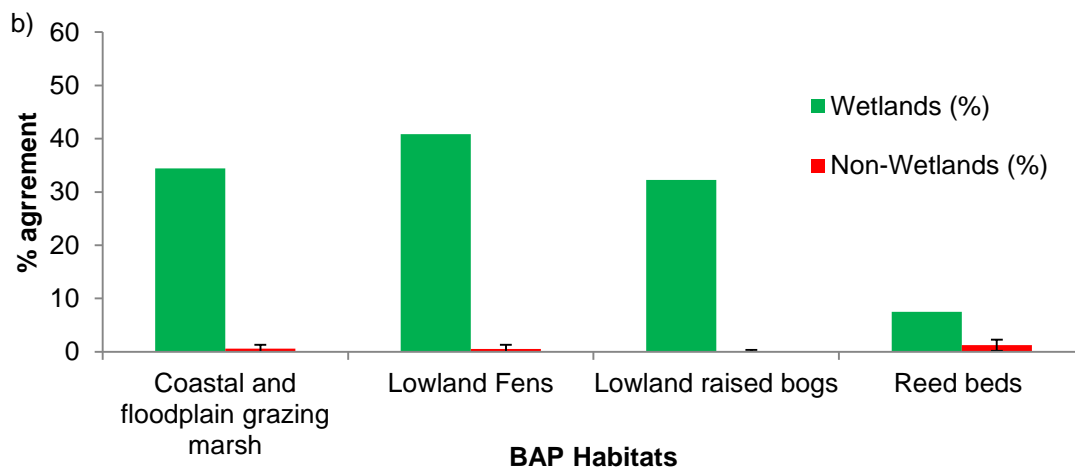
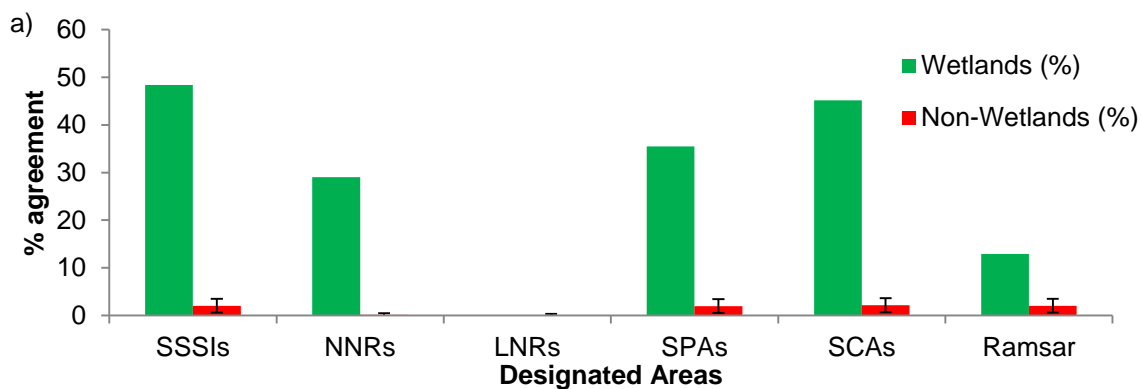
There is a lot of variation in the results between the biodiversity metrics and current wetland patches at the 1000 m scale (see Figure 4.8). In general, except

for the LNRs, the protected areas have strong potential associations with the current patches. For example, Figure 4.8(a) shows that SSSIs have a 48% agreement with current wetland patches, compared to only 2% with non-wetland patches. If protected areas are used as a proxy for potential areas of higher biodiversity in the landscape these results suggest that the current wetlands patches could deliver high biodiversity in the HHL. In general, all four of the BAP habitats included, show good associations with current wetland patches compared to non-wetland patches, as seen in Figure 4.8(b). The strongest association comes with “lowland raised bogs”, with a result of 32% with wetland patches and nearly 0% with non-wetland patches, meaning that raised bog is more than 300 times more likely to occur in a current wetland patch than a non-wetland patch. Again, if these BAP habitats are used as a proxy for potential areas that deliver higher biodiversity in the landscape, then these results suggest that the current wetland patches could deliver higher levels of biodiversity compared to the non-wetland patches.

Figure 4.8(c) displays the widespread and focal species results at the 1000 m scale. These terms are used because these are used in the NIA Monitoring and Evaluation literature, where widespread species covers groups of generalists, and focal species means more specific species that are conservation priorities in an area. In general, widespread (or groups of) and focal species tend to have much weaker associations with current wetlands, but it could be due to the fact that these metrics are not often recorded across the whole of the landscape. It would be expected that there would be more occurrences of the species within current wetland patches, but the records are small and possibly incomplete. It also needs to be taken into consideration that these patterns reflect where species have been recorded and not necessarily where they occur in the landscape. Also, some of the groups of species included in widespread are more general and would possibly not be expected to necessarily have a strong association with wetlands in particular. However, there are still cases in Figure 4.8(c) where the results for the current wetland patches is much higher than the non-wetland patches.

Out of the widespread species (Figure 4.8(c)) the strongest associations come from the breeding wetland birds, with the current wetland patches having over 10

times the amount of non-wetland patches. For the focal species (Figure 4.8(c)) one of the strongest associations comes from the Cranes, which gives 2% for current wetland patches and nearly 0% for non-wetland patches. It would be expected that focal species have a strong association with current wetland patches because these tend to be conservation priorities. Therefore, some of the widespread and focal species are suggesting high delivery of these species to the HHL landscape, even though some of the actual percentages are not particularly high. The inclusion of Himalayan Balsam and Japanese Knotweed are there as a disservice, the reasons for which were discussed in the previous chapter. At the 1000 m scale the current wetland patches deliver more Himalayan Balsam and less Japanese Knotweed than the non-wetland patches. However, because they are not an advantage to wetlands they will not be included in the analysis going forward. There some results in Figure 4.8 which will be useful for developing restoration options in the following chapter, such as SSSIs and wetland habitat plants.



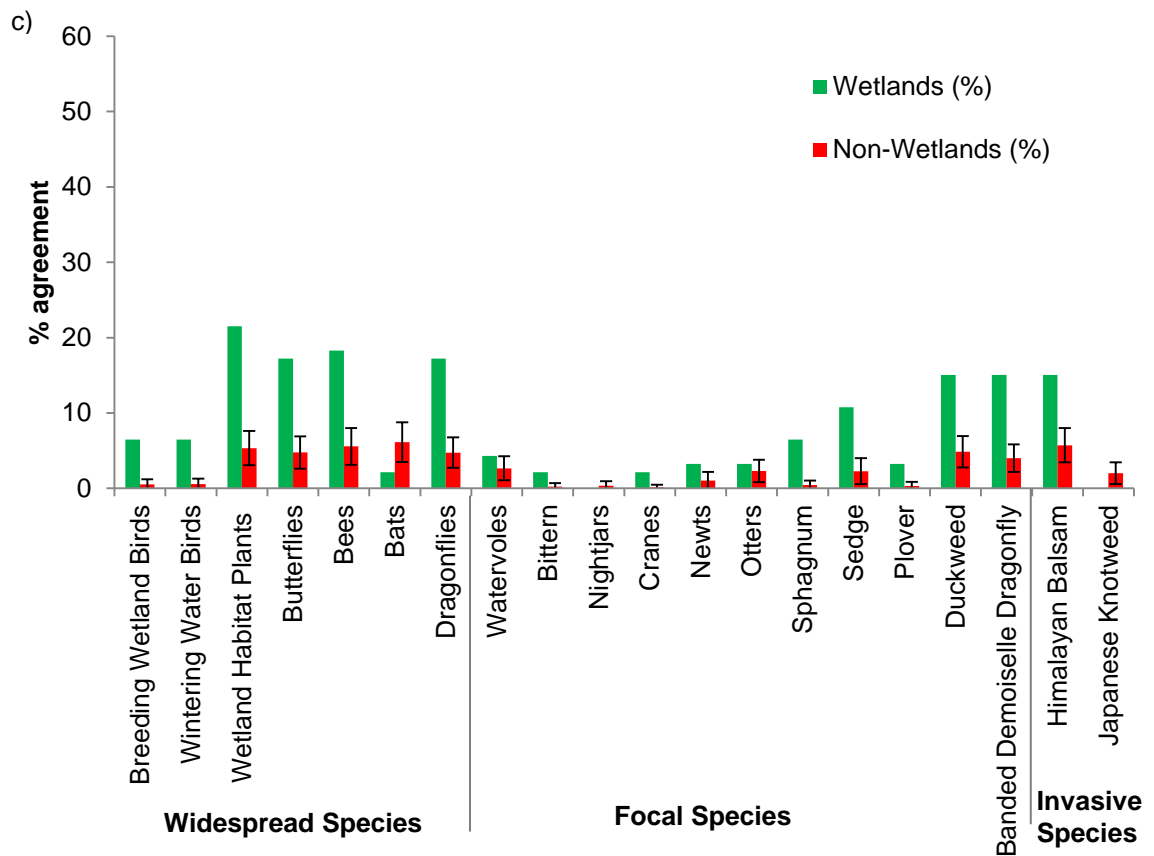


Figure 4.8: (a) Designated areas at the 1000 m scale. (b) BAP habitats at the 1000 m scale (c) Widespread and focal species at the 1000 m. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric. It is the percentage of grid squares that contain these metrics, and not the percentage of each metric that is found in the grid squares.

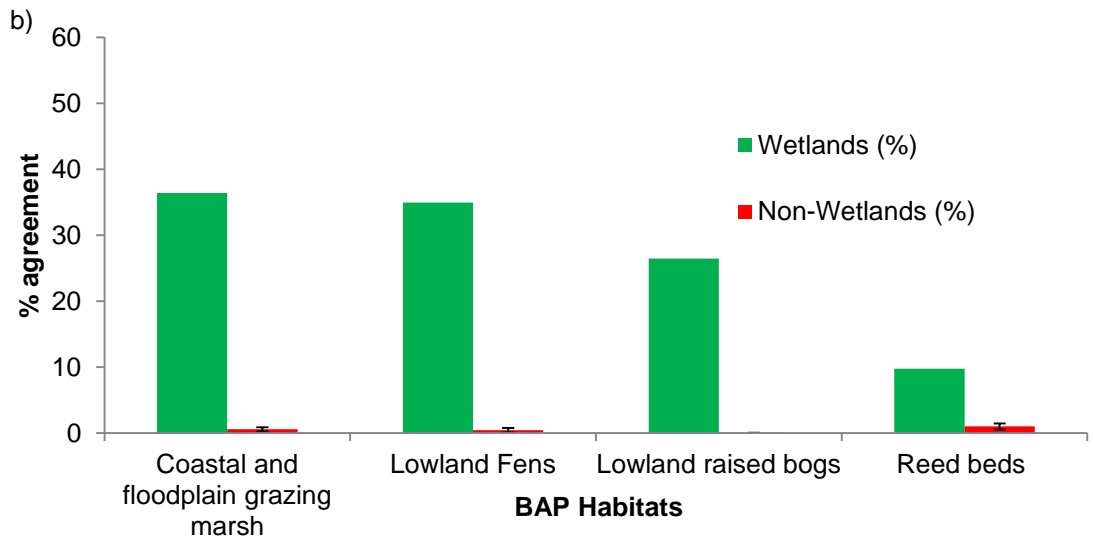
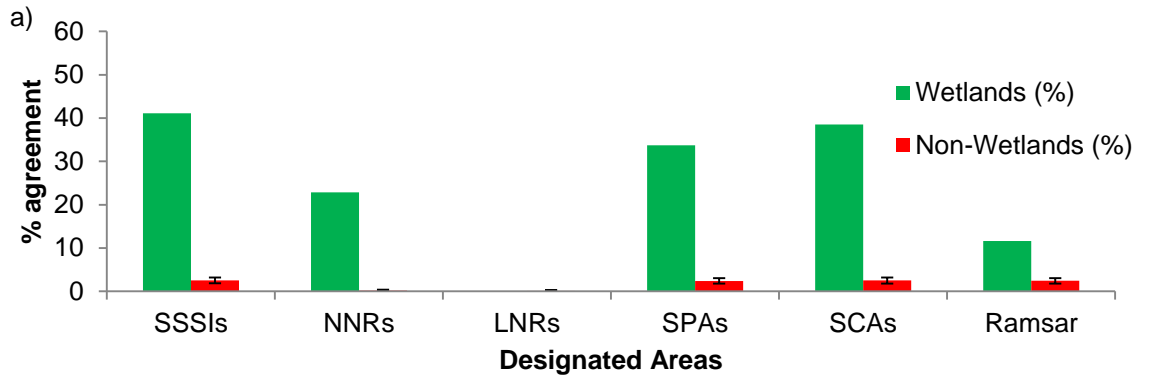
4.3.2.2 500 m

At the 500 m scale there is also a lot of variation in results between the current wetland patches and the biodiversity metrics, which are presented in Figure 4.9. Again, strong potential associations occur with most of the protected areas, except for the LNRs, especially compared to the occurrence values in the non-wetland areas, as seen in Figure 4.9(a). For example, the SCAs have a wetland patches result of 39% but a non-wetland result of 3%, which means SCAs are more than 15 times more likely to be found in current wetland patches. These strong associations are not surprising, and as discussed at the 1000m scale again suggest that the current wetland patches could deliver high levels of biodiversity in the HHL. The results for the four BAP habitats are shown in Figure 4.9(b) and again, as with the 1000 m scale results, all four show strong associations with current wetland patches. For example, the “lowland fens” have

a result of 35% with the current wetland patches, whereas the result for non-wetland patches is only 0.44%. Again, if these BAP habitats are used as a proxy for suggesting higher levels of biodiversity, then these results suggest that the current wetland patches deliver higher levels of biodiversity.

In terms of the widespread species and focal species, these again have much lower values with the current wetland patches, but are often still greater than the occurrences in non-wetland patches, as seen in Figure 4.9(c). Once again, the issue needs to be raised that many of these species hardly feature at all in the landscape, which will affect some results, and that these species records could be incomplete. Also, these records are of where a species was observed and not necessarily where they are normally found in the landscape, which needs to be considered in reading the interpretation of the results. Out of the groups of widespread species in Figure 4.9(c) a strong association is found between breeding wetland birds and current wetland patches for example, as the current patches result is over 10 times higher than the non-wetland patches result. In Figure 4.9(c) an example of a strong association with focal species and current wetland patches is with bittern, which produces a value of 1% for current wetland patches, but almost 0% for non-wetland patches, a difference of more than 10 times. Therefore, there are some results from the groups of widespread and focal species that suggest high delivery within the current wetland patches of the HHL landscape. The disservice of invasive species show similar results to the 1000 m scale, and again because they are not an advantage to wetlands they will not be included in the analysis going forward.

Overall the two scales produce fairly similar results patterns, although with slightly different values. The biodiversity metrics show bigger differences between the two spatial scales than was observed for the landscape context metrics. For example, wetland habitat plants have current wetlands scores of 22% at the 1000 m scale and 8% at the 500 m scale. Therefore, if these values were used as part of the decision making process for restoration, the difference in scale will change how the landscape is interpreted. However, the instances where biodiversity metrics have a good association with current wetland patches could be used to inform potential restoration options, such as with many of the protected areas.



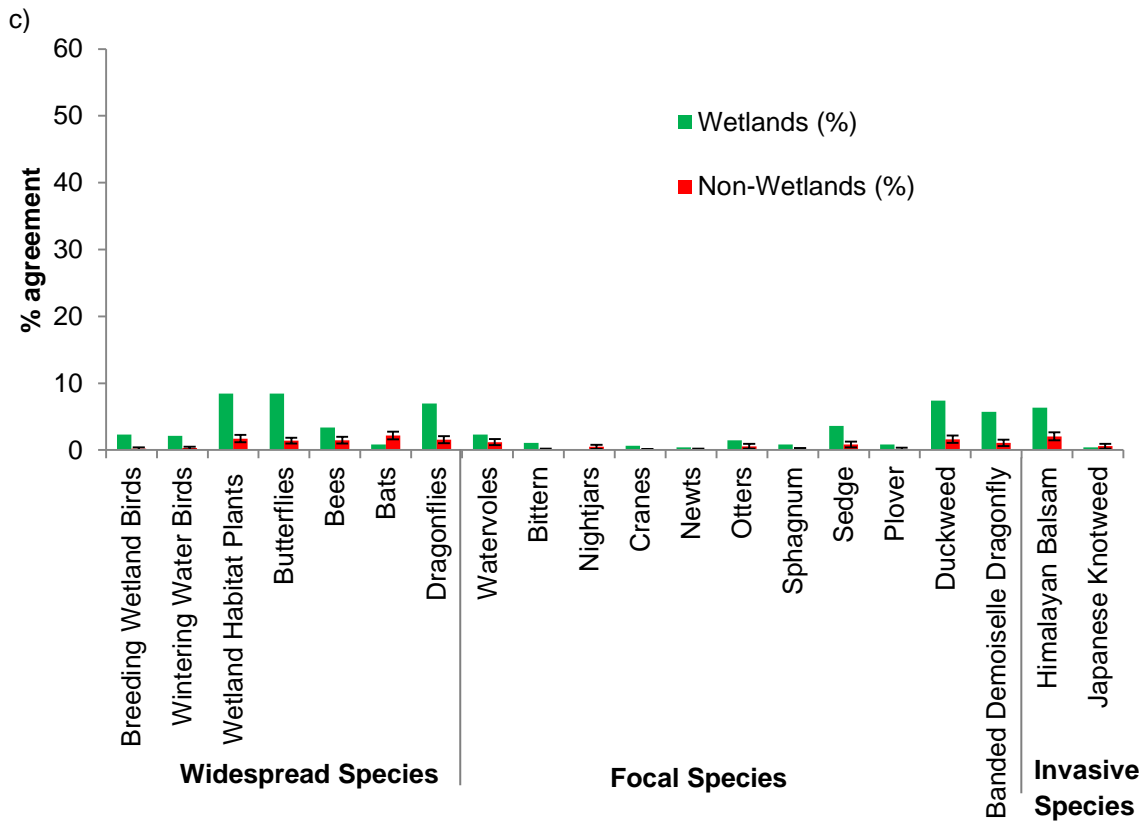


Figure 4.9: (a) Designated areas at the 500 m scale. (b) BAP habitats at the 500 m scale (c) Widespread and focal species at the 500 m. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric. It is the percentage of grid squares that contain these metrics, and not the percentage of each metric that is found in the grid squares.

4.3.3 Ecosystem services

4.3.3.1 1000 m

There is a lot of variation in the wetland results between the ecosystem services metrics at the 1000 m scale. Figure 4.10 displays the binary ecosystem service results, with the subsequent figures covering the categorical ecosystem services. The provisioning ecosystem service of agricultural production is represented by multiple metrics. The results in the first part of Figure 4.10 cover the service of sustainable agricultural production. However, there is no discernible pattern between the non-wetland patches and wetland patches for the different schemes of sustainable agricultural production, but the non-wetland patches have more overall coverage of these schemes, totalling up to 40%, whereas the total value for wetland patches is 20%. For the agricultural production ecosystem service there are the LUCI model results (Figure 4.11(b)), with clear categories, and

agricultural land class information (Figure 4.11(a)), with “Grade 1” being the best agricultural land. The LUCI results suggest that the clearest category for wetland patches is “negligible production” with a value of 38%, opposed to a value of 1% for non-wetland patches. “Grade 5” from the agricultural land classes gives the strongest association with the current wetland patches, as the result is more than 50 times that of non-wetland patches, which is the worst quality agricultural land. Therefore, the LUCI and agricultural land class results suggest that the current wetland patches in the HHL landscape are not the best areas for agricultural production, and the sustainable production results back this up with more of the schemes located outside current wetland patches.

There are two sets of metrics representing the regulating service of flood mitigation in the HHL, which are the last two results of Figure 4.10 and all of Figure 4.12. A particularly high value is found with the flood alert areas, with an 85% agreement with current wetland grid squares, opposed to a 47% agreement with non-wetland patches, suggesting that the wetland patches are more at risk from flooding. The flood storage areas presented in Figure 4.10 are from the Environment Agency as areas that attenuate flood waters. These results suggest that the current wetland patches are less likely to be able to store water, which is a surprise for a wetland landscape. However, the LUCI model mitigation results in Figure 4.12 suggests that there is a much stronger association between “flood mitigating land” and wetland areas (39%), than with non-wetland patches (3%), which is more to be expected. Overall, the delivery of flood mitigation in the current wetland patches is unclear due to different sets of results, but it would be expected that some of these would be useful for flood water storage.

The information on woodlands in active management in Figure 4.10 suggests that, at this scale the wetland areas support more “unmanaged woodland” than “managed woodland” areas, with a value of 12% for unmanaged woodland and 0% for managed woodland. Therefore, the wetlands deliver potential for the delivery of this ecosystem service by having unmanaged woodland, but do not currently deliver actively managed woodland. The cultural service of recreation is represented by the recreation metric from InVEST (see Figure 4.13), which is in the form of user photo days per year, used as a proxy for recreational use within a landscape. In the wetland areas, there is currently a low level of recreation, with

most squares (92%) having less than five user photo days per year. For more than five days the non-wetland patches have a cumulative result of 5%, whereas the current wetland patches have only 2% over five days. Recreation is also represented by PROW in Figure 4.10. This does not suggest an increase in access to the environment (a proxy for recreation) in current wetlands, as the occurrence of PROW is higher in non-wetland patches. Therefore, these two sets of results suggest that current wetland patches deliver less recreational ecosystem services than non-wetland patches.

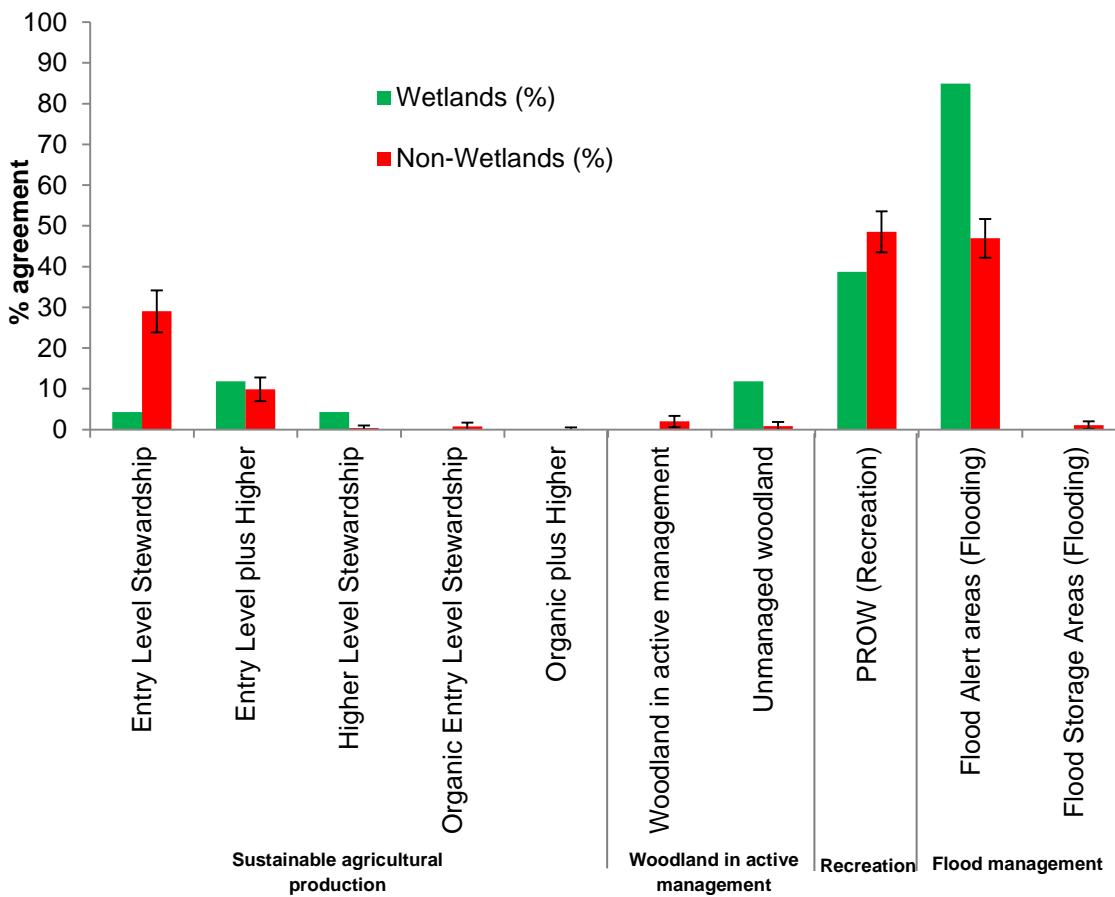


Figure 4.10: Binary ecosystem service metrics at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric. It is the percentage of grid squares that contain these metrics, and not the percentage of each metric that is found in the grid squares.

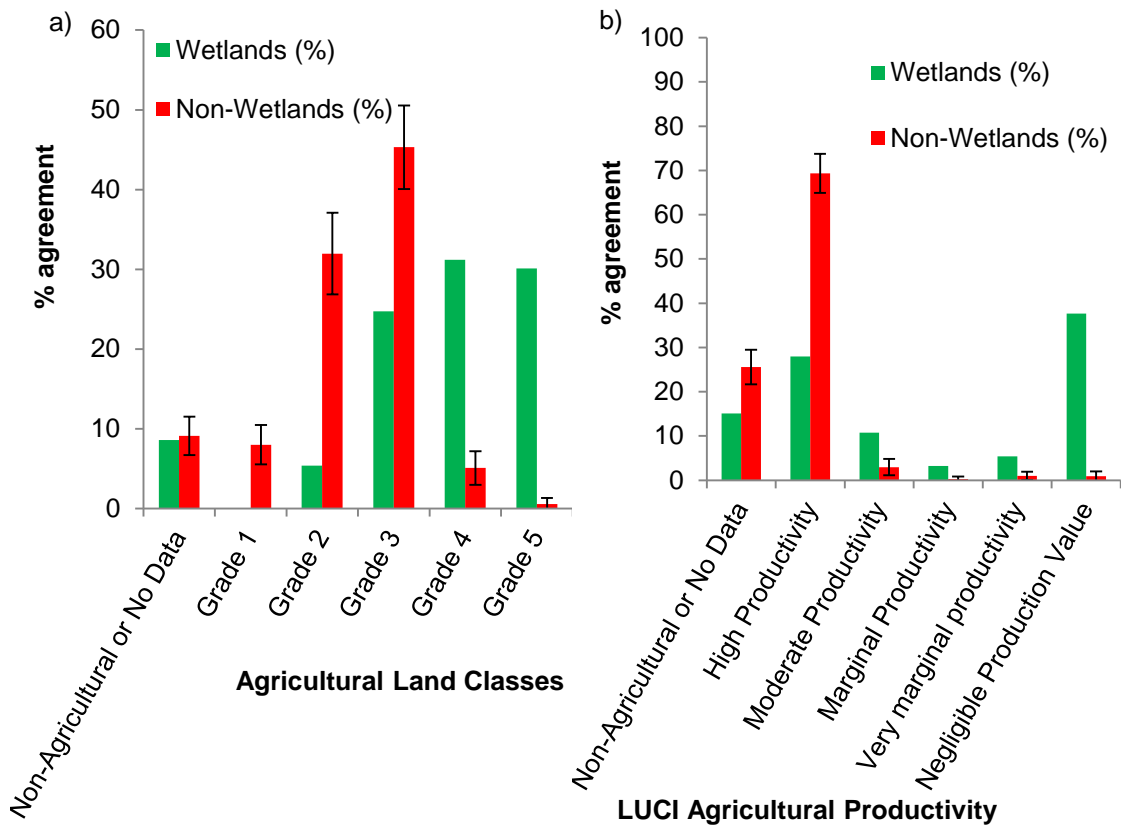


Figure 4.11: (a) Agricultural land classes and (b) results of the LUCI Agricultural Productivity model at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

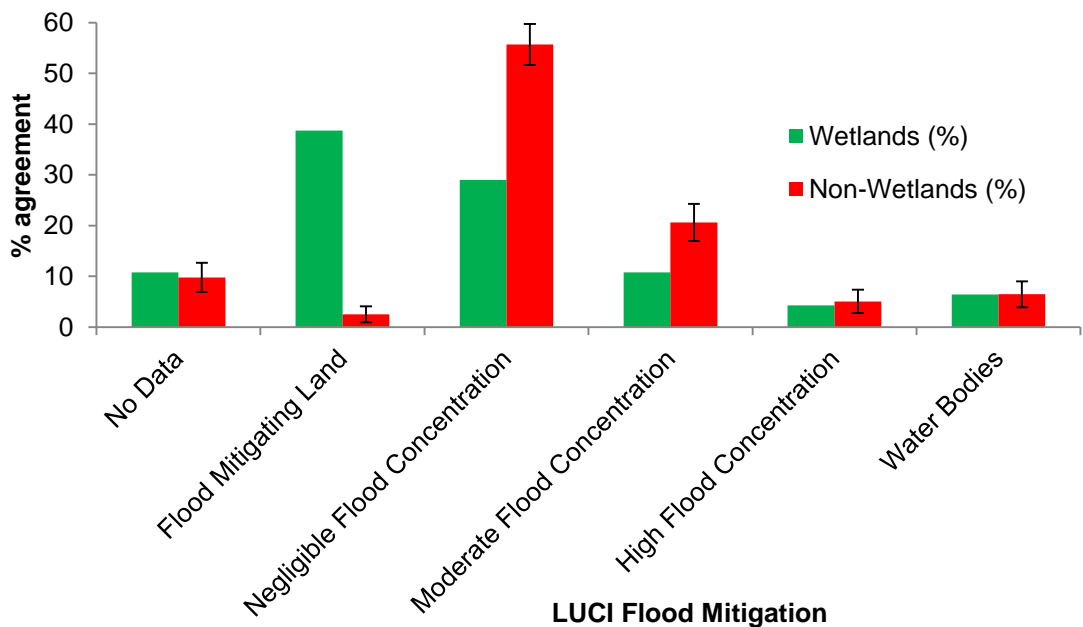


Figure 4.12: Results of the LUCI flood mitigation model at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

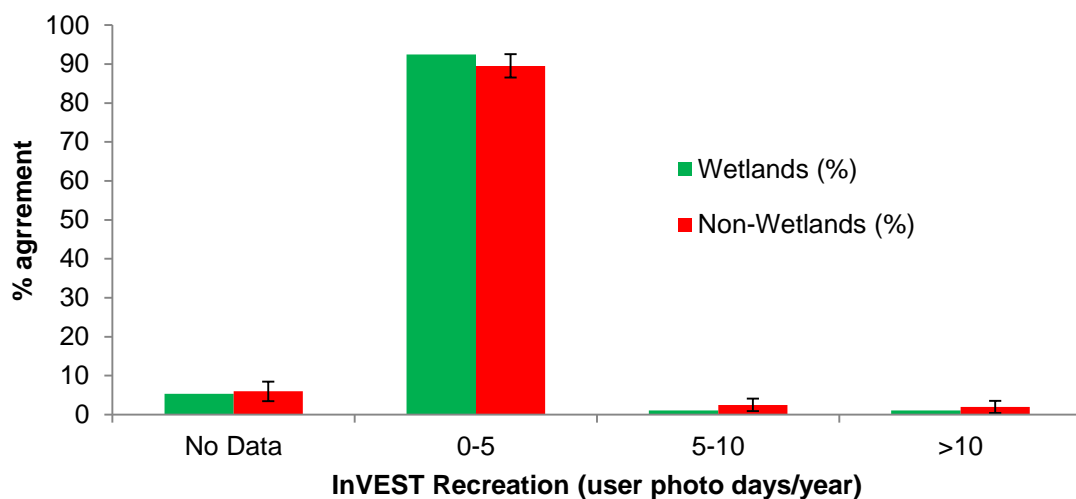


Figure 4.13: Results of the InVEST recreation model at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

There are also two sets of results looking at habitat quality in the landscape, LUCI, which uses categories (Figure 4.14(a)) and InVEST, which is on a relative scale of habitat quality between 0 and 1 (Figure 4.14(b)). Habitat quality represents the service of providing good habitat in the landscape. The InVEST results show that habitat quality is high across the whole landscape, but that it is higher in current wetland patches (94%) than non-wetland patches (88%). The LUCI results are a little more difficult to interpret, but appear to suggest that there is 20 times more “habitat of interest” (taken as wetland habitats) in the current wetland patches than non-wetland patches. In general, both sets of results appear to agree that there are more areas of higher habitat quality in the current wetland patches and so they deliver this ecosystem service.

The results for the supporting service of pollination come from the InVEST model. The model output shows a relative scale of pollination levels, and the results are presented in Figure 4.15. From these it is difficult to see any pattern in pollination between the current wetland patches and non-wetland patches, except that perhaps there is more pollination outside the current wetland patches, as the top two bins have cumulatively higher values for non-wetland patches. Therefore, the current wetland patches in the HHL landscape do not deliver greater levels of pollination than non-wetland areas. The regulating and supporting service of carbon storage is represented by outputs from both InVEST and LUCI models.

These are shown in Figure 4.16 and both represent carbon storage in tonnes per hectare, with equal categories chosen as there is no set threshold of good carbon storage. Both models show increased carbon storage in the wetland patches, with both models producing 30% with in “600-1200 tonnes per hectare” in current wetland patches and 0% with non-wetland patches, which is as expected. Therefore, the current wetland patches in the landscape deliver higher levels of carbon storage.

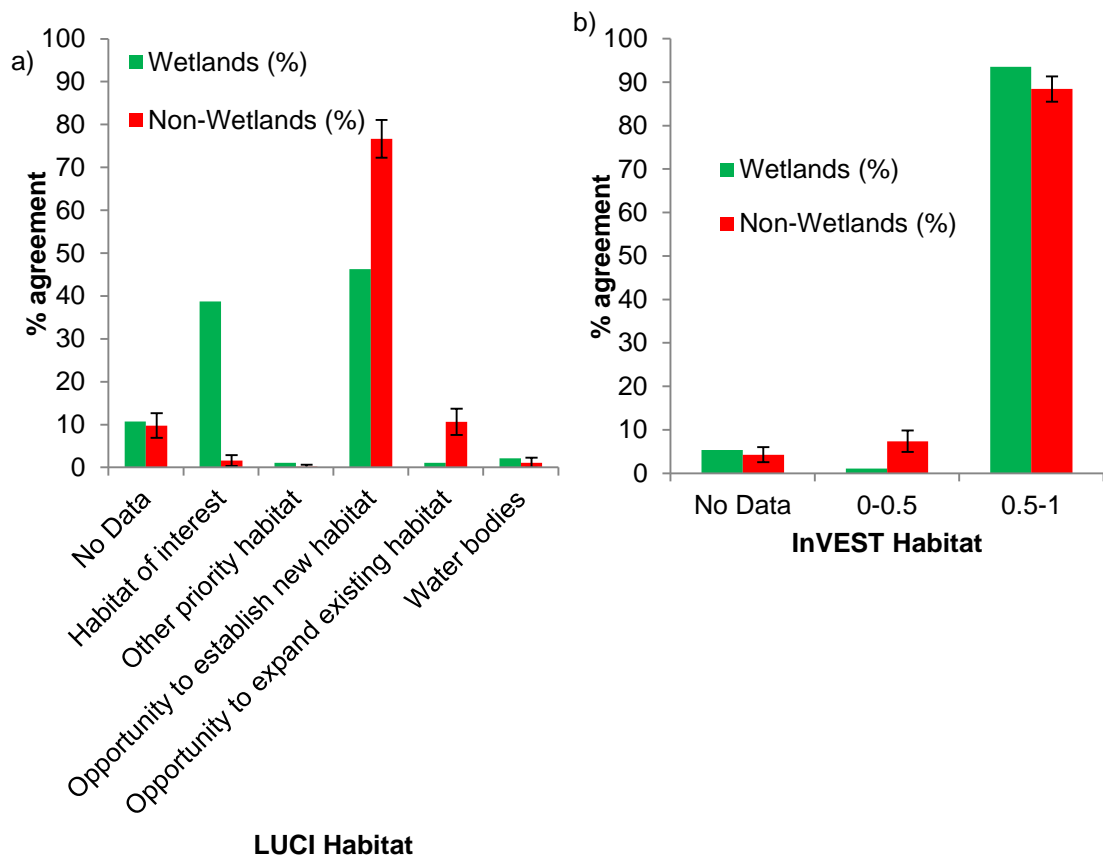


Figure 4.14: (a) Results of the LUCI habitat model and (b) the InVEST habitat model at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

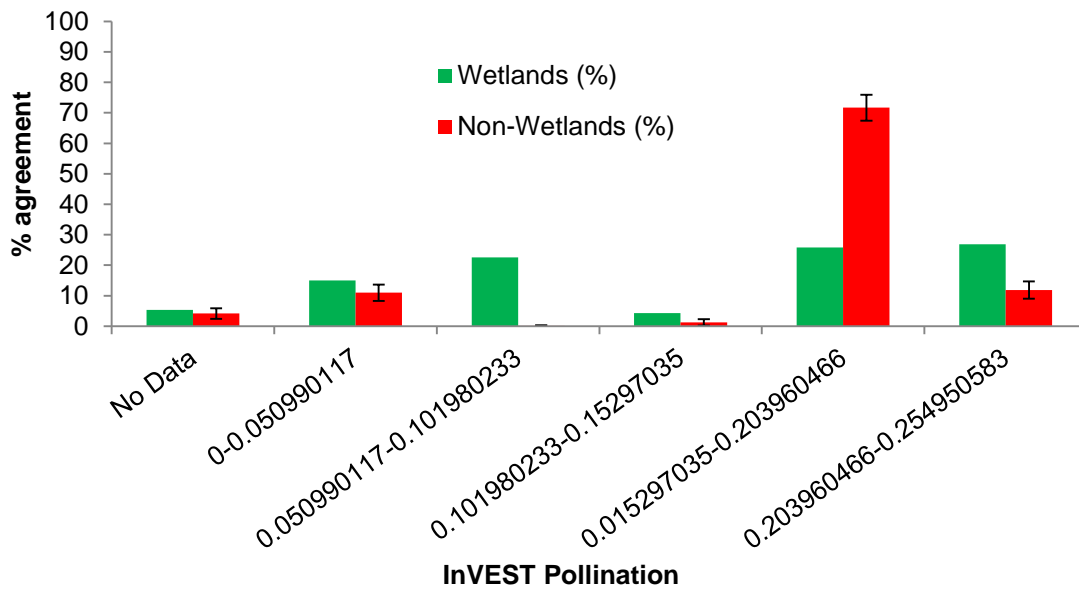


Figure 4.15: Results of the InVEST pollination model at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

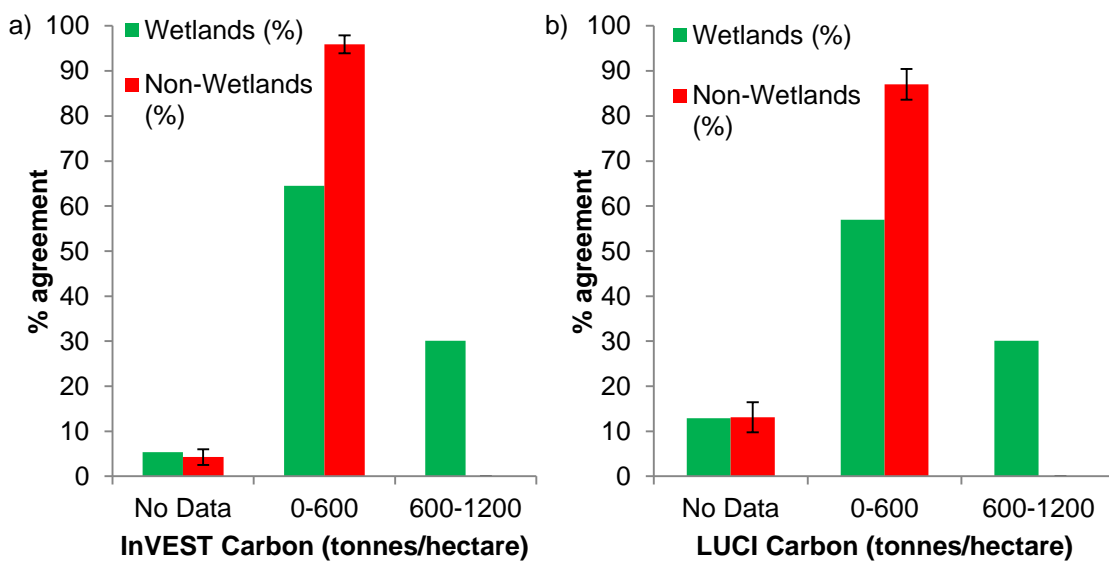


Figure 4.16: (a) Results of the InVEST carbon model and (b) the LUCI carbon model at the 1000 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

4.3.3.2 500 m

Investigation of the landscape at the 500 m scale also shows high variation in wetland results in the ecosystem services metrics with current wetland patches and therefore variation in the delivery of ecosystem services by the current wetland patches. Flood regulation shows a similar association with wetland patches at the 500 m scale compared to the 1000 m scale, with a higher proportion of flood alert areas occurring in wetland patches (Figure 4.17), suggesting more flooding, and increased flood mitigation in current wetland patches (29%) opposed to non-wetland patches (3%), according to LUCI (Figure 4.18). Therefore, the results at the 500 m scale are similar to the 1000 m scale, with the two sets of information on flood mitigation disagreeing on the current wetland patches' ability to mitigate floods. The provision of woodland shows a similar trend at 500 m and 1000 m, with current wetland patches also delivering more areas of "unmanaged woodland" than woodlands in active management (Figure 4.17). This confirms that wetlands have potential to deliver the woodland in active management ecosystem service.

Recreational ecosystem services were measured using the same two metrics as with the 1000 m scale. The PROW results in Figure 4.17 suggest that as a proxy the current wetland patches deliver less recreation, compared with the 1000 m results. The InVEST recreation results (Figure 4.19) do not really offer a difference in pattern between the current wetland and non-wetland patches, which is different from the 1000 m scale. Overall, the two different sets of recreational data do not agree and so it is difficult to understand the delivery of recreational ecosystem services in current wetland patches at the 500 m scale. This is however not the case with the three different sets of results on agricultural production as an ecosystem service, which as with the 1000 m scale results show increased agricultural production outside the current wetland patches. These results are presented in Figure 4.17 (areas of more sustainable agricultural production of which there are more of these schemes in non-wetland patches), in Figure 4.20(a) (where the strongest wetland association is with "Grade 5", the poorest grade of agricultural land) and Figure 4.20(b) (where the strongest association with wetland patches is "negligible Production Value").

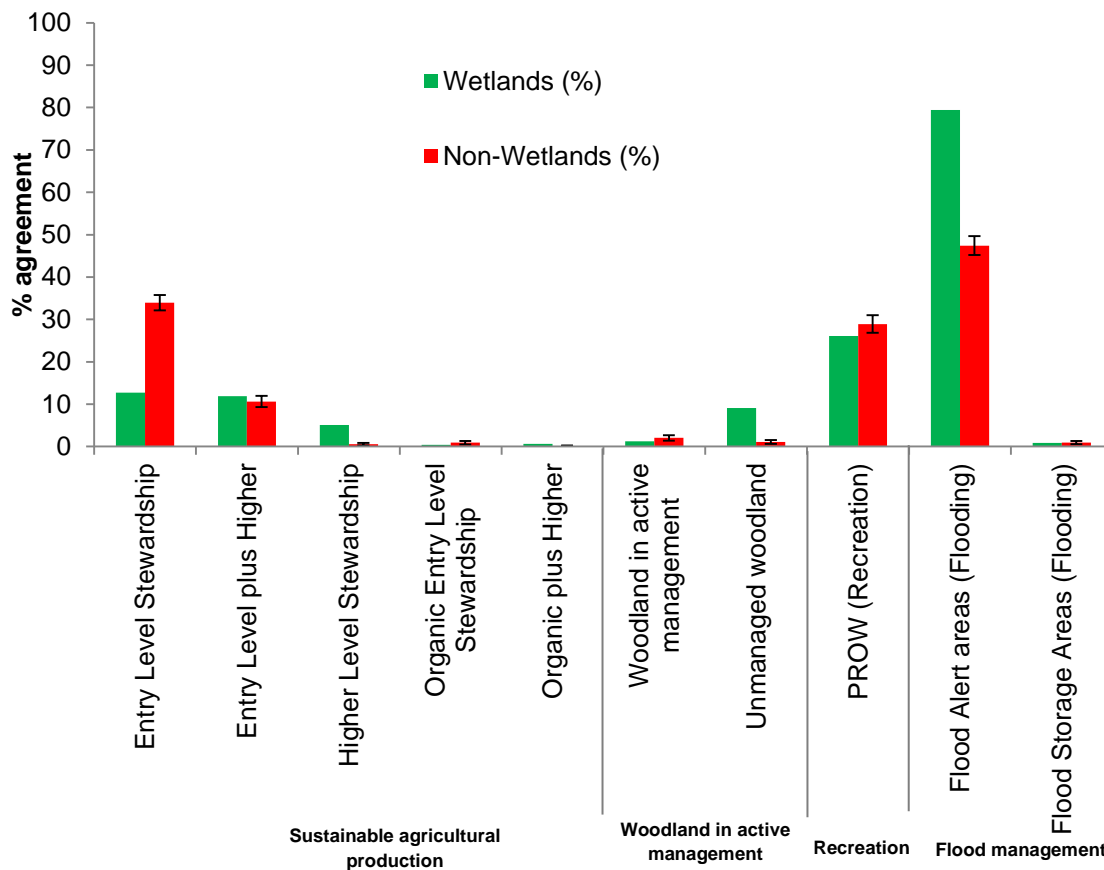


Figure 4.17: Binary ecosystem service metrics at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric. It is the percentage of grid squares that contain these metrics, and not the percentage of each metric that is found in the grid squares.

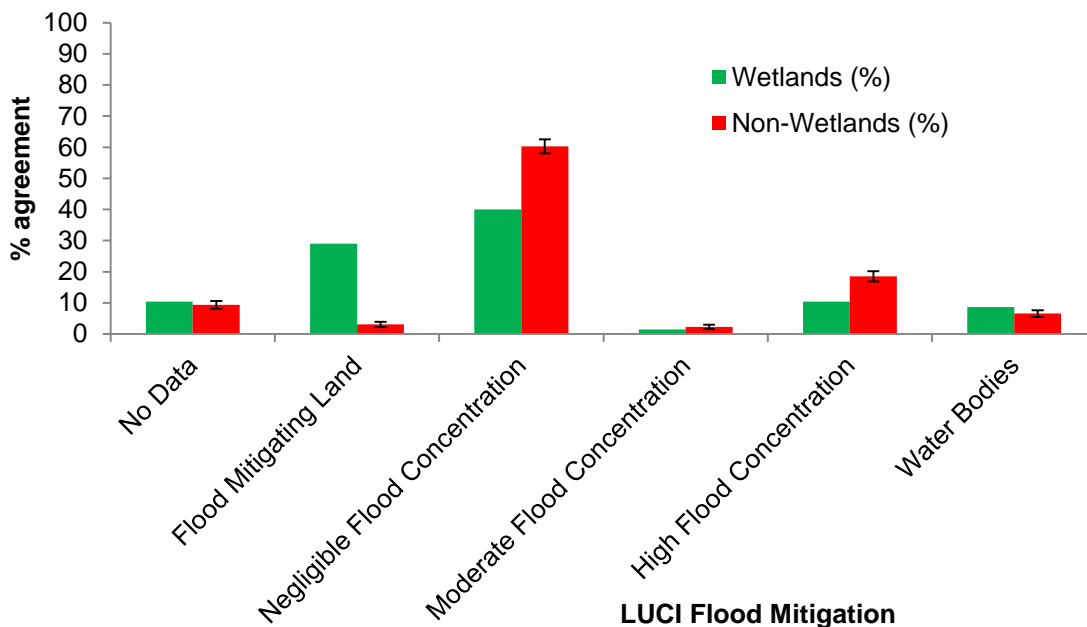


Figure 4.18: Results of the LUCI flood mitigation model at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

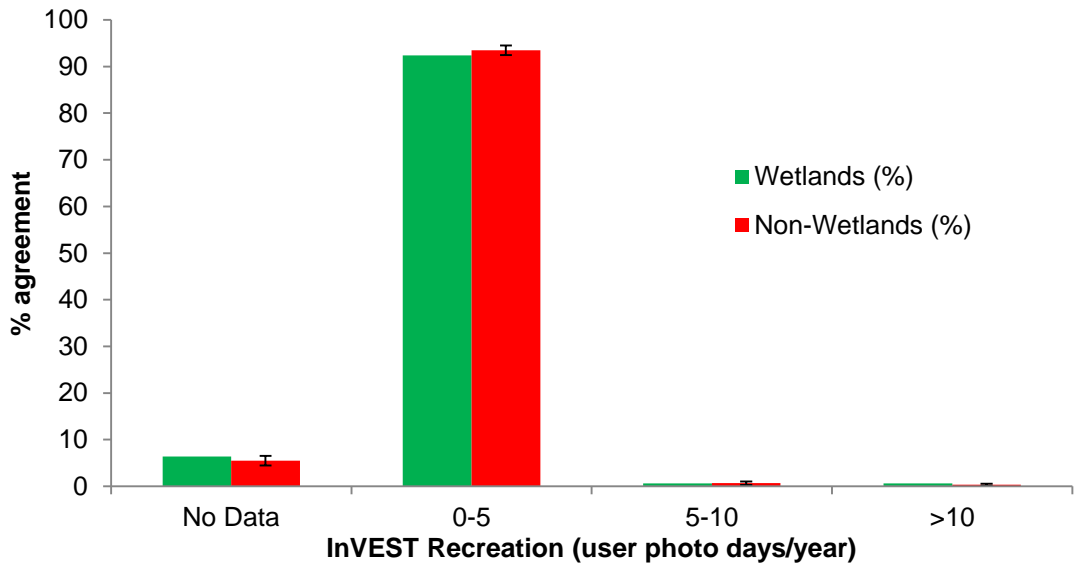


Figure 4.19: Results of the InVEST recreation model at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

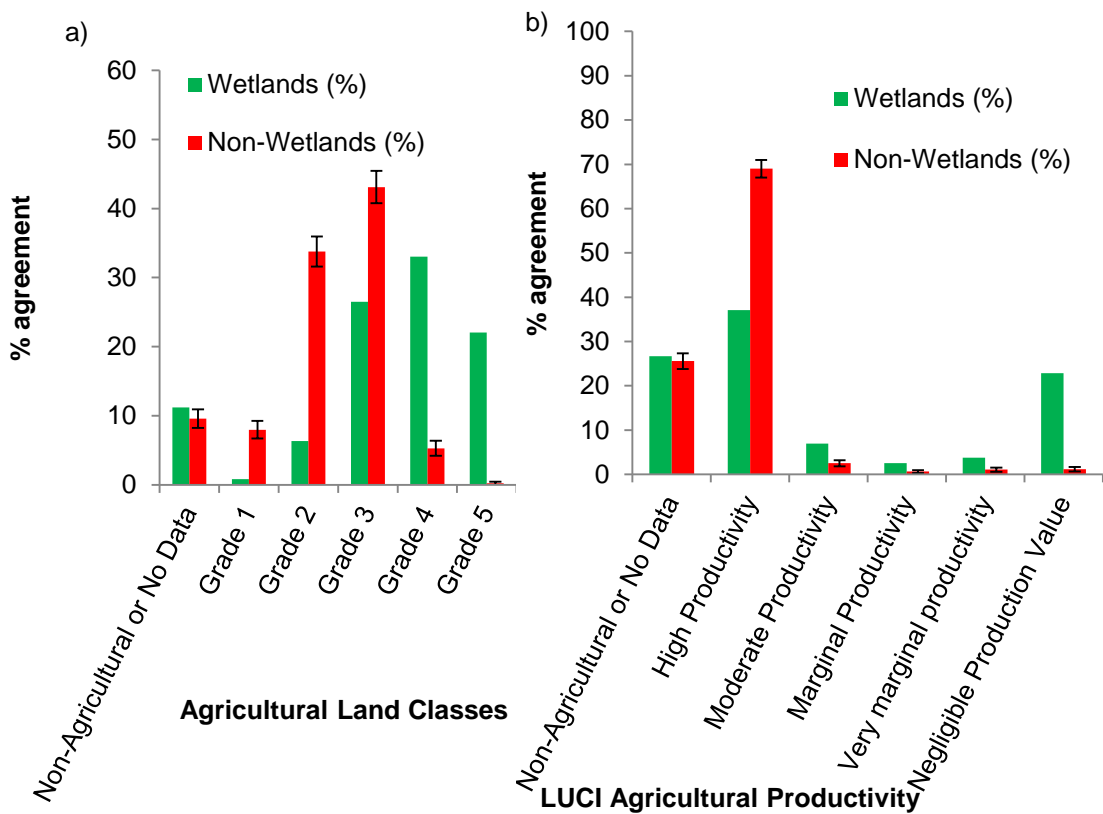


Figure 4.20: (a) Agricultural land classes and (b) results of the LUCI Agricultural Productivity model at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

The information on the delivery of the habitat quality ecosystem service is presented in Figure 4.21 and is in agreement with the 1000 m scale results that higher quality is delivered in current wetland patches. For the pollination ecosystem service (Figure 4.22), current wetland patches have a more even distribution of results across the different bins than the non-wetland patches at the 500 m scale, whilst the non-wetland patches have higher levels of pollination. Thus, it is difficult to draw any distinct patterns out, which is the same as with the 1000 m scale results. The two carbon storage metrics also show a similar trend at the 500 m scale than was observed at the 100 m scale. Both model outputs (see Figure 4.23) show increased carbon storage within current wetland patches. In general, the two scales produce fairly similar results in terms of ecosystem service delivery, a few of which will be useful for developing restoration options based on what is found in the current wetland patches in the landscape. There are, however some differences between the two scales, such as the pollination and InVEST recreation results, which could make a difference to how the landscape is interpreted and therefore lead forward into how restoration decisions are made about the landscape.

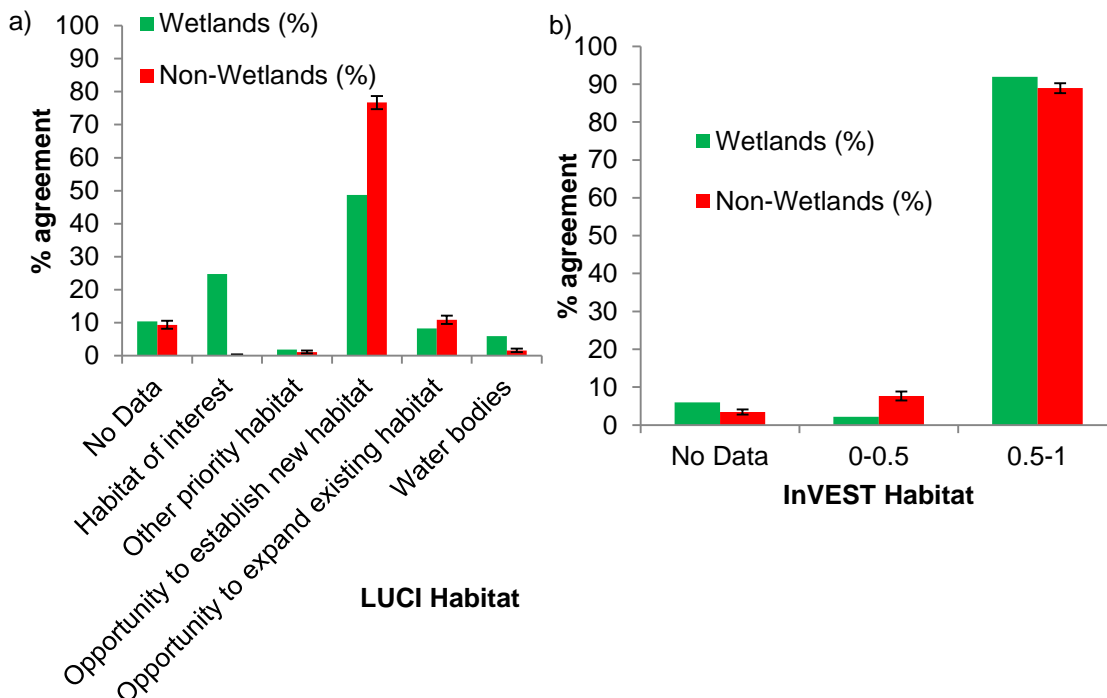


Figure 4.21: (a) Results of the LUCI habitat model and (b) the InVEST habitat model at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

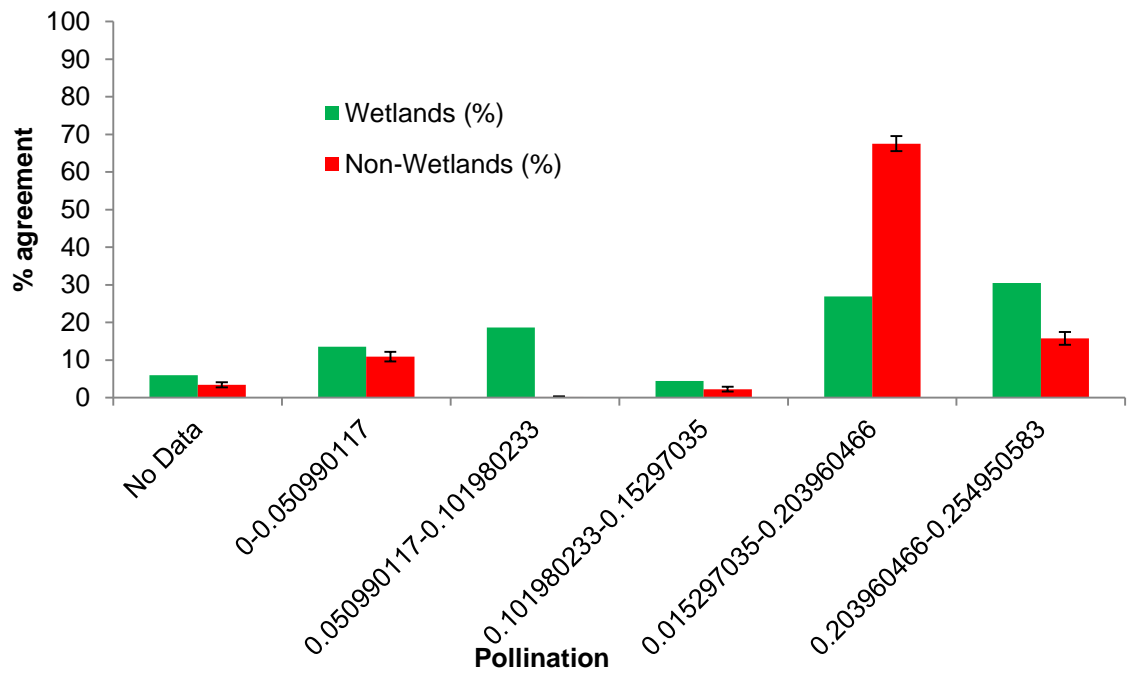


Figure 4.22: Results of the InVEST pollination model at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

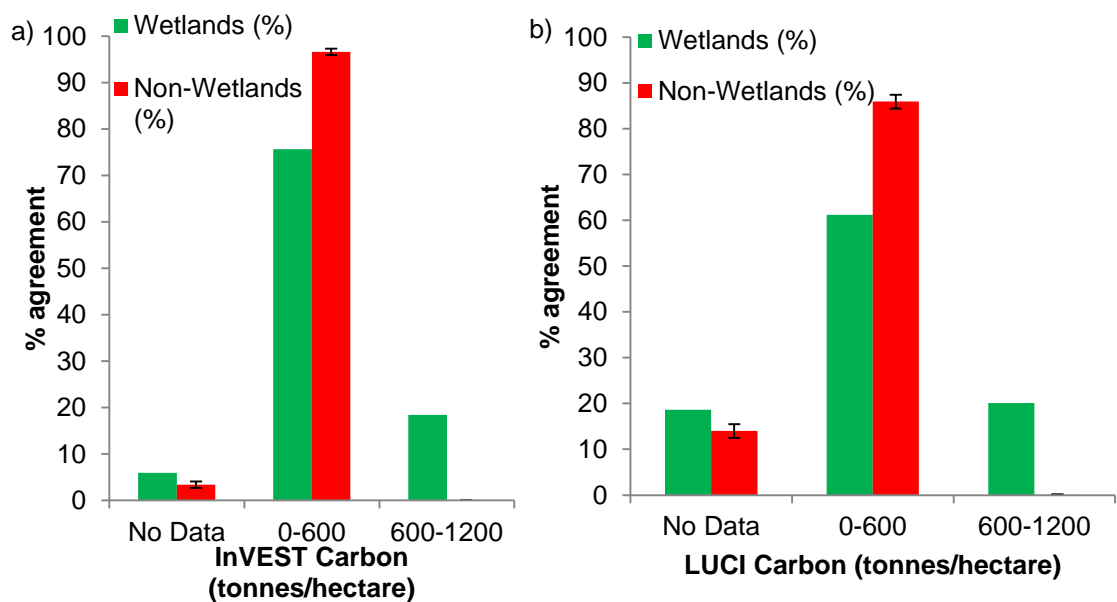


Figure 4.23: (a) Results of the InVEST carbon model and (b) the LUCI carbon model at the 500 m scale in relation to wetland patches in the HHL landscape. Wetlands (%) is the % agreement between each metric and the current wetland patches. Non-wetlands (%) is the % agreement between the non-wetland patches and each metric.

4.4 Discussion and conclusion- What do current wetland patches deliver in terms of biodiversity and ecosystem services?

A variety of potential associations became apparent when investigating which of the metrics were found in the current wetland patches, especially in comparison to the non-wetland values. Strong associations were found between some of the metrics of landscape context, biodiversity and ecosystem service and current wetland patches, and there was often considerable agreement between the two scales used. Using these results, a picture can be built up of what the current wetland patches are delivering in the HHL. The landscape context information provides background to the landscape, which is important for further understanding restoration potential and the delivery of biodiversity and ecosystem services (Botequilha Leitao and Ahern, 2002; Nelson et al., 2009; Jones et al., 2013). At both scales there were very similar results for what physical conditions with which wetlands occur.

Out of the land use metrics, the strongest association with the current wetland patches is the “bog” land use (which is found in 29% of current wetland patches and in none of the non-wetland patches), not a surprise considering the landscape context. The “broadleaved, yew and mixed woodland” and “saltwater” land uses also have fairly good results with the current wetland patches, compared to the non-wetland patches, but are not found often in this landscape. This is a historic wetland landscape, which has been converted to agricultural land (as shown by the high proportion of arable and horticulture, and improved grassland in non-wetland patches). It is not surprising therefore that the remaining woodlands are in wetland patches (as wetland drainage for agriculture would have been accompanied by deforestation) and that areas classified as saltwater are also considered “wetland”.

There are three soils that are mainly found, with “raised bog peat” occurring 62 times more in current wetland patches than non-wetlands. The current wetland patches also appear to have low levels of ammonia and DO at both scales, suggesting they are in better condition than the non-wetland patches. Finally, the topography metric results show that the current wetland patches are all found at very low levels, whereas some of the non-wetland patches are found much higher above sea level, which is not a surprise considering the landscape and the type

of wetlands. Overall, there are a series of results from landscape context that can be used to build up a description of the current wetland patches in the HHL, which will be a useful starting point for understanding biodiversity and ecosystem service delivery. Distinct from the rest of the landscape these areas are low lying, have lower levels of certain pollutants, and the main land use and soil types are bog related.

In terms of biodiversity, there are again many similar results at both scales as to what the current wetland patches deliver. In most cases the protected areas and BAP habitats have strong associations with the current wetland patches, which is not really surprising because Appendix A shows that some of these areas were designated for their wetland characteristics. Therefore, in the case of designated areas the current wetland patches deliver SSSIs, NNRs, SPAs, SCAs and Ramsar sites. This is as expected, because some of these may have contributed to the reason why particular areas were defined as wetlands, but they are still important associations to consider. The fact that the BAP habitats all have strong associations means that the current wetland patches deliver “grazing marsh”, “lowland fen”, “lowland raised bog” and “reed beds”. However, these strong associations are not that surprising considering the current wetland data derives from “statutory site data, and elements of national habitat inventories” (Hume et al., 2008). There are a few groups of species that are more prevalent within current wetland patches, although stronger associations tend to be seen at the 1000 m rather than the 500 m scale. The groups of species that the current wetland patches appear to support higher levels of are wetland habitat plants, bees and butterflies. There are also some focal species that the current wetland patches appear to support, including sedge.

Overall, the results show that the current wetland patches do deliver certain patterns of biodiversity in the HHL, which increases our understanding of the wetlands in the HHL and will therefore be useful for developing restoration options. These results can be used to build up a picture of biodiversity in the current wetland patches relative to the non-wetland patches. These areas tend to be protected by various designations, are within the wetland type BAP habitats and support higher levels of some of the groups and focal species. Biodiversity is in decline (Feld et al., 2009; Cui et al., 2012) and there are many unanswered

questions, especially how to deliver it in a landscape scale project. Wetlands are extremely important for beta and gamma diversity (UK National Ecosystem Assessment, 2011; Maltby and Acreman, 2011), and so by finding out what current wetland patches are delivering in terms of biodiversity it is useful for learning how to protect certain aspects of biodiversity to halt the loss and for developing restoration options.

For the ecosystem services results there were many similarities between the two scales, and there were a few services that showed strong associations with the current wetland patches. “Unmanaged woodland” is the first of these, which came out of the ecosystem service of looking at managed woodland in the HHL. The results show that the current wetland patches are not actually delivering the managed woodland service, but do have the potential to do so because there are unmanaged woodlands within the patches. Flood alert areas and the LUCI “flood mitigating land” also had strong associations, suggesting that the current wetland patches deliver areas where water is likely to collect and also areas to mitigate the effects of a flood, both of which is as expected with wetlands having the ability to store water. The results of the InVEST model at both scales are different and so this makes it difficult to interpret the delivery of the recreation ecosystem service, and there is also the issue brought up in the previous chapter about the transferability of the InVEST recreation model to this type of landscape compared to the North American landscape of large national parks. Both the InVEST and LUCI models were used to investigate habitat quality in the landscape, and both found higher levels in the current wetland patches than the non-wetland patches. Finally, both models show higher levels of carbon storage in the current wetland patches, which is an important ecosystem service for the HHL landscape to be delivering.

Overall, the results from the ecosystem services provide useful information for both understanding HHL wetlands and for developing future restoration options. They show that the current wetland patches do go some way to delivering some ecosystem services in the landscape, such as habitat quality and carbon storage, which is often discussed with wetlands in the literature (Tanner et al., 2013). It appears that wetlands deliver some types of services better than others, but that there are some services that are more general within the landscape whether

wetland or not, such as recreation. By understanding what ecosystem services are delivered by current wetland patches it helps to start to answer some of basic unanswered questions around ecosystem services such as “where are ecosystems producing benefits” (Bagstad et al., 2013b), what are the most important ecosystem services for a particular landscape (Peh et al., 2013) and can multiple services be delivered (Pan et al., 2013). A key point with these ecosystem service conclusions is that they come from a combination of models and field data, for which there are positives and negatives to both. Both provide really useful data for helping to understand the spatial distribution of ecosystem services, but the models do have assumptions built in and some of the species field records are affected by locational and observer bias. If this information was to be used for wetland restoration decision making these issues would need to be considered as part of the process.

There are a few differences between the results at the two scales. For example, at the 1000 m scale wetland habitat plants have a result of 22% with current wetland patches, whereas it is 8% at the 500 m scale. This could mean that if a restoration decision was based on this group of species then it could look very different if the decision was made using one or the other set of results, as is believed by Holt et al. (2015). It is not possible to say here what the optimal scale for this work and landscape is, that would require much more detailed testing, but that the scale chosen will make some difference to the results and therefore the interpretation (Holt et al., 2015). However, care was taken in choosing appropriate scales here, in accordance with the literature and original data format (Van Horssen et al., 1999; Engler et al., 2004), and so ideas of the delivery of biodiversity and ecosystem services can be built up. Perhaps an investigation looking at considerably different scales would be useful, with more drastic differences being observed, which could also address the question of what is are most appropriate scale of understanding the spatial distribution of biodiversity and ecosystem service. Therefore, to address the secondary objective of this chapter, it can be stated that the choice of scale could make a difference to how a landscape is interpreted, but to identify the correct scale would require further investigations.

While recognising the challenges and limitations of integrating many disparate data sets at the landscape scale in this way, the results here do represent the best and widest ranging data collection that has been achieved for the HHL landscape. Also, it provides a sufficiently robust picture of the relationships amongst landscape context, biodiversity and ecosystem services to allow meaningful interpretation, and improve our understanding of the HHL and other similar wetland environments. Previously it was identified that in order to aid landscape restoration, clear and rigorous science is needed (Maltby, 2010; Retford et al., 2012), which carried out in this study. On this basis, these results also provide a valuable foundation for developing restoration options in the next chapter, especially the method of using presence/absence grids, which is effective at dealing with data at different scales and making further interpretation and analysis easier (Fehmi and Bartolome, 2001; Gibson et al., 2004; Conlisk et al., 2009; Muller et al., 2011). Very few studies look at the biodiversity and ecosystem services together (Anderson et al., 2009), but by doing this analysis it will allow that to occur in the following chapter. The overall objective for this chapter was to understand the delivery of biodiversity and ecosystem services in the current wetland patches of the HHL landscape and this has been met in order to inform landscape scale wetland restoration.

Chapter Five- Identifying a range of potential locations for restoration for the HHL landscape based on biodiversity, ecosystem services and physical characteristics of the landscape

5.1 Introduction

The purpose of this chapter is to address the second part of Objective Two, which is as follows;

To use the information from Objective One and Objective Two(a) to:

(b) Identify a range of potential locations for restoration for the HHL landscape.

The objective of the previous chapter was to attempt to understand what the current wetland patches in the HHL landscape are delivering in terms of biodiversity and ecosystem services. Using this information, alongside the physical characteristics mapped in Chapter Two, the best places to restore wetlands in the landscape using the optimal biodiversity and ecosystem service delivery will be identified. This will provide information about how biodiversity, ecosystem services and physical characteristics could shape wetland restoration, as well addressing some important questions that will be identified from the literature. The results will provide useful information for decision making in particular for the HHL, alongside the work in the previous two chapters. In Chapter Seven the differences between using this information to drive restoration work as opposed to other human-driven factors will be investigated.

As identified previously, we cannot restore all previous wetlands. The classic approach for restoration is to use where wetlands were historically found as a basis to make decisions, but this is not possible at the landscape scale. Instead, the locations of particular patterns of biodiversity, ecosystem services and physical characteristics will be used to produce a range of options that identify locations for wetland restoration. It will include options targeted towards important ecosystem services in the landscape, such as flood mitigation, and patterns of biodiversity. Very few studies use scientific methods to investigate restoration (Holl et al., 2003), and it is important to try and find better methods for allocating restoration effort.

The idea of landscape scale working is important to this thesis, and so the range of options will cover the whole of the HHL landscape, as with analysis in the previous two chapters. In Chapter One (Section 1.2) it was identified that previously most conservation work was at a site by site scale, but has proven insufficient (MEA, 2005; Verburg et al., 2016). Therefore, working at a landscape scale has been identified as a way to overcome this, but we currently do not really understand how to do landscape scale wetland restoration (Wu, 2006; McKenzie et al., 2013). Therefore, working at this scale here will help to inform how landscape scale restoration could be done.

In Chapters One and Three the importance of biodiversity and ecosystem services has been seen (e.g. MEA, 2005; Maltby and Acreman, 2011, UK National Ecosystem Assessment, 2011), and that more needs to be done to protect both of them. We still do not really know how to do this, again especially at this landscape scale, which may do more to protect them than has previously occurred. For biodiversity, the spatial distribution across a landscape needs to be better understood. With the concept of ecosystem services, we again still do not understand the spatial distribution of services (Wu, 2013) and how to deal with the delivery of multiple ecosystem services (Pan et al., 2013). Also, we do not really understand the trade-offs and synergies between different services (Tallis and Polasky, 2009; HHL Partnership, 2011; Rey Benayas and Bullock; 2012 Pan et al., 2013). By identifying options for wetland restoration, it will help also to answer these important questions.

Key to effective delivery of biodiversity and ecosystem services across a landscape is understanding how the two may interact. Anderson et al. (2009) identified the scarcity of studies looking at the spatial relationship between biodiversity and ecosystem service delivery. Fisher et al. (2008) and Eastwood et al. (2016) found some studies which show a good association, some which show a poor association and some no association at all, but Bullock et al. (2011) cautioned that there was insufficient evidence to be confident in any relationship. In addition, Rey Benayas and Bullock (2012) argue that biodiversity often loses out to ecosystem services, as it has different requirements in restoration and now is sometimes seen as being of lesser importance. Therefore, the suggested locations for wetland restoration through biodiversity need to be compared to

those for ecosystem services, because both are vital for achieving restoration of wetland environments, and it will help answer a key question.

As well as considering biodiversity and ecosystem services, it is also important to investigate the physical characteristics to see how they define restoration locations. In the literature the importance of understanding the context of a landscape is discussed (Botequilha Leitao and Ahern, 2002; Jones et al., 2013), and management decisions are frequently made based on the spatial distribution of physical characteristics (Casado-Arzuaga et al., 2013). This approach was taken as part of Wetland Vision project (Hume et al., 2008), which generated maps that showed potential for wetlands based on ecological and historic environment criteria (Hume et al., 2008), and will therefore be compared to the outcomes of this chapter. Conservation at a landscape scale is about trying to deliver a multi-functional landscape, and so it will also be important to see how restoration based on physical characteristics compares to the biodiversity and ecosystem services options.

The idea of ecological networks will be a key consideration with each of the location options produced, as it is an important approach for landscape scale work. The idea is to create mosaic landscapes, with important habitats connected together, whilst the rest of the landscape has other land uses (Lawton et al., 2010). This concept is particularly appropriate for the HHL, as the restoration of the whole landscape is not feasible. A mosaic landscape is created with core areas connected together through corridors and stepping stones, to make the landscape “more, bigger, better, joined” (Lawton et al., 2010). By having a pattern of connected areas across a landscape, it will allow for the movement of organisms, matter and energy, for breeding, migration and climate change adaptation (Finlayson, 1999; Peacock, 2003; Catchpole, 2006; The Wildlife Trusts, 2007; Hume et al., 2008). Therefore, this concept will be considered with the wetland restoration options produced.

The purpose of this chapter is to explore ways of using the spatial arrangement of existing biodiversity, ecosystem services and physical characteristics to identify a range of potential wetland restoration options for the HHL landscape, by looking at the areas that are not currently wetlands. These options will inform

the ideal areas for restoration under the three factors, individually and combined, and answer key questions identified here from the literature, which will overall answer the objective of the chapter. These questions are listed below, with an accompanying justification, before the methods, results and discussion.

The analysis seeks to answer the following questions:

- What locations are identified as optimal for individual biodiversity metrics?

It is important to think about specific aspects of biodiversity across the HHL, and how targeting these might look in the landscape in order to produce a restoration option. There are many aspects of biodiversity that could be used as the basis for restoration decisions, but it is not feasible to investigate them all. Here two metrics will be used, chosen based on the data availability and quality, how they were delivered by current wetland patches in the previous chapter and how they could realistically be used to inform decisions. The first is wetland habitat plants, chosen because it was identified in Chapter Four as being delivered by current wetland patches, there is a sufficiently good dataset and plants are a realistic basis for making some restoration decisions on within the HHL because they can be an important part of wetland environments. Breeding wetland birds was selected because it was also identified in previous chapter as being delivered by current wetland patches and because birds are iconic in conservation, they receive considerable protection and are favoured by the public. Alternative metrics from the previous chapter were not used because the data quality/amount was not as good, they were not quite as relevant to wetland environments or they did not cover as many species.

- What locations are identified as optimal for delivery of individual ecosystem services?

It is important to target restoration locations based on individual ecosystem services because there are still many unknowns about the distribution and delivery of services (Wu, 2013), and there is increasing importance put on the use of the concept as a conservation approach (MEA, 2005; Fisher et al., 2008; Maltby and Acreman, 2011; Norris, 2011; UK National Ecosystem Assessment, 2011). Again, two will be chosen, as it is not feasible to do all, based on their importance to the landscape and their delivery in the current wetland patches as identified in Chapter Four. Carbon storage was selected because it is a key ecosystem service for the HHL, and for wetlands in general (Tanner et al.,

2013). Flood mitigation was chosen because of the threat to the low lying landscape from flooding events, especially in the face of future climate change, making it a key regulating service to investigate (Peacock, 2003; Cook and Hauer, 2007; Maltby, 2010; HHL Partnership, 2011). The other services were not chosen because of problems with the models, as identified in the previous chapter or because they were not quite as relevant for wetland environments.

- What locations are identified as optimal for biodiversity?

Looking at locations for restoration based on a wider range of biodiversity metrics will mean that more will be understood about the spatial distribution of biodiversity in the HHL and in general, which will then be useful for moving forward with restoration at the desired landscape scale (Otte et al., 2007). A wide range of the biodiversity grid system metrics from the previous chapter will be used, based on their delivery in current wetland patches in the previous chapter, as will be discussed in the relevant section.

- What locations are identified as optimal for delivering multiple ecosystem services?

As well as unknowns about the delivery of individual ecosystem services, there are still many unknowns about the delivery of multiple ecosystem services (Pan et al., 2013), which by investigating can provide a restoration option for the HHL. Several services will be considered, based on their identified delivery in current wetland patches in Chapter Four, as again will be discussed in the relevant section.

- Can both biodiversity and ecosystem services be delivered in the same places?

This is one of the key questions to answer, because there are very few studies that do (Anderson et al., 2009) and the ones that exist have varying answers (Fisher et al., 2008; Rey Benayas and Bullock, 2012; Eastwood et al., 2016). This will be undertaken using the results from the previous two questions.

- What locations have the appropriate physical characteristics for wetland restoration and do they coincide with biodiversity and ecosystem service provision?

The previous questions have all been based on using what already exists in the landscape to identify locations for wetland restoration. It is also useful to consider where to restore based on where there may be potential in the landscape. There are many ways that potential could be looked at, but here

the physical characteristics of the current wetland patches are used to look where else might be appropriate for restoration, as was done with Wetland Vision (Hume et al., 2008). By looking at what physical characteristics were identified in the previous chapter as having a good association with current wetland patches that are typically associated with wetland environments, and then where the location of this configuration of characteristics exists, will be used for another restoration option. The use of particular physical characteristics will be discussed in the relevant section. It is also important to consider where this configuration of physical characteristics exists in relation to the biodiversity and ecosystem service options, and to that of Wetland Vision.

- What do these restoration options mean for managing this landscape?

It is important to think about these results in relation to the configuration of the landscape for management purposes, because this work is attempting to provide information for those working on the ground.

In the overall discussion it is important to consider the following question in reference to the results: Can a multi-functional landscape be delivered?, as this is the desired state for the restoration of wetland environments.

5.2 General methods

The methods build on those used in Chapter Four, utilising the data originally mapped in Chapter Three and taking a similar approach to other studies in the literature (e.g. Pereira and Itami, 1991; Van Horssen et al., 1999). It expands on the methods used throughout Chapter Four of converting the metrics for biodiversity, ecosystem services and physical characteristics into a grid square system to allow for comparison when the data sources were in different formats and scales (Chapter Four, Section 4.2). When the word metric is used it means those originally mapped in Chapter Three and then converted into a grid system with presence/absence in Chapter Four. The range of potential options for restoration will also be presented using this grid system. Only the 500 m scale results from the previous chapter will be used here, because 500 m is the more likely scale that restoration decisions would be made at, as they tend to occur at smaller scales. In order to compare some of the map outputs and answer some of the questions identified Spearman's rank correlation will be used, as has been

carried out with other studies (e.g. Anderson et al., 2009; Holt et al., 2015). The significance for this test cannot be used because the data set is spatially structured, with non-independence, and is very large.

The options presented in this chapter are just a few potential suggestions for the HHL landscape, but they were chosen to address some important questions emerging from the literature (see Section 5.1), and so they are the most important to investigate. For more information on the biodiversity and ecosystem services metrics and the categories used for the following analysis please refer to Chapters Three and Four. The specific methods to answer each question are covered in the following section, with the results.

5.3 Methods and results

5.3.1 What locations are identified as optimal for individual biodiversity metrics?

5.3.1.1 Wetland habitat plants

In previous chapters this metric was considered using just a presence/absence grid, but here diversity and abundance is built in to give categories to the suggested locations for restoration. Each square that contains the previous presence/absence data is given a value between one and five as with the scale displayed in Table 5.1 (along with the summary statistics for each category). The results are displayed in Figure 5.1, shown against the current wetland patches in the landscape. Figure 5.1 shows that the locations identified for restoration under this option appear to be quite spatially diverse across the landscape, with very few squares suggested. However, there are quite a few squares suggested around and below the three main wetland patches in the centre of the landscape.

Table 5.1: Breakdown of the scores for wetland habitat plants in the wetland habitat plants restoration option, of which “4” and “5” provide more appropriate suggestions for restoration because of high levels of diversity and abundance.

1 - One occurrence of one species	
2 - One species but multiple occurrences	
Average no. of occurrences	2.35
Range	2-5
3 - Multiple species but only one occurrence of each	
Average no. of species	2.63
Range	2-8
4 - Some species with one occurrence and some with multiple occurrences	
Average no. of species	5.21
Range of species	2-22
Average no. of records	14.29
Range of records	3-76
Average no. of occurrences	2.79
Range of occurrences	1-59
Average no. of multiple species	2.79
Range of no. multiple species	1-16
Average no. of records for the multiple species	4.28
Range of no. of records for the multiple species	2-59
Average no. of single species	2.33
Range of no. of single species	1-8
5 - Multiple species each with multiple occurrences	
Average no. of species	2.36
Range of species	2-4
Average no. of records	92.5
Range of records	4-275
Average no. of occurrences	39.24
Range of occurrences	2-229

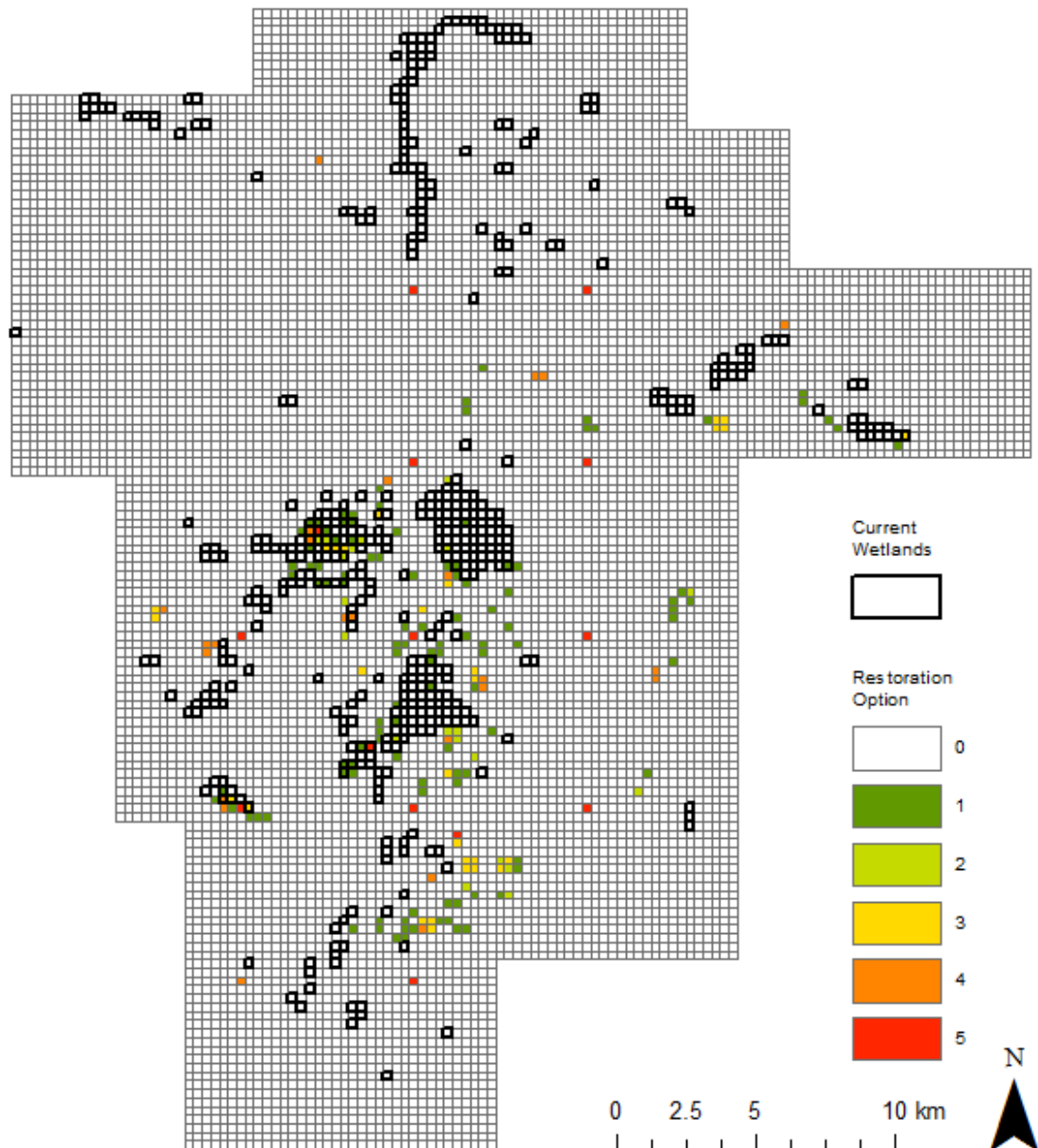


Figure 5.1: Wetland habitats plants restoration option, where a higher score indicates greater abundance and/or diversity in the taxa, and therefore a stronger suggestion for restoration.

Looking in more detail at the individual categories in Table 5.1 is important, because certain categories will be more appropriate restoration suggestions, especially “4” and “5”. Of the squares that are classed as “5” there is a lot of variation in terms of the number of species and the number of occurrences for each. For occurrences, the average number per species is around 39, which is a relatively appropriate number for considering where to decide to restore new core areas. Although “4” is not as strong, it still suggests the possibility of good habitat already in this area. Areas classed as “1” or “2” have low numbers of records and species in general, and consequently are not such strong suggestions for

restoration. The higher categories of “4” and “5” tend to be found close to or adjacent to the current wetland patches, so suggesting expansion of the core areas. There are also a couple of squares with “5” in the upper central part of the landscape, which are not close to other biodiversity scores or current wetland patches, and so these would be less appropriate for a focus for restoration as they might become or already be isolated. Overall targeting restoration using existing wetland plant distributions would mainly mean new core areas around the two NNRs, with a little connectivity between these and other current wetland patches.

5.3.1.2 Breeding wetland birds

As with the previous option this is created by giving each of the 500 m squares with presence/absence data in, a value between one and five (see Table 5.2). The restoration map under this option is presented in Figure 5.2, again with current wetland patches. Figure 5.2 shows an example of where data could be an issue when trying to do landscape scale conservation work, as the small amount of data here is suggesting very little restoration, caused by the lack of data issue rather than the results. In Figure 5.2 from the very few squares suggested for restoration, most are classed as “4” or “5”, which are the most appropriate categories for suggesting restoration, because of the multiple species and occurrences. In the case of squares of “5”, the average number of species is 18.5 and the average number of occurrences per species is 63.31 (as seen in Table 5.2), with some found adjacent or close to current wetland patches. Overall, if this option was used to restore the landscape then there would be very little restoration, but with some expansion of core areas.

Table 5.2: Breakdown of the scores for breeding wetland birds in the breeding wetland birds restoration option, of which “4” and “5” provide more appropriate suggestions for restoration because of high levels of diversity and abundance.

1 - One occurrence of one species	
2 - One species but multiple occurrences	
3 - Multiple species but only one occurrence of each	
Average no. of species	3
Range	2-5
4 - Some species with one occurrence and some with multiple occurrences	
Average no. of species	12.6
Range of species	3-22
Average no. of records	390.6
Range of records	5-2102
Average no. of occurrences	31
Range of occurrences	1-357
Average no. of multiple species	9.07
Range of no. multiple species	1-21
Average no. of records for the multiple species	42.39
Range of no. of records for the multiple species	2-357
Average no. of single species	3.47
Range of no. of single species	1-8
5 - Multiple species each with multiple occurrences	
Average no. of species	18.5
Range of species	12-22
Average no. of records	1171.2
Range of records	117-3356
Average no. of occurrences	63.31
Range of occurrences	2-1869

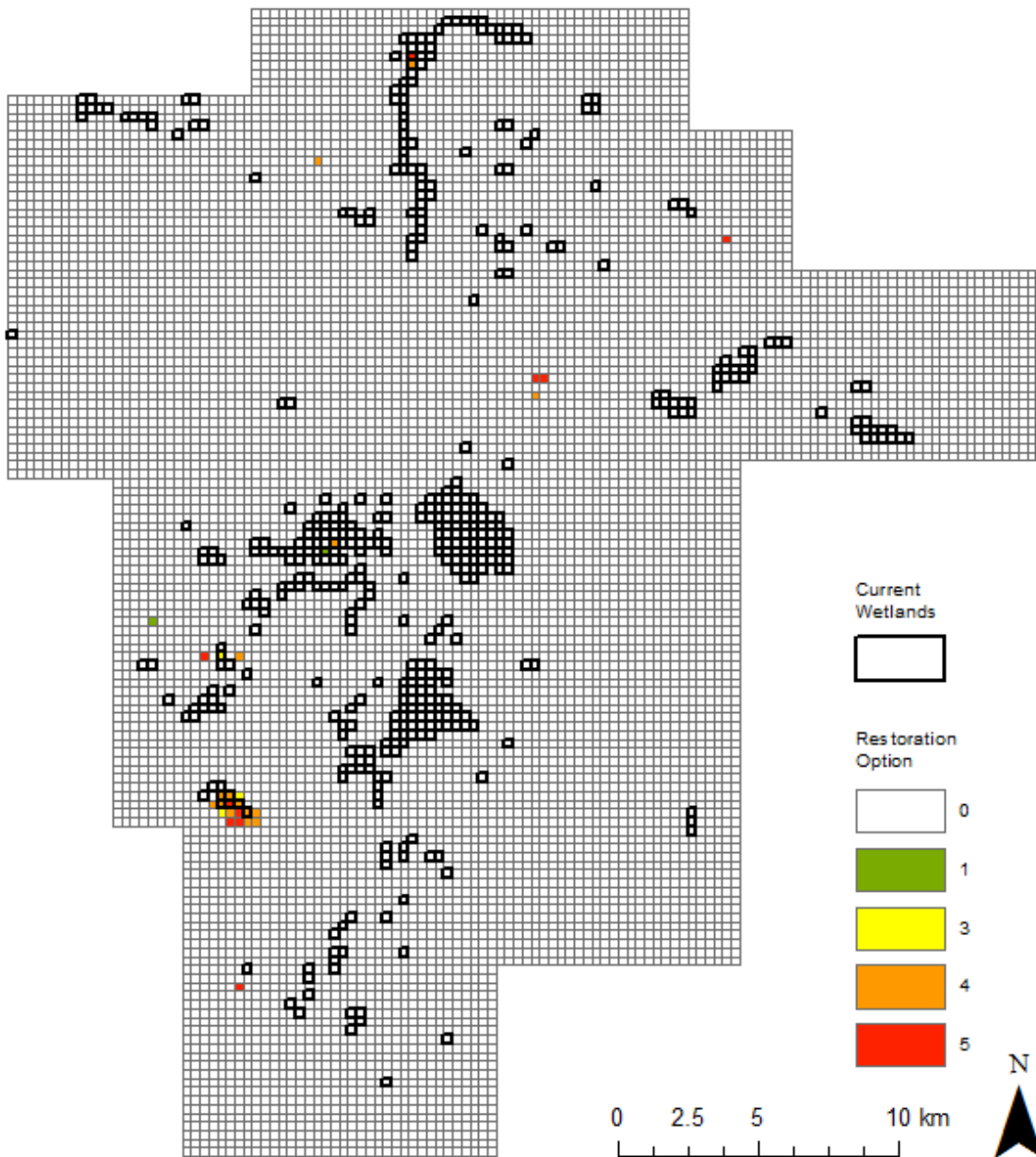


Figure 5.2: Breeding wetland birds restoration option, where a higher score indicates greater abundance and/or diversity in the taxa, and therefore a stronger suggestion for restoration.

5.3.2 What locations are identified as optimal for delivery of individual ecosystem services?

5.3.2.1 Carbon storage

This option for identifying locations for wetland restoration was produced using the model results from LUCI and InVEST (the inputs of which will have an influence and are discussed in Chapter Three). In the previous chapter the results from both models were put into three categories; no data, less than 600 tonnes per hectare and 600-1200 per hectare. For this restoration option the top category

is used, to look at optimal delivery of this service. If both models provided this category for a square then it was given a value of 2, if it was just one model then it was given a value of 1, and otherwise it was 0, creating different levels of service delivery. The restoration idea behind this option is that by carrying out restoration where there is carbon storage currently it will protect the high level of carbon storage and potentially improve it, as well as restoring wetlands, because carbon storage is often related to wetland environments.

Figure 5.3 shows the carbon storage option, which has very few suggested locations for wetland restoration outside current wetland patches. As this carbon storage option is made out of the top category defined from the previous chapter, these results indicate that the best delivery of this service is found within wetland environments. Only one of the squares outside the current wetland patches is given by both models, whilst the other four are only given by one or the other. If the carbon storage option was used for making decisions about where to restore wetlands, there would be very little new restoration work, but this option advocates a focus on one large area, rather than multiple.

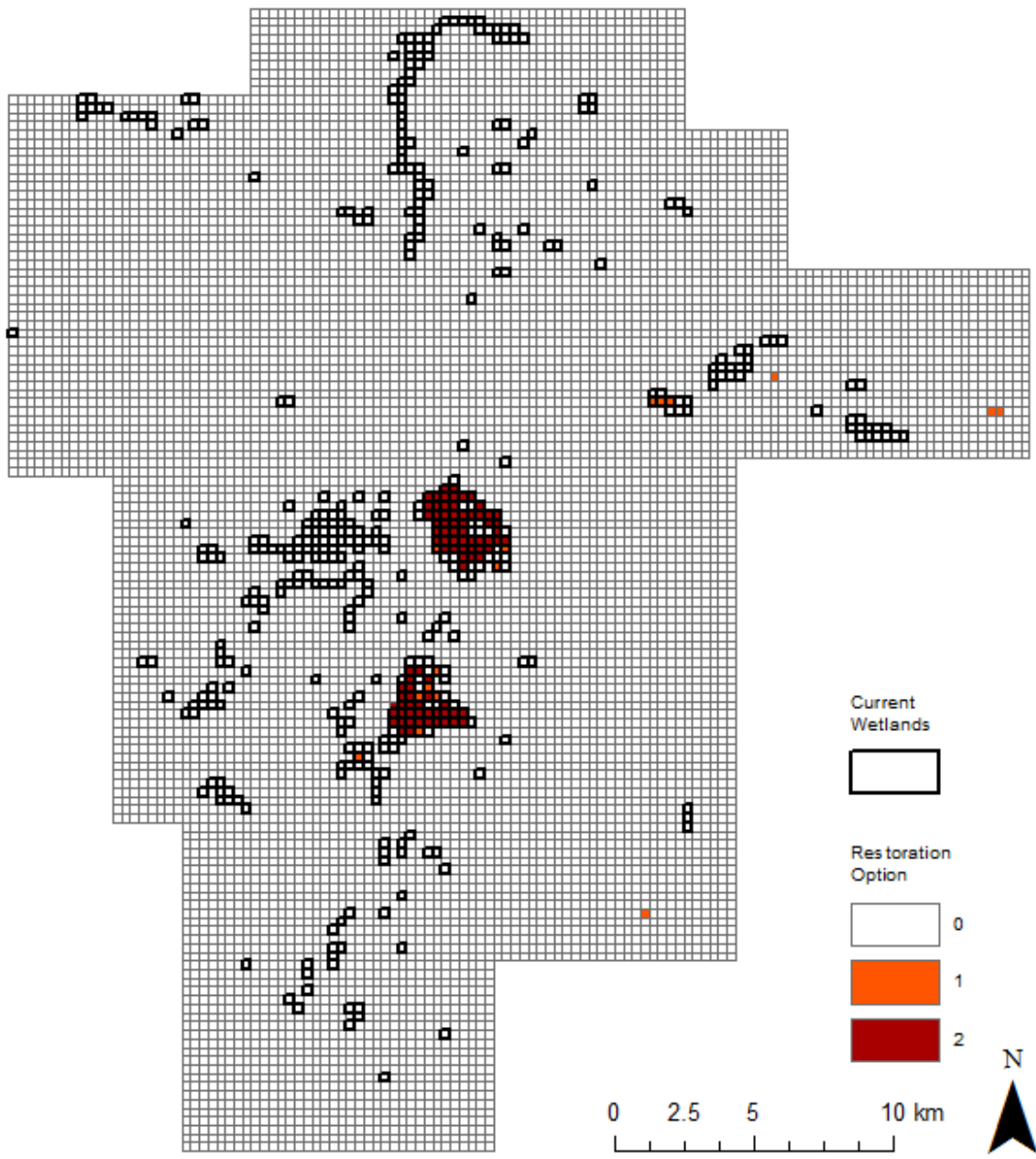


Figure 5.3: Carbon storage restoration option, where a higher score indicates a higher delivery of the service, and therefore a stronger suggestion for restoration.

5.3.2.2 Flood mitigation

The flood mitigation option was created using the LUCI Flood Mitigation model results (the inputs for which are covered in Chapter Three) and the areas that the Environment Agency identified as flood storage areas to suggest locations for restoration. According to the Environment Agency a flood storage area is “those areas that act as a balancing reservoir, storage basin or balancing pond...their purpose is to attenuate an incoming flood peak to a flow level that can be accepted by the downstream channel” (Data.gov.uk, no date). Therefore, these

areas could provide good locations for wetland restoration because of their physical characteristics. The flood storage areas were in the presence/absence grid systems from the previous chapter. The LUCI results were in a categorical grid system format from the previous chapter, and here only the results in the category called “flood mitigating land” are included. The results from the two datasets were then combined, where 1 indicates that one dataset was found, and 2 where two datasets were found, again creating two levels of delivery.

Figure 5.4 shows the 500 m flood storage option for locations of restoration, with the current wetland patches. There are no squares where both datasets were found. There are a fair amount of squares suggested for restoration, many of which are in or around the two NNRs, with some river networks also obvious across the landscape. There is an interesting area in the bottom right corner where there is only a small amount of current wetland, but with many areas suggested for restoration. This option develops ideas of some small expansion of current core areas and some connectivity. Overall, the flood mitigation option shows some restoration options for the landscape, which advocates both one big area and networks.

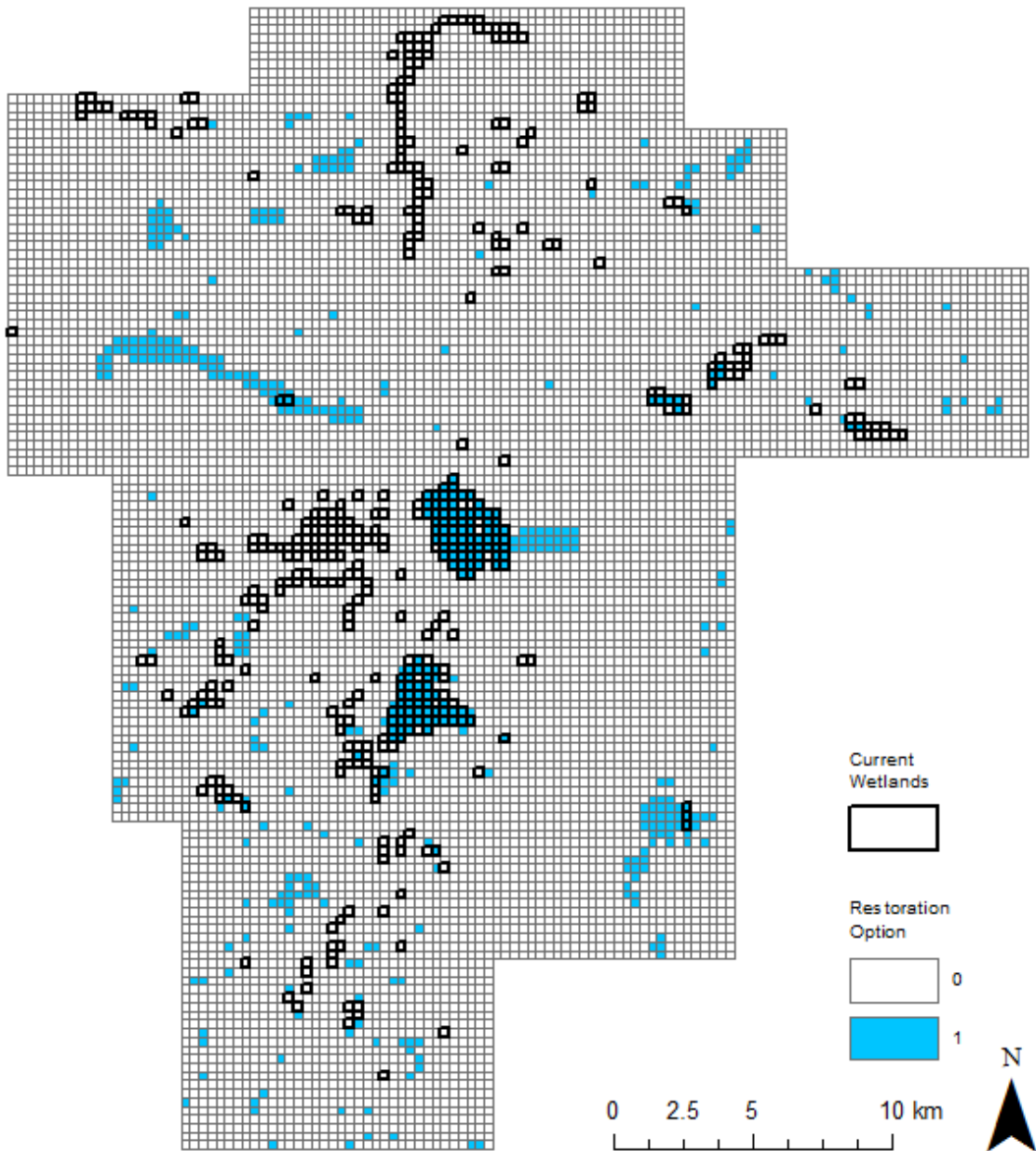


Figure 5.4: Flood mitigation restoration option, where a higher score indicates a higher delivery of the service, and therefore a stronger suggestion for restoration.

5.3.3 What locations are identified as optimal for biodiversity?

This option seeks to understand the spatial distribution of biodiversity across the HHL and use this to suggest locations for wetland restoration, to enhance and protect existing biodiversity resources. The data used within this option is obtained from the biodiversity data that was converted into the presence/absence grid system (see Chapter Four section 4.2). A sub-set of metrics is used, focussing on those that showed the strongest association with wetlands, as were

identified in Chapter Four. Metrics that were at least four times more likely to occur in current wetland patches than non-wetland patches were selected and included both groups (widespread) and individual (focal) species: breeding wetland birds, wintering water birds, wetland habitat plants, butterflies, bittern, crane, sphagnum, plover, banded demoiselle dragonfly, newts and sedge. These were given the same scoring system as the biodiversity options in Section 5.3.1, between one and five. For the individual species these could only be scored either “1” or “2” under the same system. The scores for each dataset were then combined together to create Figure 5.5 (with current wetland patches). Figure 5.6 shows the number of metrics within each square that were “3”, “4” or “5” for groups of species and “2” for individual species, because these would be more appropriate categories for restoration. Some of the results here could be function of the spatial resolution or inaccuracies discussed previously of the species records, or the grid system method used (see Chapter Four Section 4.2), which are potential limitations of this method.

In Figure 5.5 the maximum score that one square could have is 34, but the highest actual score by a square here is 29, and the majority of squares have a value of 0. Of the few squares suggested for restoration (those with any value) a large number of these are in the southern half of the landscape, and most have a low score. Some current wetland patches have no score at all, which raises an interesting question. The very few grid squares that contain a higher score are close or adjacent to current wetland patches, so building on core areas. The distribution in Figure 5.6 is fairly similar, but with fewer locations suggested. The maximum possible number for this figure is 14, but the highest recorded is nine, which occurs in two squares. Spearman's rank correlation of the values in the grid squares was used to assess the correspondence between the two distributions, and gave a value of 0.747, which is a fairly strong positive relationship. As these are relatively similar and the Figure 5.6 option covers less current wetland patches, Figure 5.5 will be used for further investigation of restoration suggestions.

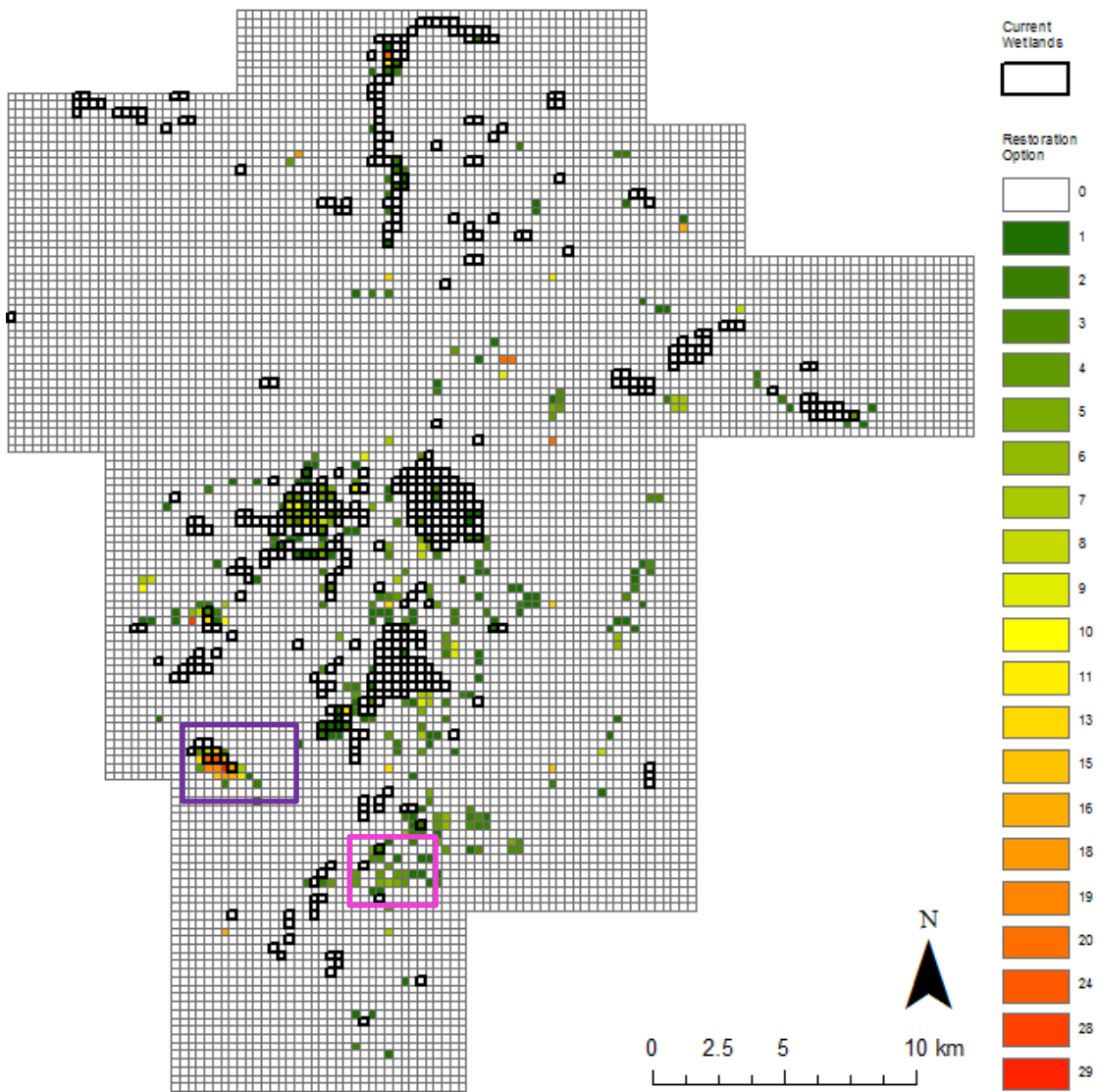


Figure 5.5: Biodiversity targeted restoration option, where a higher score indicates greater diversity and/or abundance of the taxa, and therefore a stronger suggestion for restoration.

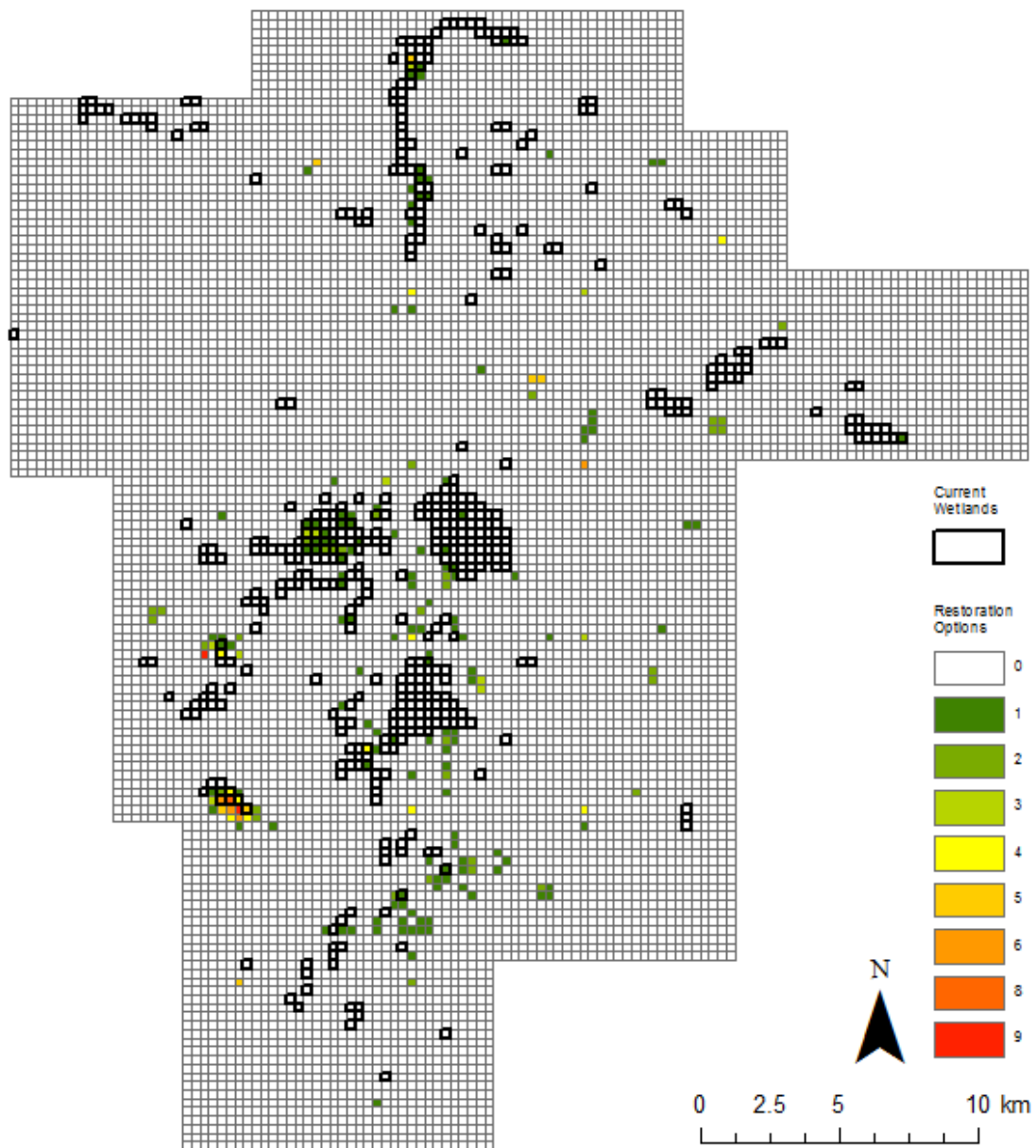


Figure 5.6: Biodiversity targeted restoration option, but with the scores indicating the number of metrics in each square over the limit (identified in the text), which are the strongest suggestions for restoration.

By looking at particular areas in the landscape, it is possible to see what makes up the areas of higher scores in Figure 5.5, i.e. whether it is the presence of many species, or many occurrences of a few species. The two chosen spots are presented in Figure 5.7 and Figure 5.8, but their locations within the landscape are displayed in Figure 5.5. For more context specific information see Chapter Two Figures 2.1 and 2.11. The hotspot in Figure 5.7 was chosen as an example of a situation that might suggest expansion of one of the current wetland patches, and the breakdown of these scores is seen in Table 5.3(a). The square with the highest value of 29 is 1902. This score features nearly all of the species included,

with many of them at the highest category. The high score, with the fact that it is next door to a current wetland square, means that this square would probably be a suitable suggestion for restoration. There are also some other squares with relatively high values, that could be good suggestions for restoration and expansion out from this current wetland patch.

The hotspot in Figure 5.8 was chosen to consider an example where this restoration option appears to increase connectivity between current wetland patches. However, the squares featured for restoration in this spot do not have very high values. Many of these squares do not include a wide range of biodiversity metrics, as seen in Table 5.3(b), although a couple do have good scores for single group metrics. These squares do not have such a strong argument for restoration purely in biodiversity terms, but might be useful as stepping stones between core sites if they were developed further. Overall, the biodiversity option would mean mostly further expansion of the current wetland patches found in the central part of the landscape, which is where most of the higher biodiversity scores are found across the HHL.



Figure 5.7: Hotspot focus for the purple box in Figure 5.5, where a higher score indicates greater diversity and/or abundance of the taxa, and so a stronger suggestion for restoration.



Figure 5.8: Hotspot focus for the pink box in Figure 5.5, where a higher score indicates greater diversity and/or abundance of the taxa, and therefore a stronger suggestion for restoration.

Table 5.3: Breakdown of the scores from Figures 5.7 and 5.8, of which “4” and “5” provide more appropriate suggestions for restoration because of high levels of diversity and abundance.

Figure 5.7	Wetland Habitat Plants	Breeding Wetland Birds	Wintering Water Birds	Butterflies	Bittern	Crane	Sphagnum	Plover	Banded Demoiselle Dragonflies	Newt	Sedge	Total
1758	0	0	0	0	0	0	0	0	2	0	0	2
1762	0	0	0	0	0	0	0	0	2	0	0	2
1829	0	5	4	4	1	0	0	2	0	0	0	16
1830	0	5	4	4	2	2	0	2	0	0	0	19
1831	1	4	4	4	1	0	0	2	0	0	0	16
1832	1	4	4	0	0	0	0	0	0	0	0	9
1833	1	0	0	0	0	0	0	0	0	0	0	1
1899	0	0	0	4	0	0	0	0	0	0	0	4
1900	4	3	4	4	0	0	0	1	2	0	0	18
1901	1	4	4	4	2	0	0	2	2	0	0	19
1902	5	5	4	4	2	1	2	2	2	0	2	29
1904	0	4	3	0	0	0	0	0	0	0	0	7
1971	1	4	4	4	0	0	0	0	0	0	0	13
2043	0	0	0	4	0	0	0	0	0	0	0	4
2046	0	3	0	0	0	0	0	0	0	0	0	3

Figure 5.8	Wetland Habitat Plants	Breeding Wetland Birds	Wintering Water Birds	Butterflies	Bittern	Crane	Sphagnum	Plover	Banded Demoiselle Dragonflies	Newt	Sedge	Total
944	1	0	0	0	0	0	0	0	0	0	0	1
945	1	0	0	0	0	0	0	0	0	0	0	1
1006	1	0	0	3	0	0	0	0	0	0	0	4
1009	1	0	0	3	0	0	0	0	0	0	0	4
1010	1	0	0	4	0	0	0	0	1	0	0	6
1011	4	0	0	0	0	0	0	0	0	0	1	5
1012	3	0	0	0	0	0	0	0	0	0	1	4
1070	1	0	0	3	0	0	0	0	0	0	0	4
1073	1	0	0	3	0	0	0	0	0	0	0	4
1075	3	0	0	0	0	0	0	0	0	0	1	4
1076	3	0	0	0	0	0	0	0	0	0	1	4
1138	1	0	0	3	0	0	0	0	0	0	0	4
1200	0	0	0	3	0	0	0	0	0	0	0	3
1201	0	0	0	3	0	0	0	0	0	0	0	3
1203	5	0	0	0	0	0	0	0	1	0	0	6
1264	0	0	0	3	0	0	0	0	2	0	0	5

5.3.4 What locations are identified as optimal for delivering multiple ecosystem services?

The ecosystem services option uses information from metrics that were converted into the grid system in the previous chapter to produce a restoration option based on a few key services to the HHL, using what was identified as being delivered by current wetland patches. Again, this option is about restoration through looking at areas where ecosystem services already exists, rather than potential delivery. Of the ecosystem services discussed in the previous two chapters, five are used here, which were originally included due to their importance to the HHL landscape, either through their inclusion in the original HHL Business Plan or the strong links to wetland environments (see Chapter Three). Two of the original choice of ecosystem services have been taken out, which are: food production, because in this landscape it is detrimental to wetland restoration, and pollination, because it was identified as having no pattern with current wetland patches in the previous chapter. The services included are:

- Carbon storage
- Recreation
- Timber Areas of woodland in active management
- Flood Mitigation
- Habitat Quality.

For each service a scoring system was used to build in the amount/quality. Each service had two categories “1” and “2”, with carbon storage and flood mitigation scored as they were in the previous individual service options, with the other three using the same method, with two levels of delivery according to the categories used in Chapter Four. Each of the five were added together to produce Figure 5.9. The five services were added together (without considering the two categories of delivery) to identify areas that deliver multiple services in Figure 5.10 and just category “2” of delivery was added together to identify areas where the highest levels of multiple service to produce Figure 5.11.

In Figure 5.9 the maximum number found in any one grid square is seven (the maximum possible value is ten), with some squares containing no value at all. There is a lot of variation across the landscape, with the two NNRs scoring very highly. Figure 5.10 shows that no squares deliver all five services, but there are a few that deliver three of four services, with many of these found in, adjacent or close to the current wetland patches in the central part of the landscape. Figure 5.11 shows a very different spatial pattern to the other two, with much fewer squares suggested as locations for wetland restoration. This map however shows just the highest delivery of the five ecosystem services, the vast majority of which are in current wetland patches, which presents a good case of restoring wetlands for ecosystem services delivery. No squares deliver the highest levels of four or five of the services, and only one delivers three services. There are a very small amount delivering two or three, the vast majority of which are found within current wetland patches.

Table 5.4 shows the Spearman’s rank correlation results between the three different figures in this section, to compare the varying approaches to understanding the spatial distribution of ecosystem services. There is a very strong positive relationship between Figure 5.9 and 5.10, and weak positive in either case with Figure 5.11. Figure 5.11 will not be used for further analysis, as

it suggests so few squares for restoration outside the current wetland patches and only one of the other two will be selected because they produce such a high result in Table 5.4. Figure 5.10 was selected because it best helps answer the question of can we deliver multiple ecosystem services.

Table 5.4: Spearman’s rank correlations between the ecosystem services restoration options.

Comparison	Results
Figure 5.9 vs Figure 5.10	0.997
Figure 5.9 vs Figure 5.11	0.238
Figure 5.10 vs Figure 5.11	0.198

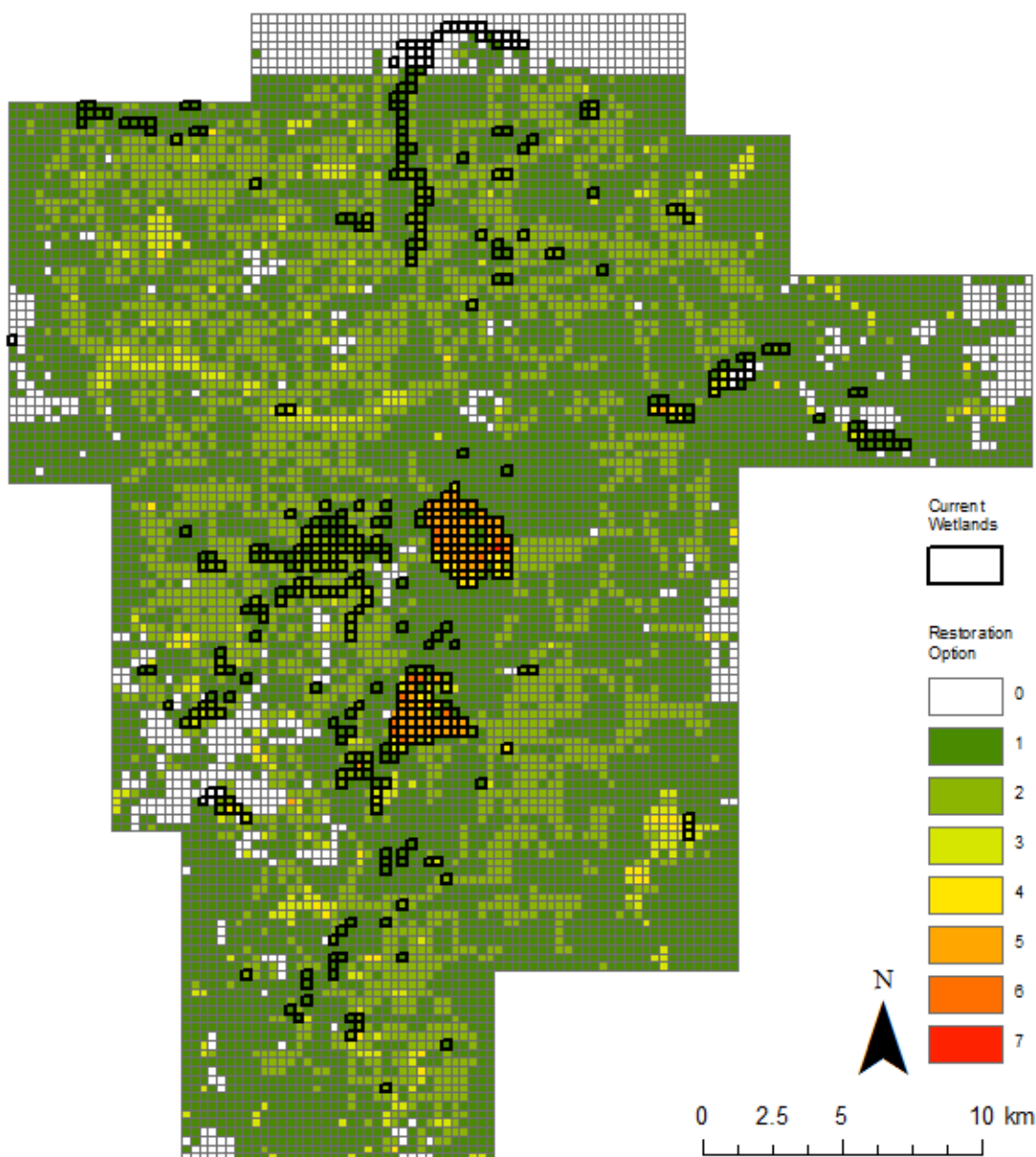


Figure 5.9: Ecosystem services targeted restoration option, where a higher score indicates more delivery of ecosystem services, and therefore a stronger suggestion for restoration.

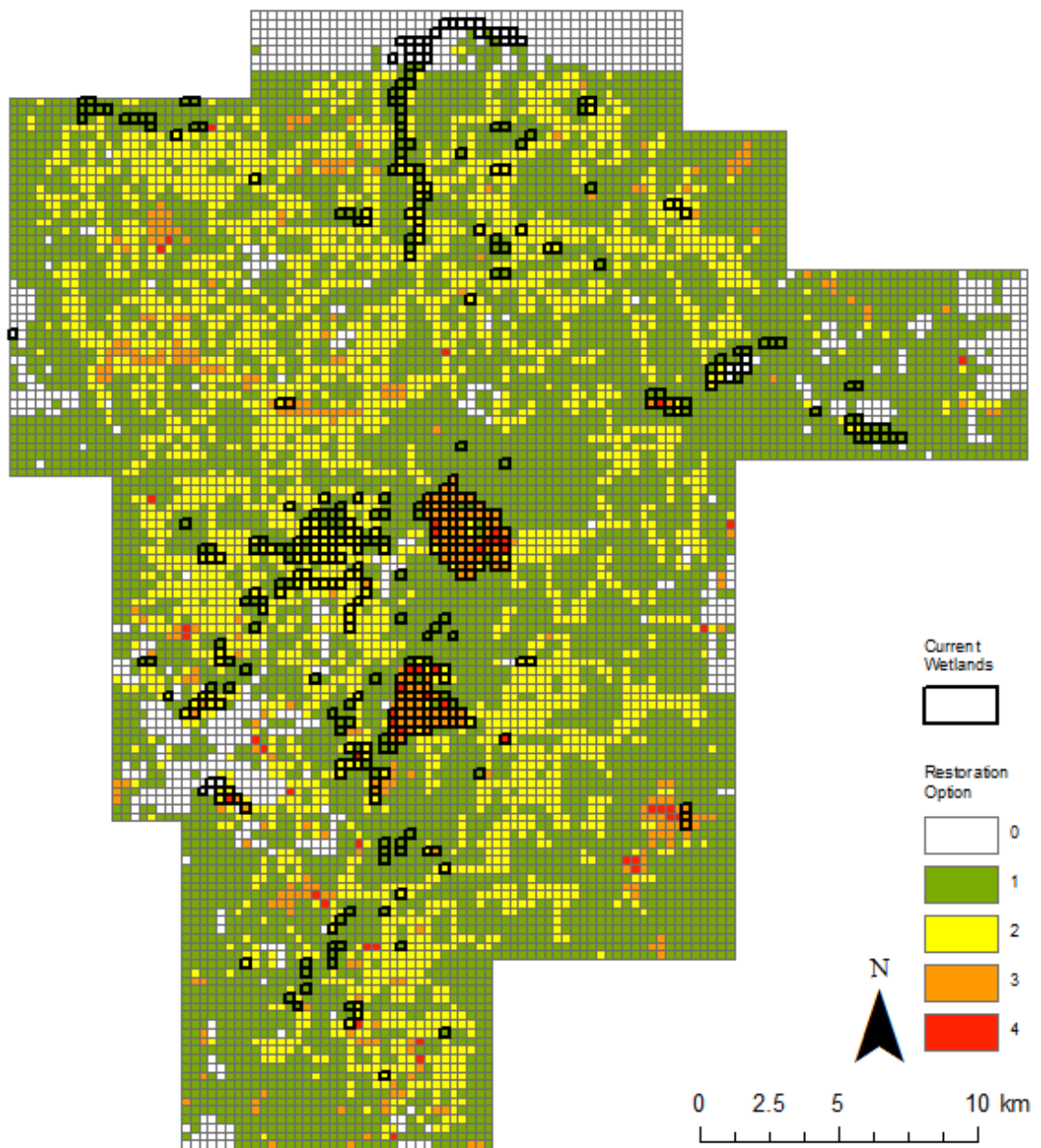


Figure 5:10: A second ecosystem services restoration option, where the five services were added together without considering the two levels of delivery, and so a higher score indicates more services are delivered, which would then be a stronger suggestion for restoration.

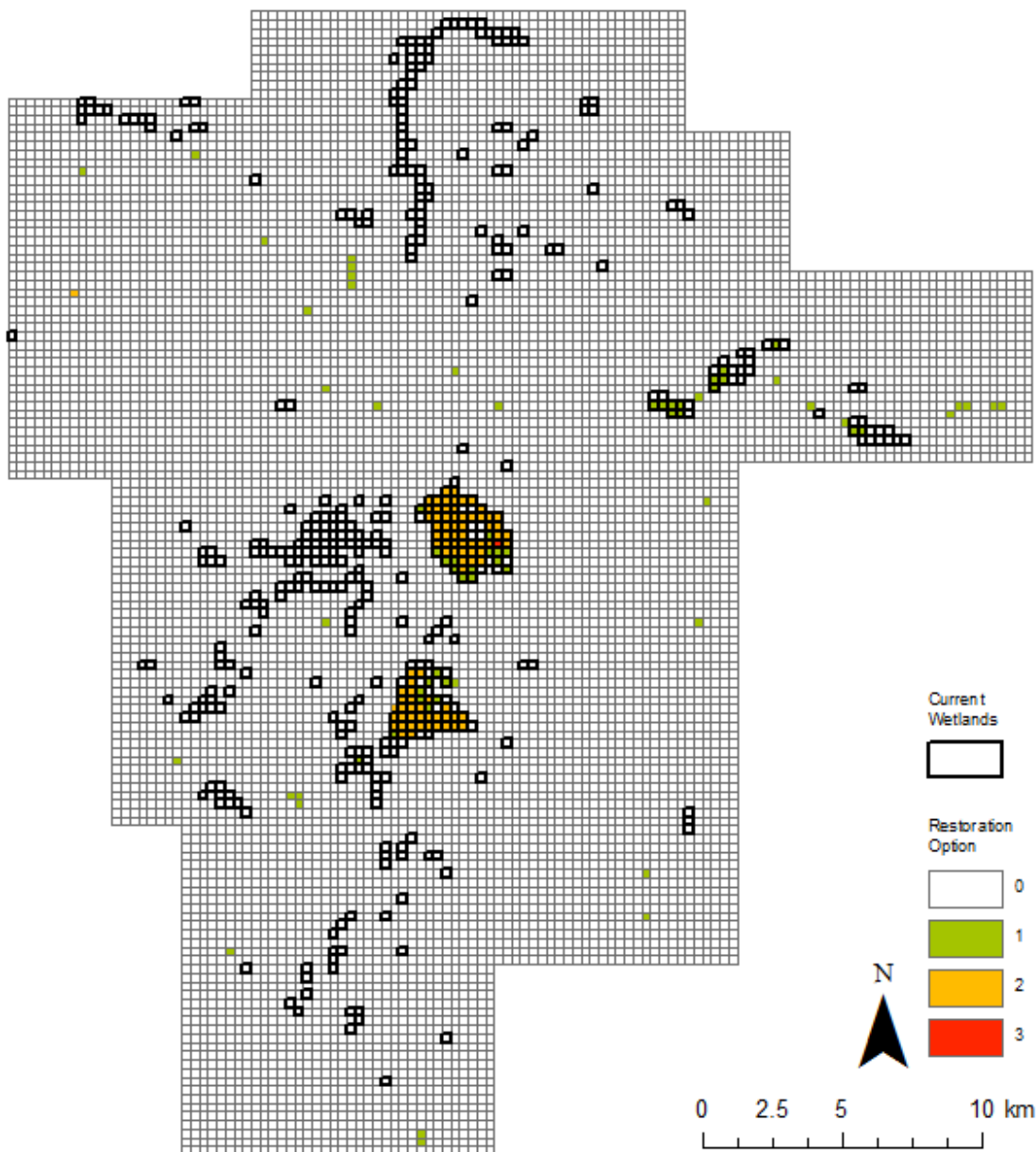


Figure 5:11: Another ecosystem services restoration option, where the category “2” of delivery for each of the five services is added together, and so the higher the value the stronger the suggestion for restoration.

Figure 5.10 can be used to answer the questions of which locations are identified for restoration using ecosystem services and can we deliver multiple ecosystem services. With the first question, there are many squares that are suggested for restoration, with the ones that deliver more services found around the central current wetland patches. It also connects other parts of the landscape, developing connectivity. With the second question, there are no squares where all five ecosystem services are found, but a handful where there are four, and more where there are three. This suggests that therefore it is possible to deliver multiple

services, as there are many synergies, just not all of them at once, and there would have to be trade-offs in the landscape.

5.3.5 Can both biodiversity and ecosystem services be delivered in the same places?

This question concerns the correspondence between the areas identified as priorities using biodiversity metrics and those identified using ecosystem service metrics. The results from the previous two sections are mapped below, firstly Figure 5.12 maps the biodiversity option from Figure 5.5 with the ecosystem services option from Figure 5.10 and then Figure 5.13 also includes the current wetland patches. It is obvious visually that the hotspots for one are not the same as the hotspots for the other, and the correlation (using Spearman's rank) gave a value of only 0.061, which is an extremely weak positive correlation. Therefore, these results suggest that biodiversity and multiple ecosystem services are not really being delivered in the same places for this landscape, which was a key question to answer with this chapter. Figure 5.12 does show some areas, for example the two NNRs, that deliver high biodiversity and some ecosystem services, but there is little evidence of multiple ecosystem services and a range of taxa in the same place.

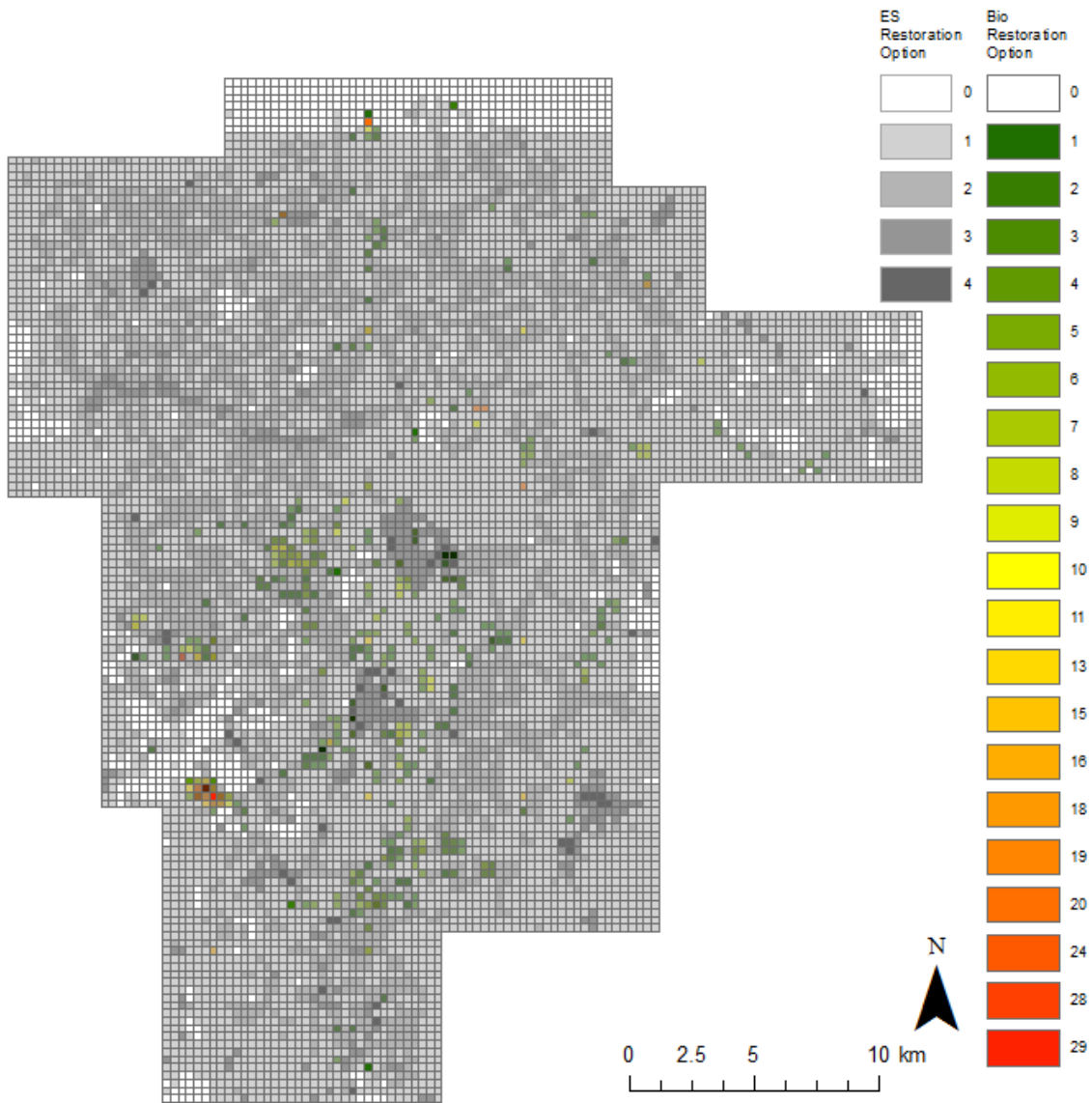


Figure 5.12: Biodiversity and ecosystem services targeted restoration options. Higher scores are stronger suggestions for restoration.

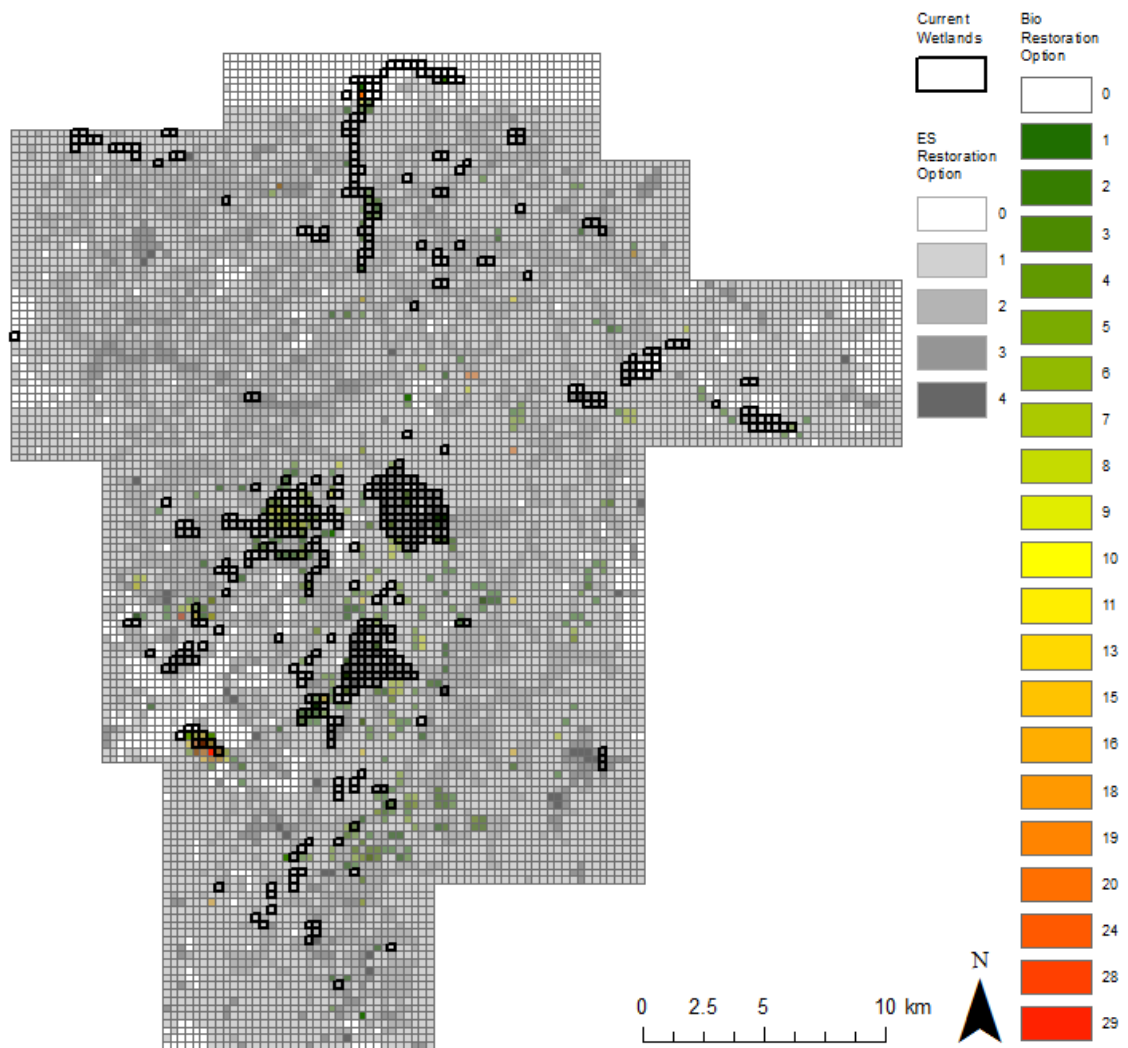


Figure 5.13: Biodiversity and ecosystem services targeted restoration options, with the current wetland patches. Higher scores are stronger suggestions for restoration.

It is also important to compare the biodiversity option to individual ecosystem services, especially because of the poor relationship between biodiversity and multiple ecosystem services. Therefore, the biodiversity option is compared to the two individual ecosystem services covered earlier in this chapter: carbon storage and flood mitigation. Figure 5.14 shows the biodiversity option with the carbon storage service, which has a correlation value (Spearman's rank) of 0.032. Figure 5.15 shows the biodiversity option with the flood storage service, which has a correlation value (Spearman's rank) of 0.012. Both of these in terms of visual and correlation results show poor associations between the biodiversity and single services. Therefore, in the case of these two services it is not possible to deliver them with biodiversity, but it may be different for other services, but this is the case with these two examples.

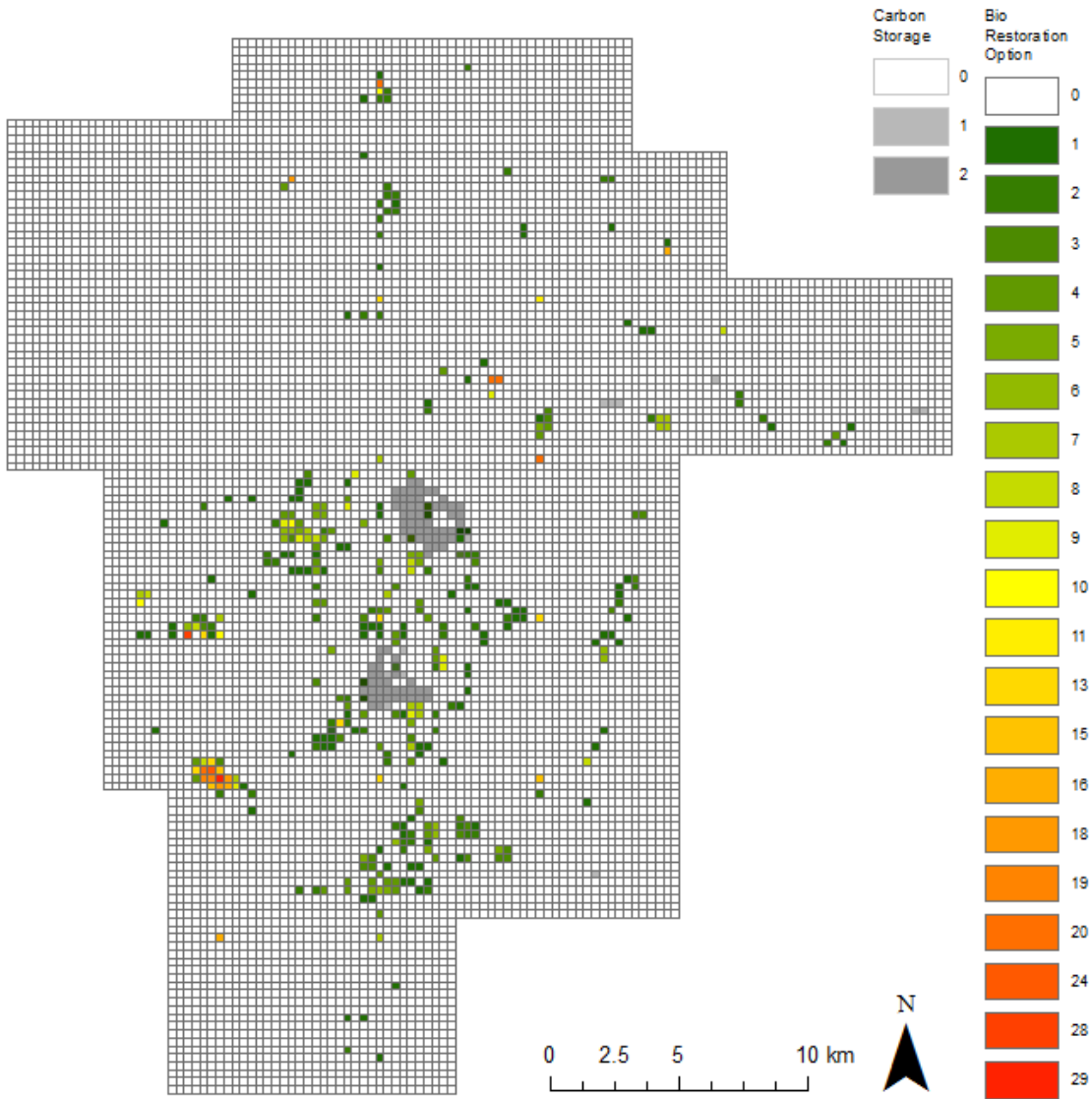


Figure 5.14: Biodiversity and carbon storage restoration options, where higher scores are stronger suggestions for restoration.

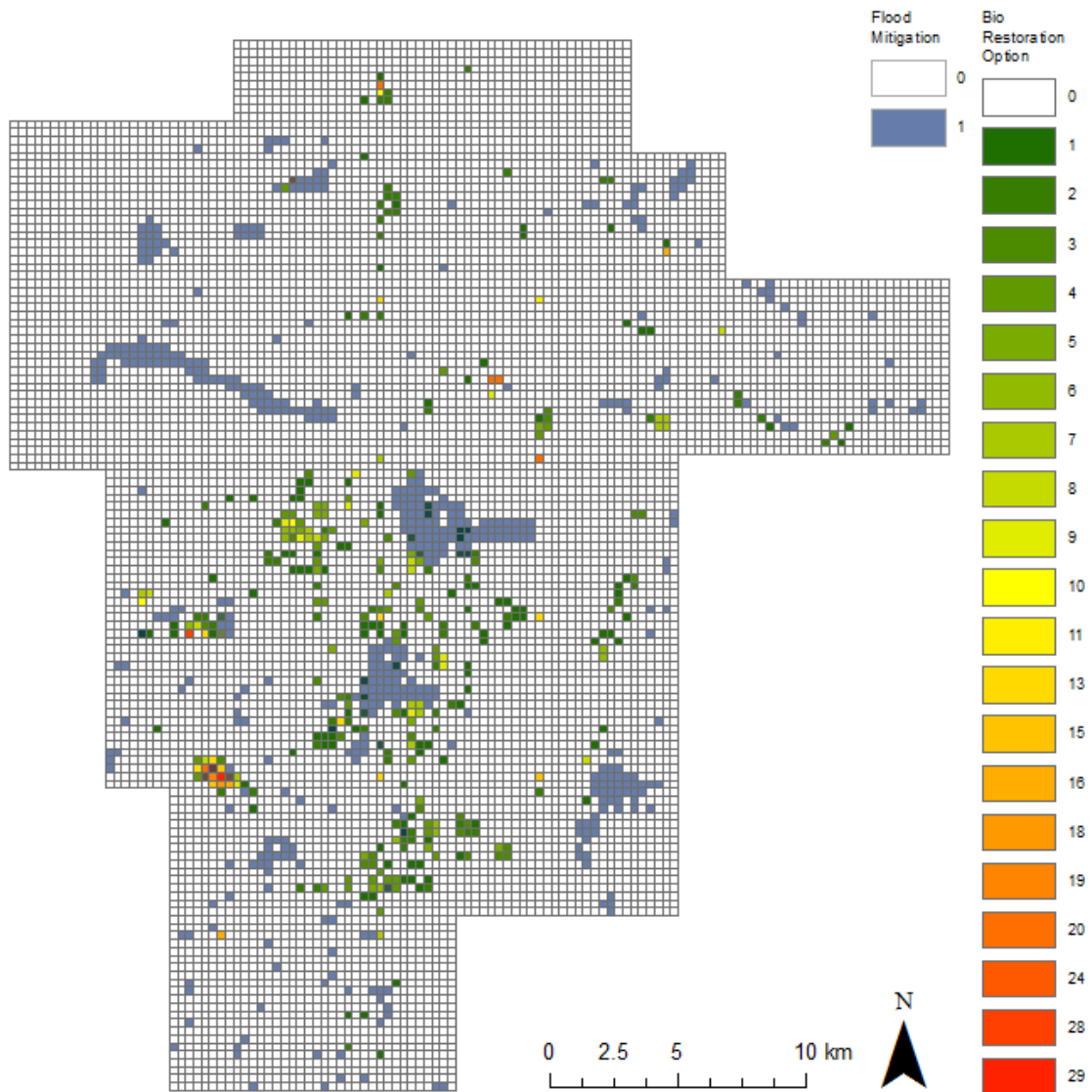


Figure 5.15: Biodiversity and flood mitigation restoration options, where higher scores are stronger suggestions for restoration.

5.3.6 What locations have the appropriate physical characteristics for wetland restoration and do they coincide with biodiversity and ecosystem service provision?

To address this question the range of physical characteristics were used to create a series of rules (see Figure 5.16) to identify which squares might provide the greatest wetland potential, with an added sense check and using those characteristics that are most conducive to creating the right conditions for wetlands. Firstly, topography was included because in the previous chapter it was identified that the vast majority of current wetland patches were in the lowest bracket of height above sea level, which was up to 16.7 m above sea level.

Further investigation revealed that actually most of these squares were 5 m or less above sea level, and so the first rule was to only include the squares below 5m above sea level. Next using the results from the previous chapter, the soil types delivered by current wetland patches were considered, which resulted in four being used in this rule. These were selected by looking at which soils types were delivered more in current wetlands than non-wetlands (see Chapter Four Section 5.3.1), of which there were five, but one of these was not present in any of the left squares at this stage. Finally, the squares identified as historic wetland patches (as defined by Natural England) were included as a rule, because they were previous wetlands and so it would suggest that these areas are more conducive to be wetlands. These three rules results in 524 squares been suggested for restoration (Figure 5.17), and covers 44% of the current wetland patches.

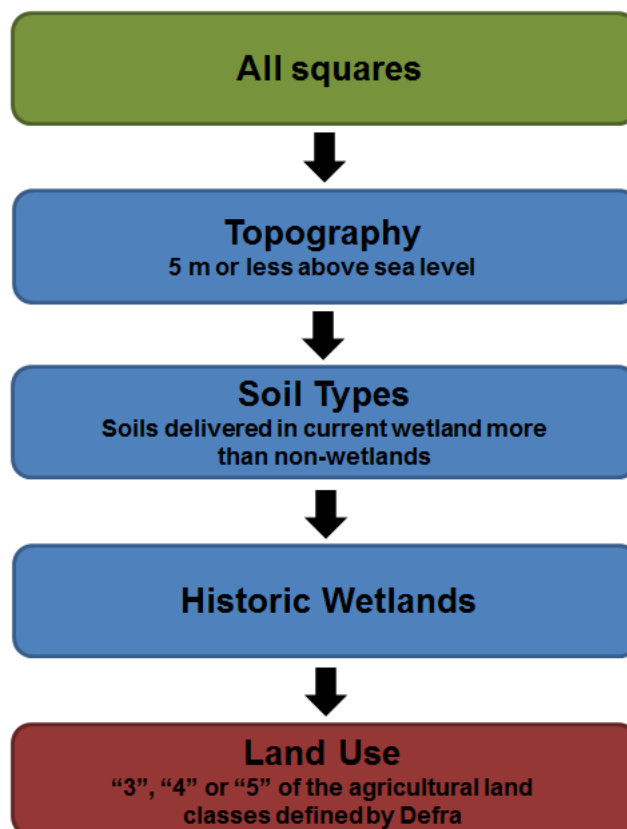


Figure 5.16: Flow chart showing the creation of the wetland potential restoration option. At the beginning there were 9724 squares.

Figure 5.18 has another filter on of land use. This chapter is about the idealistic use of biodiversity and ecosystem services, without other human influences, but agriculture is such an important issue in the landscape it is important to think about how it might influence a wetland potential option. Squares were included here if they were classed as “3”, “4” or “5” out of the agricultural land classes defined by Defra, because these are the three that current wetland patches mostly lie in. In addition, these are the three poorest agricultural land classes, which is where restoration has much more of a chance of actually occurring, as it cannot really compete for land with the high earning and productivity agricultural areas of Grades “1” and “2”, and so restoration is unlikely to occur there. This results in 367 squares being suggested for restoration under this wetland potential option, which covers 40% of current wetland patches (Figure 5.18).

The results in Figure 5.17 show that many of the suggestions are actually current wetland patches (which validates the results, but it is the areas outside the current wetland patches that are of interest in this chapter), or are close to current wetland patches. There is also evidence of some of the suggestions following rivers in the landscape, which could be appropriate suggestions for wetland environments. With the addition of the agricultural filter in Figure 5.18 there are similar patterns in the landscape, but with a third fewer squares suggested for restoration. With both options, there is an interesting area to the west of the Thorne Moor NNR, where there is a large area of current wetland patches that are not at all covered by the wetland potential suggestions, which will be investigated further in the following section.

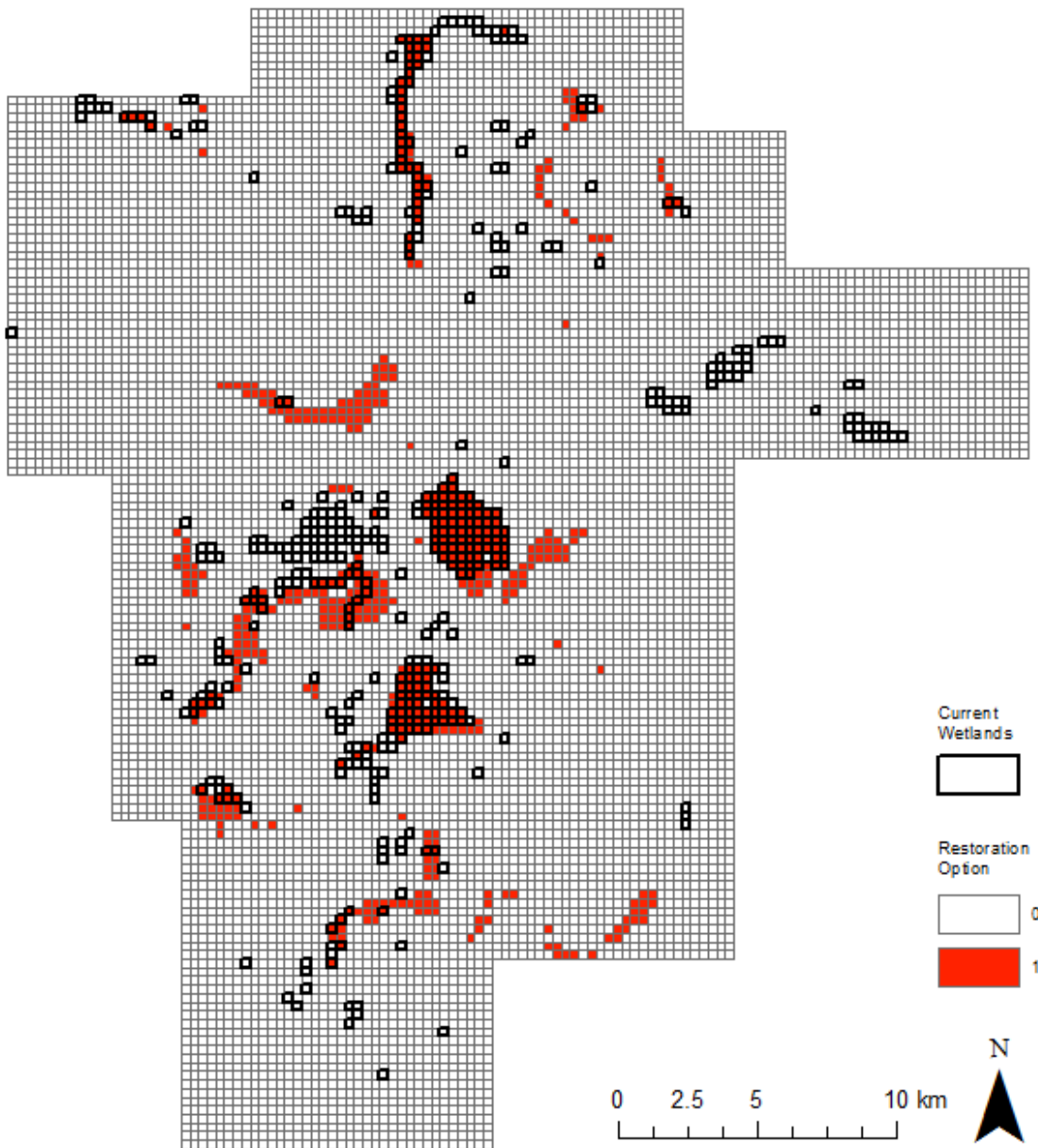


Figure 5.17: Wetland potential restoration option, where squares marked with “1” are suggested for restoration.

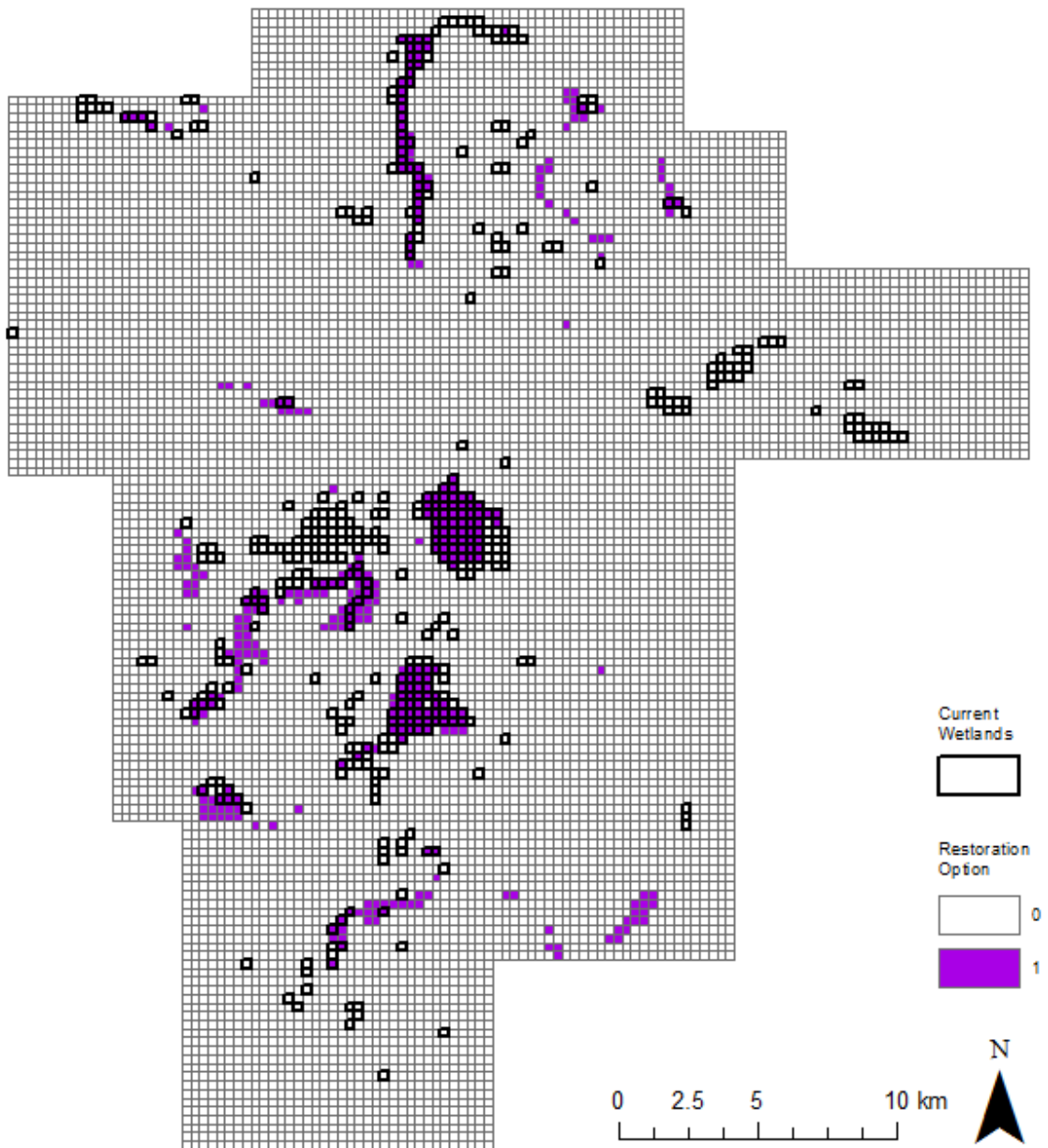


Figure 5.18: Wetland potential restoration option (with the added agricultural filter), where squares marked “1” are suggested for restoration.

It is important to compare these options to the previous ones on biodiversity and ecosystem services. Also, as something similar was covered under the Wetland Vision project using physical characteristics to show wetland potential (Hume et al., 2008), then it is also important to compare their work to this option. The approach used by Wetland Vision was to identify a baseline potential of habitats considering soils, slopes and floodplain location, which then had weightings applied with from a range of factors, including indicator species, protected areas and low grade agricultural land, then creating scores for each sub-catchment (Hume et al., 2008). The comparison is carried out again using Spearman’s Rank

correlation of the values in the grid squares, and the results are presented in Table 5.5. Of these results, all of the relationships are positive, but they are all relatively weak. The strongest results for both wetland potential options is with the ecosystem services option, and the weakest with the biodiversity option. However, none of these, even with ecosystem services, are particularly high values and so in general are not suggesting similar locations for restoration in the HHL. The low correspondence between these results and those from the Wetland Vision are also interesting, because the project used a similar method to look at wetland potential, but with different inputs (Hume et al., 2008), suggesting that these obviously make a big difference. Overall, creating these two wetland potential options has been useful and in general the areas they suggest for wetland restoration in the HHL landscape appear to make a lot of sense, but there are still some areas that would require further investigation.

Table 5.5: Spearman's Rank correlations between various restoration options.

Spearman's Rank	Wetland Potential	Wetland Potential with Agriculture
Biodiversity Option	0.127	0.122
Ecosystem Services Option	0.400	0.416
Wetland Vision	0.204	0.170

5.3.7 What do these restoration options mean for managing this landscape?

These restoration options highlight the areas of ecosystem services and biodiversity to target for restoration and the areas where current physical characteristics have the potential for wetland restoration. All of these could inform restoration decisions, allowing conservation effort to be prioritised around areas of existing high biodiversity or ecosystem services or the areas of greatest wetland potential, where restoration action is likely to be most effective. However, depending on which of the options used, the resulting configuration of wetlands within the landscape may be quite different. This section considers how the different options could inform restoration, the resulting landscape configurations and their alignment with current principles of landscape ecology.

A founding principal in landscape scale biodiversity conservation is ecological networks, i.e. having core areas of high biodiversity value and facilitating

connectivity between these within a landscape (Bennett and Mulongoy, 2006; Lawton et al., 2010; Harrington et al., 2010). Even though this principal is only directly relevant to biodiversity, applying this to the ecosystem services and wetland potential options aligns the three options and allows exploration of coincidence and divergence, thus informing approaches for restoring multi-functional landscapes (i.e. landscapes that support high biodiversity and deliver ecosystem services within multiple land uses). How each option and the coincidence of them together meets these points is touched on in the previous sections and will be discussed in further detail in Section 5.4. Here what the different restoration options suggest for on the ground management is investigated in order to provide information for practitioners. Three examples (see Figure 5.19) are considered that demonstrate the application of this research for informing on the ground restoration in the HHL, which could be applied beyond.

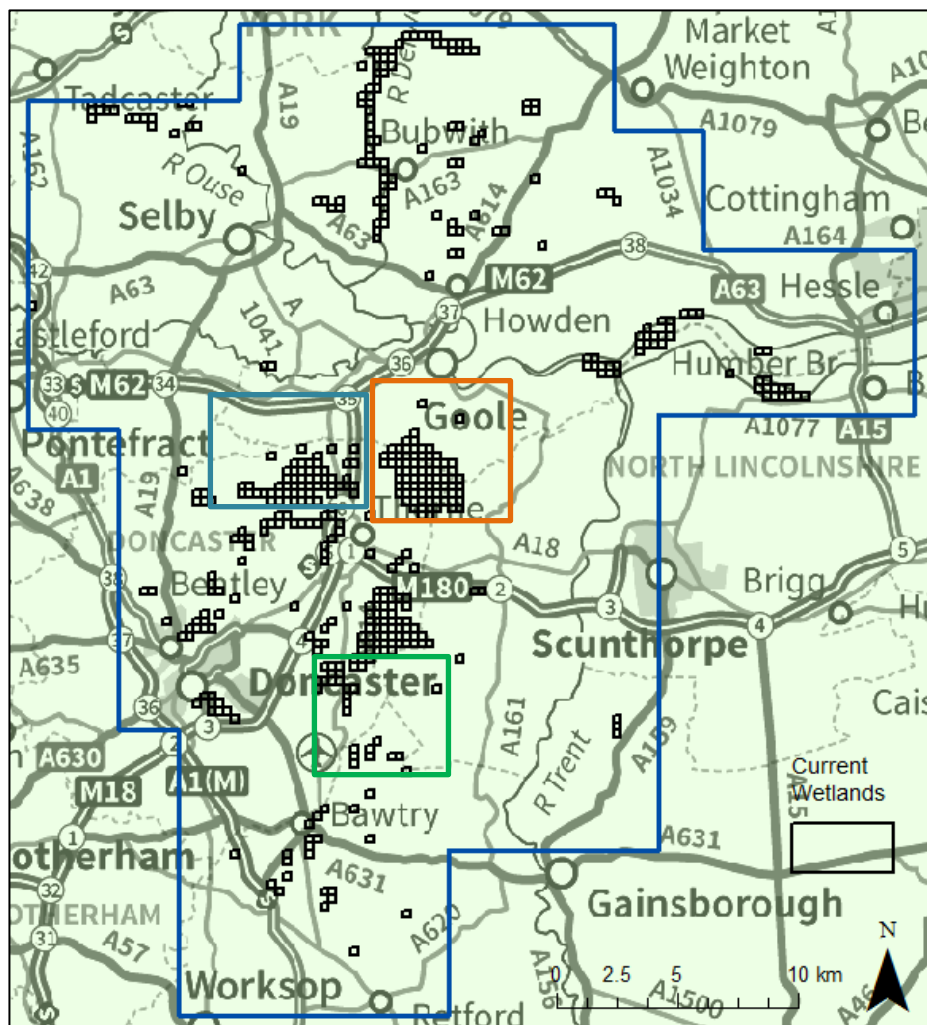
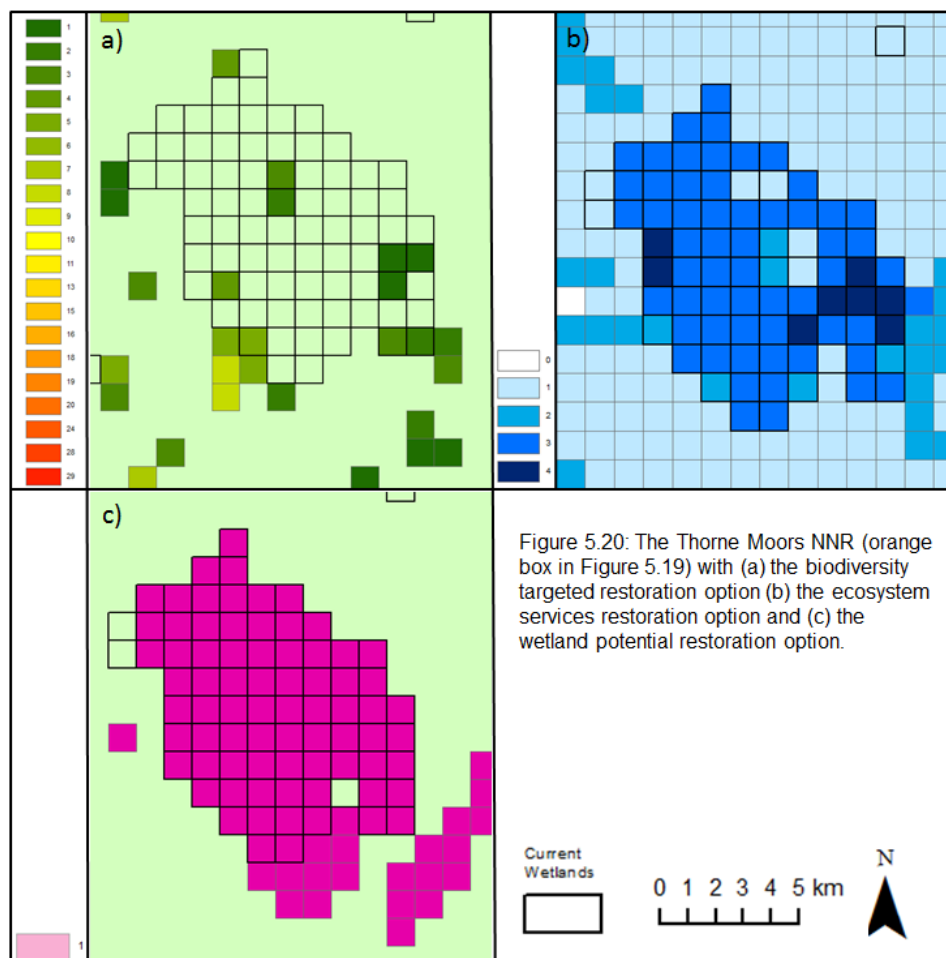
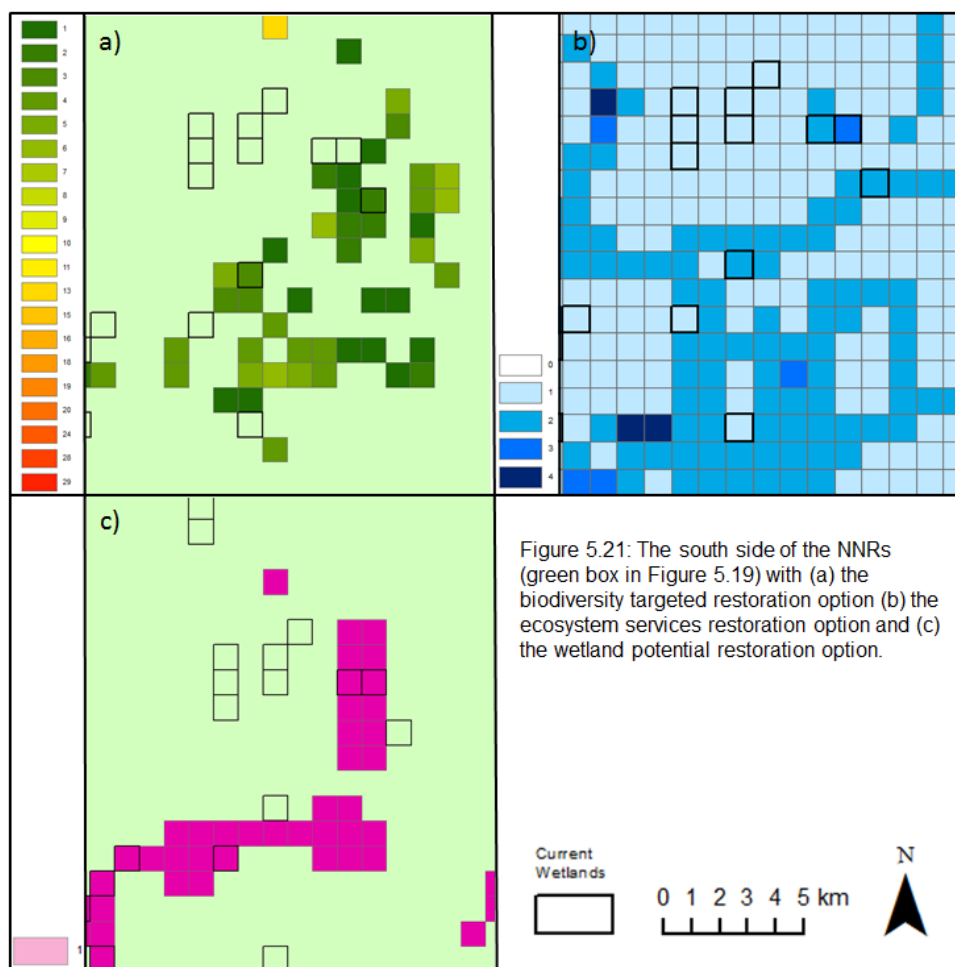


Figure 5.19: Locations for the examples in this section (see individual examples for more detail).

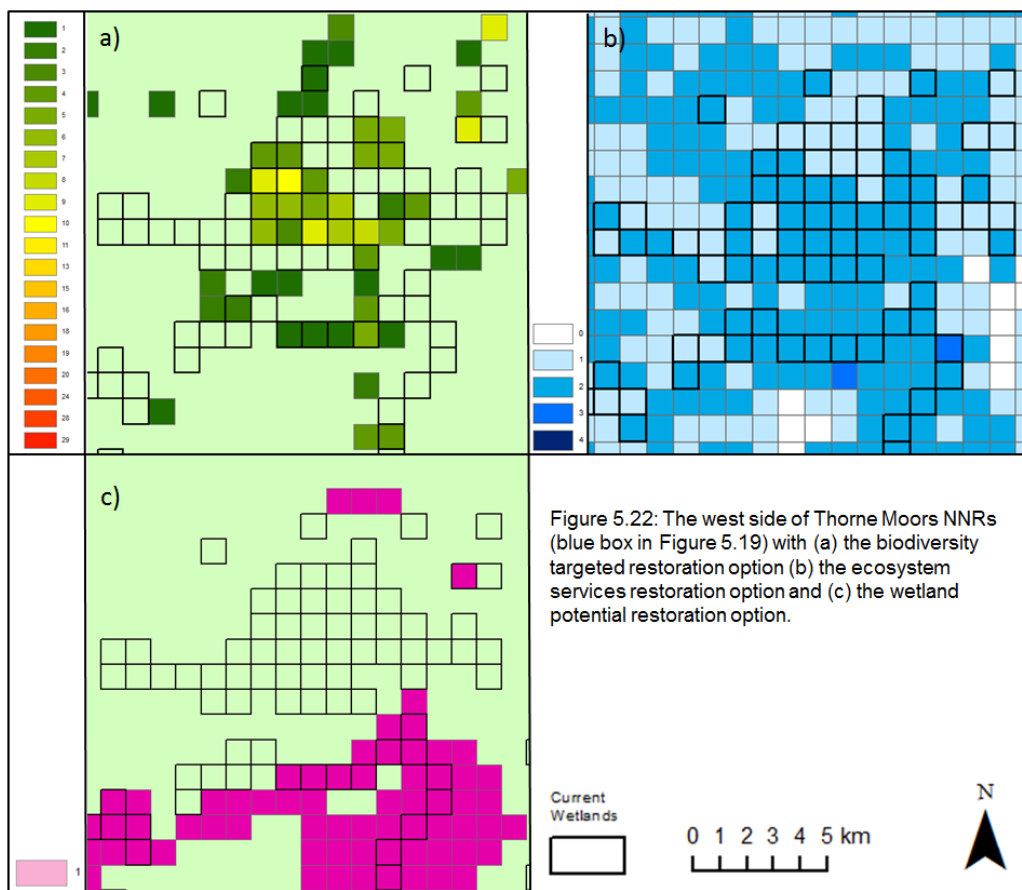
The first of these is looking at the expansion of core areas around Thorne Moors NNR, the orange box in Figure 5.19. This area was chosen because it is already an important core area for the HHL and appeared in all the restoration options. Core areas and the expansion of them are important because these areas currently support a range of species and so have good ecological quality (Harrington et al., 2010), but we need to keep increasing them to provide more space and resources for species. Figure 5.20 shows the three options for this area. The biodiversity option shows existing biodiversity to be highest in isolated patches around the NNR area. Therefore, restoration to improve biodiversity should involve expanding the NNR area to incorporate these assets. The ecosystem services option does show good coverage of the current wetlands, mostly delivering three out of five services. There is a small amount of suggested expansion to the edges of the current wetlands, but many of the surrounding squares only deliver two services. The wetland potential option has good coverage of the current wetlands, with some expansion to the southeast. Overall, these options clearly show that there should be some enhancement and expansion of the core wetland areas as the approach for management.



The second example is a collection of small wetlands that exist to the south of the NNRs (the green box in Figure 5.19). This area (seen in Figure 5.21) was selected as it would require a very different restoration approach to the NNRs. Restoration in this area would require connecting the existing patches, using corridors and stepping stones, facilitating ecological networks (Rodriguez-Iturbe et al., 2009; Harrington et al., 2010; Tjorve, 2010). With the biodiversity targeted option there are quite a few squares suggested for restoration that would result in current wetlands being physically connected, although most of these have low values, and so are weaker suggestions. For the ecosystem services option there are some darker squares indicating that current wetland patches could become physically connected, but only delivering two services in most cases, and so again are weaker suggestions. Finally, the squares suggested under the wetland potential option do expand out on current wetlands, but do not always make them physically connected, the importance of which would vary between species. Overall, the options suggest some physical connection between current wetland patches, and so therefore potentially increasing connectivity in this part of the landscape as a restoration approach for management.



The final example in Figure 5.22 is the area to the west of Thorne Moors NNR (the blue box in Figure 5.19), which was chosen because of the variation in results between the three different options. Both the biodiversity and ecosystem services options have areas between the wetlands that deliver them, and actually show more coincidence than the other two examples in this section. If restored, these areas would link up the current wetlands and expand them. The results for the wetland potential option is unexpected, as the area is dominated by current wetland patches, but very few are covered by this option. These results suggest that there might be a different type of wetland here that does not have the same physical characteristics as the other wetland patches, but still delivers the biodiversity and ecosystem services. Looking back at the BAP habitats in Figures 3.9 to 3.12 (Chapter Three), it can be seen that this area is coastal and floodplain grazing marsh, whereas most of the others are lowland fen or lowland raised bog. This habitat is less permanently covered with water than the others, and so this physical aspect might be the reason why there is a difference. Overall, the three HHL landscape restoration options do in general suggest some expansion of core areas and connectivity, which is good as it has been identified that the whole network is important for successful conservation (Lindermayer and Nix, 1993).



5.4 Discussion and conclusion

This chapter set out to produce a series of wetland restoration options for the HHL targeting different aspects of biodiversity, ecosystem services and physical characteristics, as well as providing evidence to help answer some of the core questions from the literature. Over the series of options there is a lot of variation between the potential visions of the landscape, in terms of both the information inputted and the results. Many of the later options involve multiple metrics, and are probably more appropriate than those basing restoration plans on a single metric. The options were just considered at the 500 m scale, because there were similar results between the two scales in the previous chapter and 500 m is more appropriate for on the ground restoration decisions. The scale of the original data and that chosen for the grid squares will have made a difference to the results, but appropriate grid scales were chosen based on the original data, the size of the landscape and the literature (e.g. Van Horssen et al., 1999).

These options use the information from the previous two chapters, which are mostly based on where biodiversity and ecosystem services currently exist. Therefore, the options are looking to protect what we already have by targeting them through restoration, other than the final option, which looks at the potential of the landscape. Some options suggest many grid squares for restoration, which would not be realistic to achieve for a range of reasons (Van Kersbergen and Van Waarden, 2004), which is discussed in the following chapters. Also, the grid squares are at a fairly large scale at 500 m and so it would not necessarily be possible to restore the entirety of any one square. Thus, once a square had been selected for restoration then more detailed work may be required to identify specific local areas for restoration. Analysis here at a smaller scale for the whole landscape was not possible because of the large area and the format of the secondary data that was collected and curated for this project. There are strengths and weaknesses to using the methods in this work for informing wetland restoration. For example, it uses a huge amount of information, which makes the results more credible, but is fairly time consuming. However, even though there are issues, these options are useful for looking at the restoration of the HHL landscape.

5.4.1 So where should we restore?

A series of questions identified from the literature at the beginning of this chapter were used to identify locations for restoration options, each will be covered here before making broader conclusions. However, the question on multi-functional landscapes will be addressed in the following section. The first of these questions was what happened when using individual biodiversity metrics? From mapping the occurrences of two biodiversity metrics in the landscape (wetland habitat plants and breeding wetland birds), both highlighted few areas for focussing restoration effort. These were focussed around core areas, with wetland habitat plants also suggesting some places for connectivity. As discussed previously wetland habitat plants could be a useful metric to make biodiversity based restoration decisions on, but perhaps more data would be needed in the real-world, as the source of data may not be completely reliable. This is an even bigger issue for breeding wetland birds because there was even less data. However, breeding wetland birds could be the type of information that decisions could feasibly be made on.

The second question from the literature, for which carbon storage and flood mitigation were investigated, was: what locations were found with individual ecosystem services? There are still many unknowns about the spatial of distribution services (Wu, 2013), but the concept is being increasingly in conservation (MEA, 2005; Fisher et al., 2008; Maltby and Acreman, 2011; Norris, 2011; UK National Ecosystem Assessment, 2011). For carbon storage, nearly all the suggested locations were within current wetland patches and so there would be little restoration of new wetlands based on this option, but it does advocate the expansion and enhancement of core areas. The flood mitigation option showed suggestions for locations of wetlands both around current patches, so advocating the idea of the expansion of core patches and developed connectivity across other parts of the landscape. The core areas for carbon storage are covered with flood mitigation, but the latter has more core areas. Overall, the two individual ecosystem service options showed slightly different approaches for landscape restoration.

The next restoration option looked to answer the question of which locations are identified as optimal for biodiversity? Targeting restoration to enhance

biodiversity may help to halt the decline in biodiversity, be of benefit to humans, species and the environment, and help move forward with restoration at the landscape scale (MEA, 2005; Otte et al., 2007; Butchart et al., 2010). Focusing on biodiversity means restoration would occur mostly in the central and southern parts of the landscape, around the two NNRs, and at Potteric Carr in Doncaster, expanding on core areas and creating connectivity between these through corridors or stepping stones. Very little was suggested in the northern part of the landscape. However, the output should not just be taken on the visual map value; there is also a need to understand the values, in terms of number of species and their abundance, if robust decisions are to be made about where to restore wetlands based on biodiversity.

The optimal locations for restoration of multiple ecosystem services were also evaluated. This restoration option is a little more complex because so many squares are suggested for restoration and the ones with the highest values are found within current wetland patches. Otherwise, the pattern of ecosystem services is fairly scattered across the landscape, but with comprehensive coverage, and areas seem to deliver specific types of ecosystem services, rather than several. Therefore, it would be difficult to make decisions, but restoration could occur across many parts of the landscape, with expansion of the core areas and developing some connectivity. By targeting restoration for ecosystem services, it means understanding more about delivering multiple ecosystem services at a landscape scale (Pan et al., 2013). It has also been identified that there is a need for looking at trade-offs and synergies with ecosystem services, as they are fairly unknown (Opdam et al., 2002; CCI and Birdlife International, 2011), to which this work contributes. In terms of delivering as many ecosystem services as possible, the maximum is four, but this is rare and so there will have to be trade-offs, which will be addressed in the following section. These results are not really surprising as many papers (e.g. Tallis and Polasky, 2009; HHL Partnership, 2011; Rey Benayas and Bullock, 2012; Pan et al., 2013) have found both trade-offs and synergies.

The next question to be addressed is what locations have the appropriate physical characteristics for wetland restoration? This is important because management decisions are frequently made based on the spatial distribution of physical

characteristics (Casado-Arzuaga et al., 2013). Using these results, many of the suggestions for restoration occur around the two NNRs and following some of the river corridors in the landscape. Therefore, this option is advocating both expansion of core areas and developing connectivity. This option used a similar method to that of the Wetland Vision project (Hume et al., 2008), but there was limited correspondence between the two sets of results, suggesting that choosing different physical characteristics or different sources of information will change how a restoration option might look, even in the same landscape.

The final question in this section is what do these restoration options mean for managing this landscape? It is important to understanding what these results may mean when people are making decisions on the ground and one way of doing this has been identified as through ecological networks (Bennett and Mulongoy, 2006; Lawton et al., 2010; Harrington et al., 2010). Overall, the HHL landscape restoration options through biodiversity, ecosystem services and wetland potential do suggest some expansion of the ecological networks on the ground if they were implemented, through enhancement and expansion of core areas and developing connectivity, but it does not always work as a management approach in this landscape. As identified previously there is core area enhancement and expansion, and connectivity, which are both important approaches for conservation (Pringle, 2003; Bennett and Mulonguy, 2006; Piper et al., 2006; Lawton et al., 2010; Gonzalez et al., 2011).

5.4.2 Can a multi-functional landscape be delivered?

Through this chapter the spatial distribution of biodiversity, ecosystem services and physical characteristics have been used to target restoration locations in the HHL. However, according to these results a multi-functional landscape would be difficult to deliver, especially between biodiversity and ecosystem services, which can be visually seen in Figure 5.23. The question has been a particular focus of attention in the literature (e.g. Anderson et al., 2009; Fisher et al 2008, Eastwood et al 2016), but doubt has been voiced as to whether there is currently sufficient evidence to answer it (Bullock et al 2011), and so these results add important evidence. Rey Benayas and Bullock (2012) argue that biodiversity often loses out

to ecosystem services as it has different requirements in restoration and now is often seen as being of lesser importance, but both are important for achieving the restoration of wetlands. To use these results to achieve both, trade-offs will need to occur, both between biodiversity and ecosystem services, and between different ecosystem services, which will be context dependent and informed by a range of human related factors, as covered in the following chapters. For example, in Figure 5.20 most the area delivers at least one ecosystem service, but very few squares deliver high levels of biodiversity. However, many of the ecosystem services are based on models on the entire landscape, whilst the biodiversity data is based on individual records and so it is inevitable that the ecosystem services will show more extensive recovery and that biodiversity shows isolated hotspots.

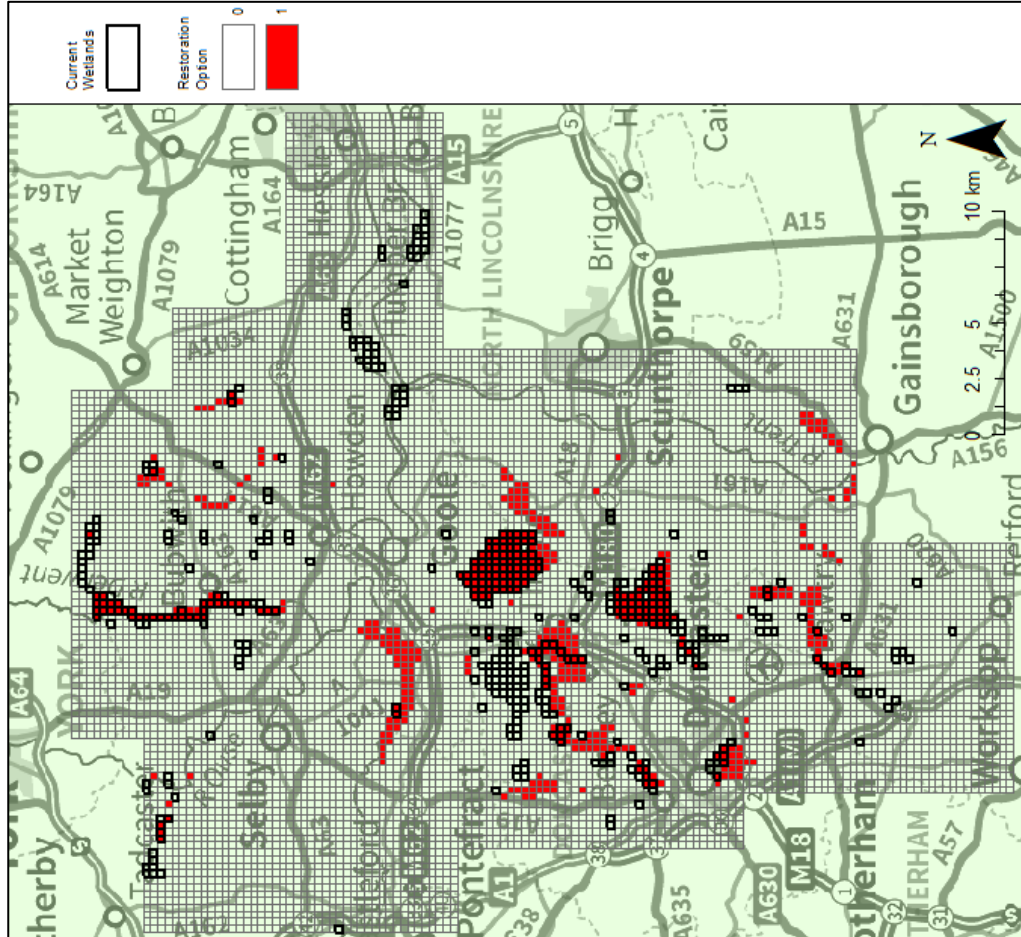
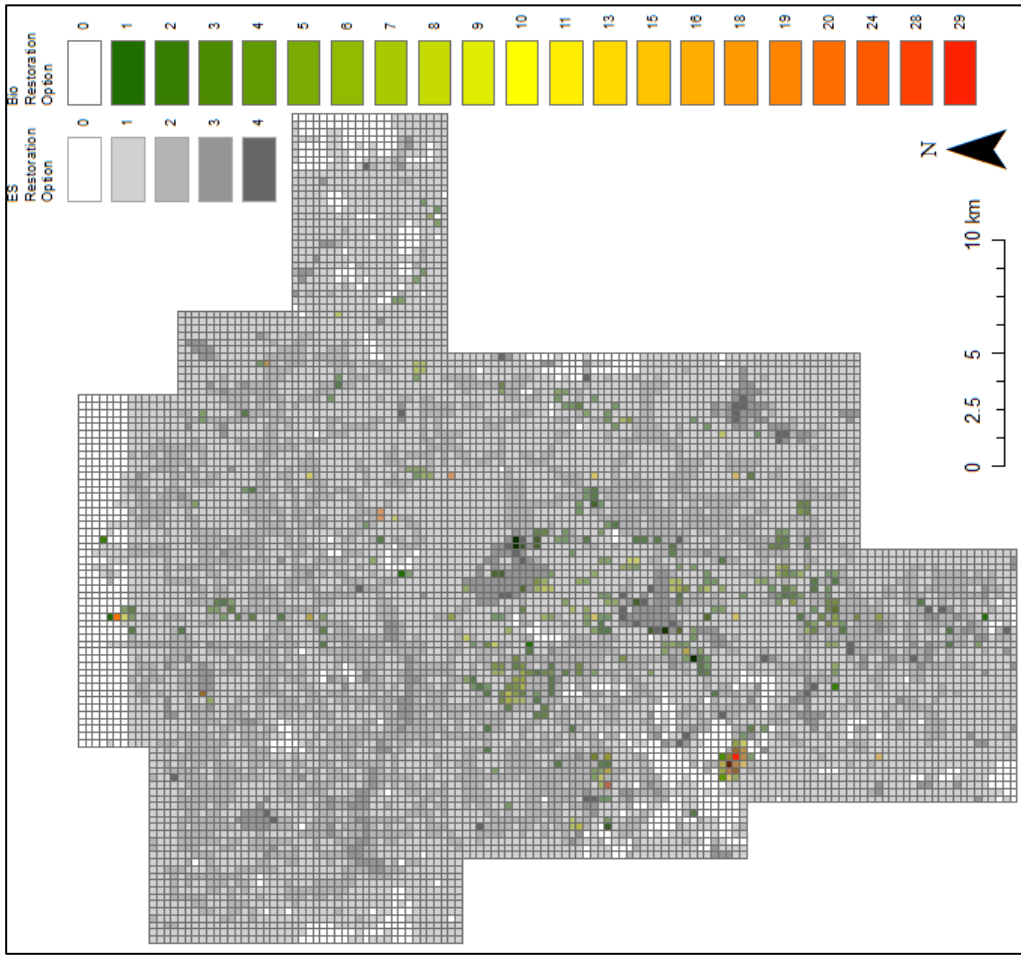


Figure 5.23: A synthesis of the biodiversity targeted, ecosystem services targeted and wetland potential restoration options, with the background landscape map. (It was not possible to lay all three on top of each other, as it was possible to read all the information when attempted.)

Between the ecosystem services, Figure 5.23 shows that very few areas deliver more than two services, and so there would need to be decisions concerning the trade-offs between these. There is also the issue of trade-offs between biodiversity and individual ecosystem services, as it was found that both cannot be delivered with the examples used in the work. However, trade-offs are an inevitable part of restoration decisions (Buckhard et al., 2012). When comparing the biodiversity and ecosystem services options to the wetland potential option there were poor relationships, and so it would be very difficult to deliver even two of these options at the same time.

5.4.3 Summary

The objective of this chapter has been addressed, looking at the best places to restore wetlands in the landscape in an ideal world, where other human based factors are not involved. This is important because in order to plan restoration, knowledge of the scientific concepts that affect the landscape need to be taken into account. The information collected in the previous two chapters, mapping and understanding the delivery of biodiversity and ecosystem services in current wetland patches, was very useful in producing the results here, and all three chapters together also give improved understanding of biodiversity and ecosystem services within the HHL and beyond. The results provide potentially useful information to the HHL Partnership about the landscape, and a useful methodology for other projects and landscapes. A range of options have been produced in order to produce important science-based information for decision making, but not one direct answer has been given in order to not be too prescriptive. Overall, the options suggested have contributed towards some important questions from the literature. Some of the options suggest opportunities to expand on current wetland patches and increase connectivity, and so potentially work with the ecological networks concept. This would address some of the issues with conservation identified in “Making Space for Nature” (Lawton et al., 2010), that too many sites are too small, natural connections have been lost and that sites are poorly managed and protected.

Using this method to investigate a landscape and create restoration options is important and not something that has been done before, as Holl et al. (2003) points out that very few studies use scientific methods to investigate restoration. It could be attempted by decision makers working on the ground, as it is relatively simple, if they have access to data and models. However, it should only really be used with multiple metrics, because as seen previously, the simpler restoration options are not as strong. The analysis in the last three chapters uses both field data and models, especially for spatially understanding ecosystem system services, as already covered in Chapter Four Section 4.4. In a couple of cases, for example, flood mitigation, the two sources of information provide very different results, which would need to be considered. For decision makers, both approaches would be useful, especially to increase the likelihood of having the best information to attempt to make decisions with, but the models could be difficult for many practitioners to use.

This chapter, and the previous one, have started to highlight some of the issues of what can get in the way of science targeted restoration options. It would be impossible to make the landscape look like any of these options, there are too many other issues playing a part other than just biodiversity and ecosystem services in the real-world (Van Kersbergen and Van Waarden, 2004; Faith, 2012). Here, issues of data availability scale and format, as well as what is realistic in the landscape, have all come forward. There are many other potential issues, such as those driven by politics, economics or social factors, which will be investigated in Objectives Three and Four.

Chapter Six: HHL NIA decision making process

6.1 Introduction

The purpose of this chapter is to address the third objective, which is;

To identify the actual decision making processes used in the NIA project, focusing especially on the factors used to choose the location of intended wetland restoration.

In the previous chapter a series of options for the location of wetland restoration in the HHL were produced using information on biodiversity, the concept of ecosystem services and physical characteristics. Whilst the optimal locations for wetland restoration can be predicted using physical and biological criteria, in reality there are a number of other factors which impact where restoration will actually occur. These factors will have influenced the decision making of the HHL Partnership for the NIA Programme, and so this will be used as a case study for understanding decision making. Some of these other factors were highlighted in the previous chapter, such as data availability, but there are many more. Here the term decision making refers to the HHL NIA process, specifically the selection of locations for wetland restoration, including the identification of the geographical boundary of the NIA. The main interest is in assessing what role biodiversity and the concept of ecosystem services had in the process, as well as trying to understand all the other factors that were involved; making up the idea of real-world management. Currently, there is much uncertainty in conservation decisions (Regen et al., 2005), especially over the outcomes (Pullin et al., 2013), and so this study addresses an important research gap.

There is an acknowledged need for work at a landscape scale and to move beyond protected areas, in order to achieve more successful conservation, as addressed in Chapter One Section 1.2.2. The idea has been addressed in policy, as well as promoted in academic literature. The discussions in the academic literature in general support of this way of working and see the benefit of it in linking existing projects to do conservation at a bigger scale and achieve more, but more knowledge is needed to attempt to better integrate it into policy (Botequilha Leitão and Ahern, 2002; McKenzie et al., 2013). There is evidence

for its involvement in policy with documents, such as “Making Space for Nature” (Lawton et al., 2010) and “ThinkBIG” (English Biodiversity Group, 2011), which both discuss the increasing need for action at the landscape scale. Both also discuss the need to achieve multi-functional landscapes for wildlife and people, with both biodiversity and the concept of ecosystem services (Lawton et al., 2010; English Biodiversity Group, 2011).

There is extensive conservation literature on biodiversity and a growing volume on the concept of ecosystem services (see Chapter One Sections 1.2 and 1.3). However, there are questions over whether this literature is used by people making the decisions, and so the practical application of these concepts in decision making is currently unclear. Biodiversity has been explicitly considered in conservation decision making for some years, for example in the EU 2020 Biodiversity Strategy (European Union, 2011). In the past, biodiversity decision making has focused on protected areas, species led approaches and the idea that ecological processes can be manipulated (Carpenter et al., 2006; Fisher and Brown, 2014). Now conservation has started to adopt landscape scale approaches and the concept of ecosystem services, a more anthropocentric approach, as the decline of biodiversity was not being addressed.

The concept of ecosystem services is still relatively new, but its advantage to large-scale restoration has been recognised (Sitas et al., 2013; Albert et al., 2014a; Pagella and Sinclair, 2014). However, it appears to have not been explicitly used in much decision making yet, but will probably have been used implicitly in terms of the services without recognising them under the concept. As identified in previous chapters there are still many unknowns with the concept of ecosystem services (Wu, 2013), which will mean that the necessary information is not getting to those who make the decisions. Also, there is a lack of research on practical tools, a lack of good data and there are problems with understanding how the environmental and social sides are linked (Carpenter et al., 2006; Daily et al., 2009; Casado-Arzuaga et al., 2013; Hatton MacDonald et al., 2014; Fletcher et al., 2014; Pagella and Sinclair, 2014). Therefore, there is a clear need to understand how decision makers currently deal with the concept with the many unknowns and challenges (Daily et al., 2009).

However, a challenge for making decisions based on biodiversity and the concept of ecosystem services, especially at the landscape scale, is data scarcity, which means proxies are often used instead (Smith et al., 2008; Knight et al., 2011b). Also, often decisions that have affected biodiversity have been in areas that are not actually aimed primarily at biodiversity, for example agriculture and infrastructure, that then have unintended consequences, which is a rationale for landscape scale working. Therefore, decisions can affect the environment as a whole, and it is important to consider a whole range of factors, because there are often far reaching consequences.

There are many discussions in the literature over the role that science (such as biodiversity and ecosystem services) has in decisions and the need for “evidence-informed conservation” (Dicks et al., 2014b). However, it appears that currently the science is not really used, due to a lack of research, poor access to research and a lack of time and money to fully utilise the research, creating a “science-practice implementation gap” (Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015). With the concept of ecosystem services, and to a degree biodiversity at the landscape scale, there is also barely any precedence of their use for decision makers to learn from (Rucklehaus et al., 2015). By examining the ways biodiversity and the concept of ecosystem services were utilised in the HHL NIA, this chapter can inform clearer decision making in the future.

Inevitably within multi-functional landscapes there are competing land uses and a range of other factors that need to be considered when making decisions. However, there has been little work identifying these factors. Faith (2012) states that the act of conservation occurs anthropogenically and intrinsically, and therefore even though ideas about biodiversity and the concept of ecosystem services will have some role in restoration, the process is human driven as well. These other factors need to be considered in order to attempt to make decisions more realistic and successful (Maltby, 2010; Pullin et al., 2013; Cook et al., 2014; Dicks et al., 2014b). The range of other factors can include social and economic issues, politics, institutional ideas, experiences, opinion, values and local knowledge (Pullin et al., 2004; Knight et al., 2011a; Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015), which all need to be acknowledged with the information that scientists provide (Sutherland and Freckleton, 2012). Specifically,

in the case of wetland restoration, Hume et al. (2008) discussed how “opportunities for wetland creation are constrained by a range of physical, social and economic factors”. Thus as well as the argument that scientific research is not utilised there is also the fact that the other factors are not really acknowledged (Albert et al., 2014b).

The way decisions are made is often defined by the governance, in terms of the individuals, organisations and processes involved. The governance of conservation, specifically landscape scale restoration, is not well covered in the literature, but it is where the different factors, including biodiversity and the concept of ecosystem services, come together in order for decisions to be made. As stated previously, when governance is discussed in this project it is meant as the real-world process, where the complex factors interact in real life, rather than what we imagine or aspire to happen. A shift in governance in recent years has seen an acknowledgement that it is important to have a network of stakeholders involved in a decision making process, as it adds a wider expanse of knowledge and experience, reduces uncertainty and achieves decisions that are more widely accepted (RSPB, 2001; Brechin et al., 2002; Hume et al., 2008; Lems et al., 2011; Ruiz et al., 2011). However, it could be said that decision making has always been shared, but that this has not been made obvious until more recently when the importance of sharing has been acknowledged.

In practice, decisions around conservation are made by a huge variety of people (Verissimo et al., 2014), such as members of the HHL Partnership. Each stakeholder and organisation will have different ways of thinking, in particular, different types of organisation will have different priorities, such as between a QUANGO (Quasi-Autonomous Non-Governmental Organisation) and an NGO (Non-Governmental Organisation). Even though both are not controlled by government, they have their own specific interests with the environment, with the QUANGOs more linked to government policy and NGOs setting their own regulations. The different definitions and organisational priorities will play out in negotiations over decisions (Lems et al., 2011). However, there is also the belief that a partnership means more joined up thinking and actions for conservation. Therefore, more research is needed into how partnerships work in making decisions and the HHL Partnership provides a good on the ground case study.

The HHL is considered a flagship example of good practice for partnership working and large-scale conservation in the UK. Thus, whilst this chapter is exploring the range of factors that affect conservation decision making, we would expect the HHL to demonstrate some of the best use of biodiversity and ecosystem service information.

In Chapter One adaptive management was identified as a useful approach to consider with conservation work, and it brings together some of the aspects already discussed here. The concept acknowledges uncertainty within decision making in the natural environment and that by working with multiple stakeholders ongoing learning can occur to improve the process (Folke, 2006; Jasonoff, 2010; Voss and Bornemann, 2011; Sharp et al., 2011; Susskind et al., 2012; Westgate et al., 2013; Westling et al., 2014). A series of criteria were put together in Chapter One to understand whether a group was achieving adaptive management, which will be applied here to further understand the decision making in the HHL. The criteria includes: a recognition that knowledge is incomplete and that ongoing learning should be fed into actions, that many stakeholders participate in the process from both social and scientific backgrounds and that the group should be “reflective” with goals to measure themselves against (Voss and Bornemann, 2011; Susskind et al., 2012; Rychlewski et al., 2014; Westling et al., 2014). These ideas fit in with that of having many factors from multiple areas involved, ideas of real-world management and working at this bigger scale of conservation. There are very few on the ground examples of adaptive management (Westgate et al., 2013), which makes it useful to study here using the criteria identified.

The purpose of this chapter is to understand the decision making process that the HHL Partnership went through to make decisions for the NIA Programme, specifically looking at the roles of biodiversity, the concept of ecosystem services, and the other factors involved. This is important to do because partnership working is on the rise and being encouraged in policy and literature, but we do not fully understand the decision making process they go through, and specifically how they use concepts such as biodiversity and ecosystem services. It is known that realistically decisions cannot not be made on these concepts alone, and we need to fully understand what else might be involved. Sutherland et al., (2012) state that we need to learn from the experiences of those doing on the ground

conservation, such as the HHL, to make better decisions. The idea of landscape scale working is also important as it has been identified in the literature that a new approach is needed and the NIA Programme was an attempt by the government to put this into practice. Many of the ideas here can be drawn together under adaptive management, which will be considered in the discussion. The chapter will start with a description of the three methods used, before presenting a case study of how the HHL NIA boundary was decided upon, and then using this to draw out the key themes of what is involved in the decision making process of the HHL NIA.

6.2 Methods

6.2.1 HHL Partnership meetings

Both HHL NIA Partnership Steering Group and Monitoring and Evaluation Group meetings were attended, where observations were carried out to understand the dynamics of decision making for the HHL NIA. Through observation more could be understood about how the partnership works, who was most involved, how the different organisations work together and to build up a working relationship with members of the partnership (Cook, 2005). A series of meetings were attended from the 12th November 2012 until 3rd March 2015. In the vast majority of meetings I was there as an observer, but there were two occasions where I also presented my own work. A research diary was used to record the events of each meeting, as with other studies (Cook, 2005), as well a place to store paperwork, such as meetings agendas. Attendance at the meetings and writing the research diary was an important step for really understanding the HHL NIA Partnership in order to start to answer the objective of this chapter, for them to accept the research project and to prepare questions and select participants for interview.

6.2.2 Documentation

Relevant documentation was examined to better understand the thinking of the HHL Partnership and the regulations of the NIA Programme, which will have influenced the decision making process. The most important document to read was the HHL NIA Business Plan (HHL Partnership, 2011). For understanding the

NIA Programme “Making Space for Nature” (Lawton et al., 2010), the White Paper called “The Natural Choice: securing the value of nature” (HM Government, 2011) and the Natural England website (Natural England, 2016) were also extremely important. Alongside these a series of agendas and meeting minutes were used. Building on the work observing the meetings, the information gained here is used again to contribute to answering the objective, but also to prepare for the interviews.

6.2.3 Interviews

By attending HHL NIA meetings and reading documentation before carrying out the interviews, relevant candidates for interview could be selected and it allowed time to build working relationships with the partners. Building relationships meant that candidates were more likely to agree to be interviewed, and once in the interviews they would be more open to discussion. Consequently, 14 people were approached for interview (out of the approximately 20-25 people involved) as it was important to talk to a range of people, representing different organisations and levels of involvement/type of role in the project (Gross, 2006; Hirsch et al., 2010). Each person was chosen based on their ability to contribute to addressing the objective in this chapter, to understand the actual decision making process for the HHL NIA. The people approached either were identified as being involved in the decision making process (at different levels) or had some awareness of it. In the end 12 participants were interviewed. The missing two either did not respond to requests or felt they were too busy to be involved. The organisations from which interviewees were drawn are listed below, alongside an organisational acronym used in reporting interviewees’ comments.

- Environment Agency (EA)
- Internal Drainage Boards (IB)
- JBA Consulting (JA)
- National Farmers Union (NU)
- Natural England (NE)
- Nottinghamshire Wildlife Trust (NT)
- RSPB (RB)
- Yorkshire Wildlife Trust (YT)

The interviews took a semi-structured approach, using core questions, but also allowing the interviewees to have some control over the direction of the discussions, in order to glean more information and understand what was important to different participants. Tallis and Polasky (2009) believe that in order to get participants to respond trust is needed as well as an interest in telling a story of success, which can partly come from care over question selection. The questions in Table 6.1 were used as starting points, drawn from literature, observations of the meetings and analysing documentation. Table 6.1 includes the reason why each question was included, which overall was to answer the objective of this chapter. Most of these questions were asked to the majority of the interviewees, but not necessarily in this order. The lettered questions were used as prompts if the interviewee was struggling or going off on a tangent.

Table 6.1: The core questions used in the interviews and the reasoning for each.

Question	Reasoning
1. Could you explain your full job role/position/organisation?	To understand each individual's context that they came to the partnership with.
2. Tell me about when you became involved in the partnership and why. (a)Has your job role changed over your time working with the HHL? (b)What do you think of landscape scale working? Was it an attraction?	To understand each individual's role in the partnership and what's driving them to be there.
3. What was your involvement in the formation of the Business Plan? (a)Were you happy with the process? (b)Did it meet the requirements of your organisation (what are these)? (c)Do you wish you were less/more involved?	To understand each individual's role in the NIA HHL decision making process. These first three questions were more for background information and to get each of the interviewees comfortable within the interview.
4. How were decisions (such as aims, objectives and locations) made in terms of the Business plan? (a)What role has biodiversity had in the decision making? Do you think this was appropriate? (b)What role has ecosystem services had in the decision making? Do you think this was appropriate?	The main question to understand the factors involved in the decision making process, especially in terms of biodiversity and the concept of ecosystem services. It also provided case study examples, especially using (c) and (d).

(c)How was the NIA boundary decided upon?	
(d)How did the ecological restoration zone plan come about?	
(e)Did you feel that anything got in the way of the decision making process?	
5. What have been the key challenges in implementing the NIA project?	Another question to understand what factors were involved in the decision making process for the HHL NIA, including issues such as the partnership, NIA Programme restrictions and more practical issues.
(a)What are about the role of biodiversity and ecosystem services?	
(b)Do you feel constrained by the NIA requirements?	
(c)Have real-life issues such as land availability and community disagreement arisen and how do you deal with them?	
(d)Does the NIA provide a good opportunity for lots of organisations working together?	
6. Do you think overall the NIA project has been successful? How do you define this success?	To understand whether each interviewee thought the project, and therefore the decision making, was successful.
7. Do you think things will change with the Landscape Partnership scheme?	To understand the future of the landscape and how the decision making process has changed over time.
(a)How will this affect existing projects?	
(b)How was the boundary for this decided upon?	
8. What do you think of partnership working- from your experiences do you think it is appropriate?	To understand how the partnership has been working together and what that has meant for the decision making process.
(a)What are the big challenges?	
9. What do you think of the concept of ecosystem services?	There is variation in the understanding and satisfaction of the concept in the literature and practice, and so it was an important question to ask to understand the role of ecosystem services in the decision making process.
(a)What are the challenges for the implementation of this concept?	
(b)What do you think of the wording of this concept?	

In accordance with the ethics approval process, an information sheet and a consent form (see Appendix B) were provided to each interviewee, all of whom agreed to be audio recorded. Each recording was transcribed word for word to allow participants to read through the transcript. Then each participant was able

to ask for sections to be removed (occurred once) or to anonymise sections (two participants). When referring to the transcripts, each participant is assigned a code for referral, based on the organisations they work for. For example, the two Yorkshire Wildlife Trust interviewees are referred to as YT1 and YT2 respectively.

Once each transcript was returned, they were imported into the Nvivo software (version 10) for coding. Between using Nvivo, paper copies and information gleaned from the meetings and documents, four broad themes were identified for answering the objective of this chapter; Biodiversity, Ecosystem Services, Decision-other (the other factors involved in decision making) and Partnerships (to understand their role in the decision making process). These themes were chosen to best answer the objective, because of the specific focus of this thesis on the role of biodiversity and ecosystem, and what else is involved in the “real-world” decision making. Child themes were developed for each theme from further reading of the transcripts, again taking into account the chapter objective. For example, within the theme “Decision-other” the child theme “Resources” was included in order to understand how resources, such as time and information, had an influence on the decision making process. For each broad theme eight or nine child themes were developed, which can be seen in Table 6.2.

Table 6.2: The four main themes (and child themes) developed from the interview transcripts.

Partnerships	Biodiversity	Ecosystem Services	Decision-other
Buy-in	Boundary	Benefits	Boundary
Funding	Decisions	Concepts	Constraints
Future	Evidence	Decisions	Funding
Governance	Future	Evidence	Management
Issues	Issues	Future	Partners/Organisations
Positives	Landscape Scale	Issues	Pragmatism/Realism
Trust	Policy/Regulations	Not	Projects
Working	Positives	Policy	Resources
	Role	Role	

In what follows, the information gleaned from all three sources is used to understand the decision making process of the HHL NIA, starting with the boundary case study. A large amount of information has been collected, and so

it is not feasible to use it all. Therefore, the most relevant and insightful will be used to answer the objective of this chapter. Throughout the results and analysis there is a need to be aware that the various sources will have differing reliability. Interviews and memories are notoriously unreliable, and provide evidence partly from what people remember occurring and partly from what they want you to know. Documents are also unreliable because they give an official version, but do not necessarily reveal tensions or difficulties. Also, the Business Plan was written at the beginning of the process, whereas the interviews and meetings occurred throughout. The aim of the discussion, as for the whole thesis, is not to assess the HHL Partnership and comment on the effectiveness of their work, but rather to understand the decision making process.

6.3 Results

6.3.1 HHL NIA boundary case study

This was chosen as a case study to provide a starting point for discussion, because the boundary came up in many discussions and documentation on HHL NIA decisions and because as it covered so many of the factors involved in the whole process. This section explores how the HHL NIA boundary was chosen, as a case study, specifically focusing is on the relative roles of biodiversity and other factors in selecting the boundary. The role of the concept of ecosystem services was never explicitly mentioned with regards to the boundary, but was implicitly mentioned in some cases. The interview outcomes will be used as a basis, with the key themes to be discussed in more detail in the rest of the chapter. Additionally, within the HHL NIA Business Plan there is a section on the HHL NIA boundary (HHL Partnership, 2011), which will also be used. This case study is looking at a snapshot in time at the beginning of the NIA. The boundary of the HHL NIA, as seen in Figure 2.11(Chapter Two), was selected by the HHL Partnership when the NIA funding bid was submitted in 2011. The NIA funding restricted coverage to a maximum area of 50,000 hectares. Otherwise the decision making process was left up to each partnership. The Partnership had a self-imposed restriction within their Business Plan (HHL Partnership, 2011) that they wanted the area chosen to mainly be within the HHL NCA, as seen in Figure 6.1, and they eventually selected 49000 hectares for the NIA.

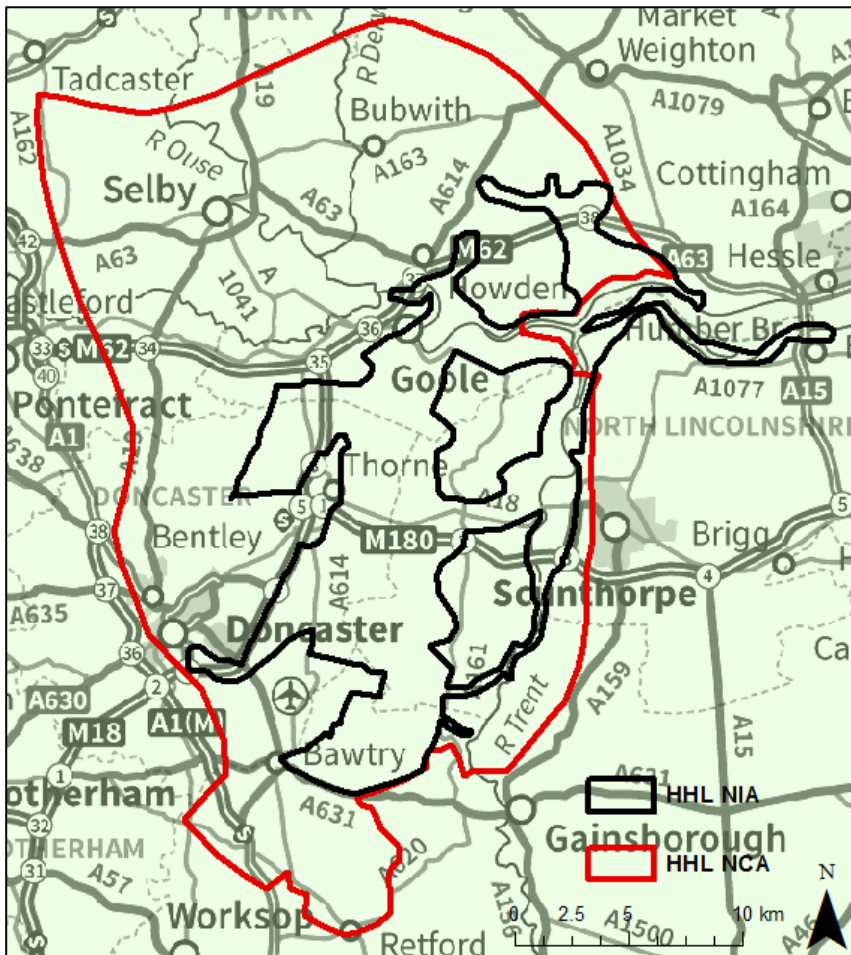


Figure 6.1: The boundary of the HHL NIA and the HHL NCA.

6.3.1.1 Biodiversity in the boundary decision making process

This section deals with discussions and decisions related to the boundary that refer to biodiversity explicitly. Interviewees had varying perspectives on the role of biodiversity, depending on who was involved, the organisation that they work for and when discussed with different contexts. YT1 suggested that biodiversity played an important role: *“in this process they [biodiversity and other ecological concepts] played a big part, they were absolutely fundamental to it. So they drove the essential need for the project, they drove the shape of the overall boundary, they were absolutely central to way we thought the project was going to be delivered”*. However, YT1 works for an NGO, who will partly be very biodiversity driven, which might not have been the case for other organisations.

Most interviewees had a much more varied response and discussed other factors that were involved alongside biodiversity. Interviewees, such as NE3 and YT2, discussed choosing major sites to include in the boundary, areas where

conservation was already taking place, such as designated areas. They discussed these sites as either key assets or priority areas that were chosen as a starting point for the NIA area, to then create ecological networks across the landscape from. The interviewees' discussion of ecological networks directly echoes the emphasis in "Making Space for Nature" of a need to become "more, bigger, better and joined up" (Lawton et al., 2010). The Business Plan includes the idea of connectivity, rather than ecological networks, but they are obviously linked. The initial identification of these key assets is explained by NE3: *"those areas are based on where the protected sites are, like SSSIs, SACs, SPAs, where the RSPB's Futurescapes areas are and where the Wildlife Trusts Living Landscapes areas are. So it involves integrating the priority areas of key partners involved in biodiversity delivery"*. These areas were also mentioned in the Business Plan, where enhancing and connecting core sites, especially the two NNRs, was identified as a priority (HHL Partnership, 2011). These decisions to include protected sites are partly driven by an implicit biodiversity agenda (albeit partially because biodiversity was involved in identifying designated areas in the first place), but the inclusion of these sites also appears to fit with some of the individual organisational agendas.

Biodiversity Opportunity Areas (BOAs) were also discussed as major starting points for the boundary decision making process, and are displayed in Chapter Two Figure 2.2. The process of identifying BOAs occurred before the NIAs (see Chapter Two Section 2.2) and followed biodiversity principles. This was clarified by NE1 who discussed how they had been created through the Regional Biodiversity Forum, for input into the Regional Spatial Strategy. Therefore, there were elements of biodiversity in these decisions to include BOAs, and NE1 suggested that it was done deliberately to make sure biodiversity was included. Discussions over the use of BOAs also came up in the Business Plan that they are based on a robust mapping process that used local ecological data (HHL Partnership, 2011). However, as there are seven large BOAs in the NCA they could not all fit into the NIA. NE3 discussed how a criteria was used to choose which to include, partly based on the partner's knowledge of biodiversity, but also indicated that there were many other (unspecified) factors involved. By including these areas they wanted to get biodiversity into the HHL NIA, but again by using areas where they had already been working.

It appears that the role of biodiversity in the HHL NIA boundary selection was through the inclusion of areas that they had identified in the past as important for biodiversity, such as designated sites and BOAs, as starting points. The use of these areas introduces the idea of conservation legacy. Conservation legacy is defined in this work as the use of areas previously worked on for conservation, which were then used as part of the NIA, the reasons for which will be explored more throughout this chapter. In this case study biodiversity is often involved through conservation legacy and knowledge which is subjective/opinion orientated. This contrasts with the data focussed approach of early chapters in this thesis. Therefore, biodiversity did have a role, but it was alongside other factors, such as conservation legacy. There was also an element of practicality involved, partly because of the area restrictions.

6.3.1.2 Other factors in the boundary decision making process

As well as biodiversity there were many other factors that went into the decision making process for the HHL NIA boundary and the major ones are covered here. As well as the major sites discussed in the previous section, there are also many other smaller areas and projects that were considered. However, compared to the larger sites, which were at least partly defined by biodiversity, the inclusion of these smaller ones appears to be due to much more practical reasons. These included smaller designated areas and visitor centres that different partners and organisations wanted to include, partly based on the idea that the Partnership should make use of areas that they own or have control of. The idea of including visitor centres, such as Potteric Carr, was discussed by several interviewees, such as NT1 and YT1. As well as these being sites they already controlled, they highlighted that part of the reasoning was so that the HHL NIA encompassed sites for “*environmental education and community awareness*”. This introduces social based reasons as well practical considerations. YT1 described the inclusion of the sites as the reason why the HHL NIA boundary has jagged edges, why the drawing of the boundary appears odd without the context. The idea of using gateway visitor sites was also mentioned in the HHL NIA Business Plan (HHL Partnership, 2011), giving more support to these discussions.

The idea of using previous sites also included the two NNRs (see Figure 3.1). NE1 described these as “*the top assets if you like in the NIA*” and said that their inclusion was based on the practical consideration that this was land they already had control of. Their inclusion was also discussed by NE3, who said that the wider ethos of the NIA Programme is not just about nature, but also people’s access to and engagement with the environment, which the NNRs provide. This point implicitly refers to the concept of ecosystem services, through cultural services such as engagement, but was not necessarily seen that way in the discussions. Overall, the NNRs, and other smaller sites, were an obvious inclusion within the boundary, for a range of different reasons, including biodiversity, ecosystem services and conservation legacy, even if these terms were not always explicitly referred to by interviewees.

Many of the decisions made, including those already discussed in this section, appear to have been based on specific interests that particular partners or organisations wanted included within the boundary. For example, YT2 talked about the inclusion of Fishlake because of work with habitats and water voles, which was an interest of Yorkshire Wildlife Trust. Each organisation will have had their own priorities, some of which will be linked to biodiversity, but not all, causing biodiversity to have a varying role in the process according to the interviewee responses. There were also requirements imposed by each organisation, as discussed by YT1, and so negotiations and compromises were part of the process. A related factor was the idea of having a wide range of people involved, which meant that more skills, experiences and knowledge were brought to the partnership, so that there was more evidence for decisions to be made on. However, as there were many people involved it appears that there was a reliance on expert opinion and that evidence was rarely looked for outside the group. Such an approach means that the boundary selection was a more human driven process, because it was based on the knowledge they already had and how it was used, rather than looking for more objective data.

One factor that appeared to have a major role in many of the decisions is agriculture. This factor could be seen as driven by the concept of ecosystem services and biodiversity, but was not discussed this way in the interviews or Business Plan (HHL Partnership, 2011). The area restriction to the NIA funding

meant that the partnership had to be very selective, and there was much agreement over not including the highly productive agricultural land. Figure 6.2 shows the most productive agricultural land, Grade One and Grade Two. YT1 pointed out that one of the main reasons for not selecting much of this land is that they would not be able to do much with these sites, and so there was no point including them; a very practical reason for making this decision. NT1 discussed the high productivity of the landscape for farming, the value associated with the land and the unresolved tensions between agriculture and biodiversity. Overall, the discussion around agriculture was a large part of the boundary decision making process that most interviewees agreed on, linking to the idea that the partners did not have control of this land and due to the land prices would struggle to ever do so. The issue of high grade agricultural land also came up in the Business Plan, which talked about the juxtaposition between agriculture and wildlife habitats (HHL Partnership, 2011), and so practically validating these discussions in the interviews.

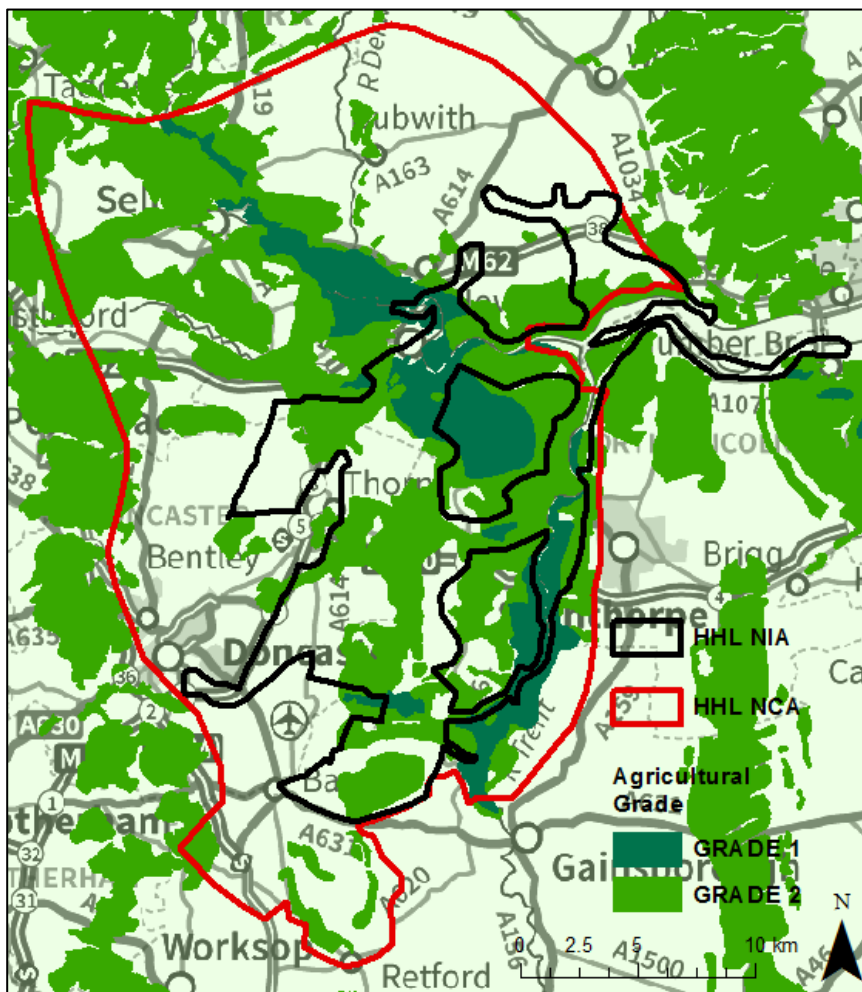


Figure 6.2: The Grade One and Grade Two (best levels) agricultural land within the HHL NCA.

Through some of these factors already discussed and many other parts of the HHL NIA boundary decision making process, an element of practicality often plays an important role. In some of the interviews the word pragmatism was used; practicality seems to be what was meant. Pragmatism is defined by the Oxford English Dictionary as “attention to facts, as opposed to opinions, ideals, or emotions; realism” (Oxford University Press, 2016). However, it has already been discussed that they did not play much attention to the facts and so their own definition of pragmatism appears to be more about practicalities and opportunistic elements. One of the main areas where practicality comes in is with the size restriction. YT1 discussed how actually they wanted to include much more land and they had to think about what sections not to include in order to meet the NIA criteria, as the boundary had to include land they could work with. For example, this factor would have contributed to the jagged edges of the boundary in order to include certain areas without wasting the limited space on parts of land of less value to the partnership. Therefore, practicality had a large role in the decision making process, possibly as big as biodiversity.

6.3.1.3 Summary of the boundary decision making process

The decision making process for the HHL boundary appears to have started on a large scale with the partnership thinking about the major areas to include, such as the BOAs. The next step was the inclusion of the NNRs, followed by many of the other smaller sites and designations, including visitor centres and specific wishes from certain partners and organisations. Therefore, the idea of conservation legacy came through several times, both in discussions about biodiversity and the other factors. Due to the area restrictions of the funding there was also much debate surrounding which areas not to include, mostly the productive agricultural land. Biodiversity appears to have been involved in many aspects of the boundary discussions, but often alongside other factors, even though in some cases interviewees believed it was the only factor involved. There were some ideas of how they choose the boundary in the Business Plan (which was written after the boundary decisions were made) that did not come out of the interviews. This included using the physical characteristics of the landscape, such as the river networks and the soil types that were most appropriate (HHL

Partnership, 2011), which must have been quite important but perhaps not so obvious to the interviewees.

The idea of practicality appeared very important to the decision making process, as is demonstrated by JA1 who stated that *“if you’re writing a pure plan for the NIA completely with a blank slate you would not necessarily write it in either of those ways, because you have got the culture filter in between”*. Therefore, practicality had a necessary role alongside biodiversity. In the Business Plan it was stated that there was both ecological and pragmatic elements to the boundary decision process (HHL Partnership, 2011), which were seen in the interviews as well. Therefore, many of the discussions in the interviews can be backed up by the Business Plan.

An important aspect to the objective of this chapter is to look at the role that the concept of ecosystem services had in decision making, but at no point during discussions over the boundary was the concept mentioned, even in the Business Plan. Many times in the interviews it was discussed that the practical application of the concept of ecosystem services is still relatively new and so therefore does not have a major role in decision making yet. However, there were cases where services were discussed, but not recognised as the concept, such as with cultural services.

Overall, the decision making process of the boundary had many different factors interacting. Biodiversity was important, but this was considered alongside: existing conservation sites and projects within the landscape, but it appears these were sometimes indirectly influenced by biodiversity as well. These ideas will be used for the rest of this chapter examining the processes in more depth. The ideas of conservation legacy and practicality seem particularly important to continue investigating. The interviews and Business Plan were very useful for understanding the boundary process, but the meetings were not used because I did not start attending until a few months into the HHL NIA, when these decisions had already been made.

6.3.2 The role of biodiversity in the HHL NIA

In the boundary case study, it was found that biodiversity was involved in many aspects of the boundary decision making process, alongside side other factors, although some interviewees thought that biodiversity was the only factor involved. Here the role of biodiversity in the general decision making, which mostly occurred after the boundary decisions, is explored. Biodiversity in some cases in the literature and policy is considered to be an ecosystem service, but as stated in the first chapter this thesis takes it as a separate concept. This was also the case for the Partnership where only one minor instance was found where they referred to biodiversity as a service. Biodiversity was mentioned many times in the interviews, meetings and documents, with much variation in its perceived role. Timing could be important as the decisions made about the NIA boundary preceded the focus of the decisions discussed from this point.

Several interviewees thought that biodiversity did have a key role in the HHL NIA, including YT1 who stated that biodiversity “*drove the essential need for the project...were absolutely central to way we thought the project was going to be delivered*”. The word “*crucial*” was used by NE3 when discussing the role of biodiversity, but also conceded that it was a difficult area to work in. In the Business Plan, biodiversity is mentioned many times, including that biodiversity conservation was one of the main projected outcomes (HHL Partnership, 2011). Therefore, there were some strong opinions that biodiversity was central to the NIA project, with the passion for biodiversity obviously a driver of its inclusion.

There were also many discussions amongst the interviewees, of decisions where biodiversity was seen as important, but not as the main influence in the process. Both IB1 and YT2 suggest that everything in the decision making process had to have an element of practicality alongside the important biodiversity principles, so that what was put forward was achievable within the time and resources available. An important factor when considering decisions where biodiversity was thought to be involved is that the biodiversity knowledge used appears to have been based on what the different partners knew, and EA1 stated this could be subjective. In the Business Plan and meetings there were many discussions of biodiversity, but these again were often tempered by more practical issues, such as site ownership (HHL Partnership, 2011).

There were some interviewees that suggested biodiversity did not have a key role in decision making. Examples can be drawn from both JA1 and YT1, who suggested that decisions are made on what individuals wanted to achieve and that biodiversity *“gets pretty horrendously overlooked”*. However, YT1 does have some hope for the future, stating that the role of biodiversity is getting better. These sorts of discussions were less obvious in the Business Plan, but were apparent in some of the meetings.

Discussions suggesting that the role of biodiversity, whatever it was, was appropriate were in the minority, but there were some. For example, YT1 had a very positive response about biodiversity and stated that *“if you were to take all other aspects out and just leave the biodiversity elements would this look the same, yeah probably, it wouldn’t be far off”*, therefore suggesting that biodiversity should and did have the main role in the HHL NIA. The quote comes from an environmental NGO employee who was involved in the decision making and so therefore may have wished to convey that the process represents biodiversity well. However, in the Business Plan the main aims and outcomes also centred on biodiversity (HHL Partnership, 2011).

Those who argued that biodiversity was partially involved in decision making, but should be more so, mostly discussed broader decision making rather than specifically the HHL NIA, but interesting points can still be considered. For example, NT1 was extremely negative about the role of biodiversity, stating that it is definitely not at the appropriate level in governmental decision making and it needs to change, but they did not offer what else they thought was involved. RB1 discussed a paper that concludes that site managers do not really use scientific evidence in decision making. Both interviewees suggest that the scientific evidence required to make decisions is not always available, as was found in the literature (Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015), and potentially why biodiversity is not involved in decision making more. This is interesting because in the HHL Business Plan they discussed working closely with the local record centres to provide data to support decision making (HHL Partnership, 2011), but it appears in practice (from the interviews and meetings) that such working was more difficult than they were expecting.

A range of reasons were given by those who thought that the level of influence of biodiversity in the HHL NIA decision making was not at all appropriate. RB2 suggested there is a lack of knowledge of biodiversity, which has lessened its inclusion. RB1 goes further in suggesting that this lack of knowledge is often as a result of not enough scientific evidence to make decisions, as found in the literature (Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015). Alongside a need for more research, NE1 suggests the way to increase the involvement of biodiversity is to highlight the other benefits that will come out of biologically based decisions. Perceptions about the role of biodiversity in the HHL NIA varies amongst the interviewees, but the three sources of information show that it did play some role, but with many other factors involved. Below specific examples will be considered to better understand the role of biodiversity in HHL NIA decision making.

6.3.2.1 Examples of how biodiversity influenced the HHL NIA decisions

The previous section argued that biodiversity considerations were seen as central in the NIA process for some partners, but not for others. Here the question of when and how principles of biodiversity played into HHL NIA decisions is addressed, alongside some discussion of the other factors involved. The role of biodiversity in landscape scale work was discussed by many interviewees, especially in terms of the Lawton principles (Lawton et al., 2010) and connectivity, an important element to the NIA Programme. The idea of landscape scale work was stated clearly by NE2 who talked about using existing core areas in good condition to act as ecological hubs to work out from. This idea was taken further by YT1, who stated *“So I think, yeah, for me the idea of landscape scale approach is making sure that we have both elements, we need the critical sites and we need the other Lawton based ideas to take forward the new approach essentially”*. The idea of using core sites of biodiversity, specifically the two NNRs, also came up in the Business Plan (HHL Partnership, 2011), and so from using these two sources of evidence it appears that some important principles of biodiversity under the landscape approach were included in HHL NIA decision making.

The concept of connectivity in relation to biodiversity was discussed by several interviewees. IB1 stated that it was about *“trying to connect the projects, the right*

projects that we [thought had] potential” and NE1 discussed how connectivity worked on a large scale within the landscape, having buffers to protect the main sites, and then connecting these up. There were also other interviewees who discussed corridors to allow movement between core sites. Connectivity was also discussed in the Business Plan, wanting to connect up various sites and habitats to allow movement across the landscape (HHL Partnership, 2011). Therefore, this part of the decision making process shows how biodiversity did play an active role within the HHL NIA. However, it had a main role because of previous work included to achieve connectivity and the Lawton principles, so the idea of conservation legacy comes up again rather than the current key areas of biodiversity.

There are regulations and other requirements that influenced the role of biodiversity in decisions, as some of the organisations in the HHL Partnership are required legally to carry out certain actions that are beneficial to biodiversity. These can come from policy, which was identified to be an issue in the interviews as these often change when a new government is voted in. There were also requirements that institutions place on themselves, which NT1 discussed in terms of the Wildlife Trusts, stating that they put biodiversity at the centre of everything they do. Such organisational emphasis on biodiversity will have been the case for some of the other HHL Partnership organisations, but not all, which will be investigated further later in this chapter. However, it seems that governmental and organisational requirements had a major influence on what role biodiversity had in the HHL NIA decision making, as there was also evidence in the meetings and the Business Plan.

The knowledge and expertise that members of the different organisations within the HHL Partnership have about biodiversity made a difference to the decision making process. NT1 states that much of the HHL NIA was based on the partners’ ecological knowledge, skills and experiences, with YT1 adding that decisions were often *“ecologists/conservation practitioners making judgements based on what they know and based on what they think the outcomes are likely to be”*. By using this knowledge it means that biodiversity will have been involved in decisions. However, there is an opportunistic side to the process, as YT2 discusses that the type of knowledge available depends on whether a particular

expertise is represented by the staff, which will then be fed into the process, rather than looking for knowledge when it is required or planned for. They did not appear to go looking for biodiversity knowledge outside of the partnership, which was also something observed from the meetings. Therefore, the role that biodiversity played in the decision making process was often influenced by the particular knowledge of individuals involved, which confirms what was already observed in the boundary case study.

From the interviews and the Business Plan it appears that a great deal of the work carried out was driven by particular species. NE2 discussed how species are important within the landscape stating that they are the “*the building blocks of biodiversity*”. NE2 uses the Scarce Vapourer Moth as an example of a species found in the HHL that is rare in the UK, and discussed how the partnership protected this for the future. In the Business Plan the term “species” is mentioned many times, discussing the idea of iconic species to the landscape and how the HHL is important biogeographically, as some species are at their northern limits and some are at their southern extreme (HHL Partnership, 2011). This evidence illustrates that particular species, and therefore biodiversity, were at the forefront of parts of the decision making process.

The process of selection for species that were explicitly incorporated into decisions was described by YT2. Selection was connected to the public’s ability to see or be aware of that species, or as discussed previously, according to partners’ priorities or expertise, which was backed up by discussions in several meetings. For example, in a meeting on the 26.02.13, the monitoring of species was discussed and decisions were partly based on what was already being carried out, with several examples given (e.g. water voles, bittern). This was continued in a meeting on the 08.11.13 where species were discussed in relation to data that could be accessed and those that show a response to habitat change, using knowledge in the room. Therefore, these decisions were less biodiversity driven, and more about human perceptions and subjectivity. Overall, species have an important role in the decision making process in the HHL NIA, both at the scale of individual sites and across the landscape, but there were other factors involved in these decisions, not just biodiversity.

The role of NNRs in the decision making process was partly biodiversity driven, as discussed in the boundary case study. NNRs were mentioned by many interviewees, including NE3 who stated that *“One of the core objectives...is to get the existing internationally important wetlands, which are the Humberhead Peatlands, into favourable condition”* and so looking at important habitats in the landscape. However, as discussed in the case study the interviews also brought up the more practical side of NNR inclusion, that the land is already owned and worked on by the partnership organisations and it is always easier to continue what has been already started. Therefore, for the NNRs the role of biodiversity was positive in the decision making process, but practicalities, such as land ownership, also had a big influence. Through this work on NNRs and other issues there appears to be an importance placed on conservation legacy; the partnership using and developing areas where work was already occurring, which were initially chosen partly due to biodiversity reasons.

6.3.2.2 Summary

The aim of this section was to understand the role of biodiversity in the HHL NIA. As with the boundary case study, interviewees had varied opinions about the role of biodiversity, however there was actually a fairly consistent picture. The evidence suggests that, as with the case study, biodiversity did have a role in the decision making processes, but often alongside and interacting with other factors. It appears that sometimes key scientific principles of biodiversity conservation were very much embedded in the HHL NIA decision making process. Examples include the use of the idea of “more, bigger, better, joined up” (Lawton et al., 2010) part of their whole ethos for the HHL landscape.

There were a variety of other factors that had an influence alongside biodiversity, found in all three sources of information, some of which are indirectly impacted by biodiversity. It appears that the idea of real-world management does influence the role of biodiversity. What the partnership actually aspired to achieve on the ground was tempered by practical considerations, for example conservation legacy, that in fact always need to be considered as well. A very consistent picture has been built up about biodiversity, with little change in the role over time, except for some introduction of landscape scale work. This is probably because there

are some very well-established practises, such as species specific activities, and so they still wanted to use those approaches, even with a landscape scale project.

6.3.3 The role of ecosystem services in the HHL NIA decision making process

The role of ecosystem services in the decision making process of the HHL NIA is extremely interesting, especially considering it is a relatively new concept that is discussed extensively in the literature, but there are still many unknowns in relation to its delivery (Wu, 2013; Pan et al., 2013). The concept of ecosystem services was discussed in Lawton et al's (2010) "Making Space for Nature" document as a way of thinking about the multiple benefits of the environment; an important part of conservation work. Here discussions centre on the role of ecosystem services in two ways; the use of the concept, and through the actual services in the HHL landscape. In the boundary case study, the concept of ecosystem services was not explicitly mentioned by any of the sources, but there were some cases where services were discussed, but they were not recognised within the ecosystem services framework.

Some interviewees believed that the concept had a role in the decision making process, with a particularly positive statement from EA1 of "*Yeah, certainly*". RB2 was a little more reserved in stating "*A little bit, I mean its obviously more than it was a year ago*". NE2 specifically talked instead about the range of ecosystem services in the HHL. A couple of interviewees discussed ecosystem services in terms of economics and how that could be a big draw of using the concept in decision making, but this appeared not to be a major part of their decision making process. The Business Plan states that the HHL provides a wide range of ecosystem services, and so they explicitly use and are committed to action with the concept (HHL Partnership, 2011), although this was written at the beginning of the process. However, there were many cases where services were mentioned, but not explicitly linked to the concept in the HHL Business Plan. For example, there were five key aims, of which two mentioned services implicitly (HHL Partnership, 2011). Therefore, there appears to be some evidence of the use of both the concept and the services in decision making.

There were also those who thought that the concept of ecosystem services in its current form had no role in the decision making process, but that it had always been there in the form of individual services, seen as physical characteristics or functions. This was discussed by both NT1 and YT2, who suggested that ecosystem services had always informed their work, but not recognised under the concept. NT1 identified that the current branding of the concept is just a repackaging of what has always been happening with physical characteristics in the landscape. However, YT1 reasons that this repackaging is good for decision making: *“So for me ecosystem services, it has a benefit of creating a rather stronger utilitarian view of biodiversity conservation. And in the modern decision making climate that’s probably a good thing”*.

Others thought that the concept had been considered in the decision making process, but they were not sure how to deal with it and so often it was just in more of the wording of the Business Plan rather than in any delivery decisions. Evidence came from YT1 and JA1, who discussed that it was more about putting the concept into decisions as an afterthought, mainly to fulfil NIA criteria, but also because of the increasing popularity of the concept. This suggests that decisions were not informed by ecosystem services, but that they were made and then put under the concept, because they felt they had to. There was a lot of variation between the various interviewees and the Business Plan over the role that ecosystem services had, mostly as a concept, but it appears that there was more agreement about the inclusion of the actual services.

There were a few explanations given by the interviewees as to why ecosystem services as a concept did not generally have a role in decision making. IB1 thought that it was a useful concept, but that they had not been given enough time to think about how it should be delivered and embedded. This was backed up by NT1, who stated that it was a very complicated area to deal with. Also, in a meeting on the 26.02.13 the partners talked about selecting ecosystem services indicators for Monitoring and Evaluation, based on what they knew or where they could get existing data from, rather than looking for new information. They did mention in their Business Plan that they had discussions about the use of ecosystem services with York University (HHL Partnership, 2011), but there was no further evidence on whether this occurred or was used.

YT1 explained that the concept is very general and does not take into account individual species and habitats that most people who work in this field are more concerned with. It is a very different approach to biodiversity conservation, which is the focus of many of the partner organisation's work and has been their legacy for a long time. Therefore, it is more difficult to get people involved to take up the concept and include it in decision making. However, others have suggested that there have been struggles in delivering conservation in terms of biodiversity, and so we need to find another approach, suggesting that this concept may be the way to work going forward.

It appears that the concept of ecosystem services had limited influence in the decision making process, for a range of reasons, but that the actual services did have some role. However, there does appear to be some evidence of the concept having more influence over time, with more inclusion here than in the boundary case study. Clearly, the lack of knowledge and understanding is a barrier to the implementation of the concept, but it appears they do have a good grasp of some of the services the landscape can provide.

6.3.3.1 Examples of ecosystem services in decisions

There are a range of areas in the broader decision making process where the concept of ecosystem services appeared to play a role, in contrast to the boundary case study. One particular example comes from discussions around the peatlands in the NNRs, which are extremely important areas to the HHL. NE1 and YT2 discussed services that were involved in decisions surrounding the NNRs, including carbon storage and cultural services. It appears that these decisions were made with biodiversity in mind too, because both of these interviewees also discuss the bog/peat habitat. From the interviews it appears that the peat was one of the main areas where ecosystem services were part of the decision making, perhaps because they had a better understanding of the services delivered here, alongside the ideas of biodiversity and conservation legacy already discussed. However, in two meetings (26.02.13 and 08.11.13) there were problems arising with how to monitor the services, with suggestions

of using university research and modelling, but there does not seem to have been an answer, and they felt any measurements would be very “broad brush”.

In the Business Plan a wide of range of ecosystem services were mentioned, including flood management, food provision, recreation and in particular carbon sequestration, which was described as a “vital” ecosystem service (HHL Partnership, 2011). There were also services mentioned in the Business Plan that were not explicitly linked to the concept, such as the green economy, tourism and health benefits (HHL Partnership, 2011). This implicit use was also seen in the interviews and meetings where some services were never connected with the concept, and some were only part of the time, but it is difficult to understand why this is the case for particular services.

Cultural ecosystem services were often mentioned in many of the interview discussions and it therefore appears that they had some role in the HHL NIA decision making process. They are thought to be the most difficult to deliver and measure (Crossman et al., 2013; Hernandez-Morcilloa et al., 2013), but several interviewees clearly discussed the value of the landscape to people. For example, IB1 states that “*there is a value for a person walking in the dog down the side of a water course, there is a value for how much pleasure they get out of it, for the well-being that it brings, the hospital visits it prevents, the crime it prevents*”. Therefore, this quote is saying that cultural ecosystem services are important, but may not be explicitly considered under the concept. NE2 and YT2 do point out that more information is needed on how to measure these cultural services, which would have caused them some problems in using them in decision making, due to a lack of evidence and guidance.

Whilst not explicitly recognised as “ecosystem services”, the HHL Partnership did embed cultural services into their decision making process by developing the Connect project. This project aimed to engage more people with the environment at visitor centres and gateway sites, through outreach and volunteering. This was involved in several meetings (including on the 03.03.15) where they discussed having nine sites with these activities, which was successful in their eyes. This was one of the main initiatives as part of the HHL NIA and again shows a case where the services, but not necessarily the concept, had a role in the decision

making process. This project was also mentioned several times in the Business Plan (HHL Partnership, 2011), but not always explicitly linked to the concept of ecosystem services. In the later meetings this was the ecosystem service most discussed, perhaps because they had achieved something with it.

Another project that took into account the concept of ecosystem services was the biomass project. This project, run by the RSPB, harvested reedbeds to produce briquettes, which is a provisioning service, providing raw materials and developing the local green economy. Here the use of the concept was from an economic point of view, as it involved selling an item to make money to further fund restoration. Even though the economic view is popular in the literature this was one of the few times where ecosystem services and economics were discussed together with the HHL. RB1 believed that this is also the only project that actually takes into account the concept of ecosystem services, which is interesting considering it is about economic value. Perhaps some partners only see the concept from an economic view, but no further information could be gleaned from any of the sources.

6.3.3.2 Summary

Overall, ecosystem services did have some role in the HHL NIA decision making process, but it was not as clear as the role of biodiversity. There are many difficulties that lie around the concept, due to a lack of understanding and information, which need to be overcome if it is to be used more. There were cases where the HHL Partnership talked about the concept of ecosystem services, but did not do anything with it. There were fewer cases where they talked about the concept and used it, and many cases where they used the actual services but not recognised under the concept. Therefore, ecosystem services have been part of conservation for some time, but just not named that way. In the boundary case study the concept was not mentioned, but services were discussed. This did occur in the broader decision making but there were also instances where the concept was used. As the boundary discussions obviously preceded the rest of the decisions it could be argued that the role of the concept grew as the project went on, as they further understood the concept or thought there was more of a

need to include it. It appears that the role of the concept of ecosystem services was very superficial at first and partly about doing what they wanted anyway. Over time the concept appears to have played more of a role, with new ideas built in surrounding it.

There were many discussions over this concept having more of a role in the future; that its influence is just going to grow and grow in decision making. Several interviewees discussed that in particular it could be useful to help with some of the challenges in the landscape, such as flood protection. However, it appears there were some tensions over the increase in the use of the concept, because using it could lead to a change in current conservation practice and the organisations that traditionally work on biodiversity did not like that, and so there is some wariness over the use of the concept. Those that were more accepting say that it needs to become easier to deliver on the ground, with more scientific evidence and guidance. Overall, the role of ecosystem services, rather than the perception of it, varies across the decision making process, more in the concept than the actual services, because of how new it still is.

6.3.4 What other factors had a role in the HHL NIA decision making process?

In the boundary case study there were many other factors that influenced the decision making alongside biodiversity (and some services), and in many cases it was several factors interacting to make a decision. There are also many other factors that were not in the case study. Here the major ones identified from the three information sources will be covered, including partnership working, the partners and organisations involved, the different projects included, the role of funding and other resources and practicalities.

6.3.4.1 The role of the Partnership in the HHL NIA decision making

An understanding of how the Partnership works is an important component of comprehending the HHL NIA decision making process, as it provides context as to how the different factors fit together. Lawton et al. (2010) identified in “Making Space for Nature” that partnership working is important for thinking about the

multiple benefits for the environment. The background of the Partnership can be found in Chapter Two, but key points can also be drawn from evidence here.

NE3 (in a role as the Partnership Secretariat) discusses much of the working of the Partnership at the beginning of the NIA process. There was a reworking (because of a learning process) of the group's objectives to something more delivery driven, which the partners had to come to a consensus on, before embedding them into the Business Plan. The Partnership commissioned another individual from the YWT to act as a coordinator for the NIA Business Plan, to include everyone's input. NE1 states that "*because the Humberhead Levels Partnership being an established partnership we had already done quite a lot of ground work in bringing the partners together around a common vision, aims and objectives*" therefore making the whole process easier. At the beginning of the HHL NIA all the partners were also asked to sign a Memorandum of Agreement agreeing on the governance and management processes (HHL Partnership, 2011).

One of the major ways that the Partnership that made a difference to the way decisions were made was the management structure. NE1 pointed out that "*the challenge for the Humberhead Levels Partnership has been the governance*". One difficulty was with the position of Programme Manager (PM), specifically with what the role should entail and who should do it. YT1 talked about how the group decided that having an NIA PM was important, even though several other NIAs did not appoint one. It added a cost, but also helped to keep the projects together for a sense of "*coherence*". The problem was in keeping PMs, as there were three over the period of the project, which was discussed in several meetings. YT1, EA1 and NE3 talked about having the right skills; needing to be good with people, a strong manager and dealing with the administrative side, as the role is more "*facilitatory*" rather than about delivery. With the earlier PMs there was a feeling that the role was not quite what they had expected.

Another difficulty that influenced the PM, and therefore decision making, is the structure of the Partnership working groups. At first they had a large Management Group that IB1 believed led to too many "*compromises*" with decisions. So they created a smaller core Management Group of around six

members, in order to provide support to the PM and allow decision making to be easier and more efficient. In a meeting on the 15.08.13 several partners discussed the suggested reorganisation of the governance groups. Some of the delivery partners were wary of not having a forum to discuss projects and of who would be involved in the new core Management Group. However, the changes did go through and were discussed again in a meeting on the 06.05.14. They believed that the previous structure restricted how decisions were made, with more compromises occurring and decisions often slow due to the PM changes, which suggests that the partnership element did not work well. YT1 suggested that having multiple organisations is an inefficient way of working and that takes a long time to make decisions. However, in the Business Plan (HHL Partnership, 2011) they identified that the Partnership was a particular strength of the HHL. It appears that the governance structure changed over time, which impacted how decisions were made.

6.3.4.2 The role of different partners and organisations in the HHL NIA decision making

As well as partnership working, the people and organisations involved also had a large influence on the decision making process, which has already been identified through discussion over the role of biodiversity. EA1 stated “*You just need a room of like-minded people obviously*” but then goes into more detail with “*But it’s about finding partners who can a) commit to the primary aim and b) deliver something, a multiple benefit in there*”, a belief backed up by others. NE3 goes further and explains which partners were brought on board in order to help with delivery and expertise after the groups’ objectives had been developed, including the Wildlife Trusts, the RSPB, local authorities and the IDBs. An important consideration for the Partnership was that they wanted to make sure they were going to achieve something and that the Partnership was not just “*a talking shop*” (NE1), as “*The NIA should be what all of us are doing all the time anyway and it should be working more collaboratively and thinking in a more joined up area*” (NE2). One way they addressed this was by giving the main responsibility for each project to a different partner organisation, as identified in the Business Plan (HHL Partnership, 2011).

The interviews also highlighted that each of the partners and organisations had different drivers and agendas, so getting a consensus was challenging. NE3 discussed the problems of this: *“Because if they are not all signed up to it you end up with something wishy washy...you can’t really achieve what you set out in the Business Plan”*. Most interviewees however are generally in agreement about achieving something good with the NIA. IB1 stated that *“if we all sit down and talk about stuff we actually find out that 90% of what we want is very similar anyway and if we can compromise on the other 10-20% we are way in front of doing it on our own”*. However, having more partners will have also increased the complexity of decision making due to their different ideas and requirements, possibly making the process longer and more difficult.

The range of organisations involved included environmental QUANGOS, NGOs and organisations who are not primarily focused on conservation, but have other priorities in managing the natural environment, such as the NFU and IDBs. Also, the HHL is not isolated, the wider roles of the partner organisations will also have had an influence. In some cases through the interviews, meetings and documents it was clear that the different types of organisations have different roles or stances in the decision making process, but the role of individual organisations rather than the type of organisation was much more obvious. Some organisations were more focused on biodiversity, whereas others had a wider focus that also included ecosystem services. Some organisations have another focus outside these concepts, such as the NFU with agriculture. Attending an early meeting on the 15.11.12 it was quickly obvious who had certain views on the NIA project as a whole, with organisations like the YWT fully involved. However, the NFU also had their own member’s views to consider alongside the NIA Programme, and so were more cautious about the activity of the HHL NIA, but still wanted actions to occur.

6.3.4.3 The role of different projects/sites in the HHL NIA decision making

Within the HHL NIA Business Plan many projects from individual organisations were included, which was a big part of the decision making process. YT1 stated that *“to be honest, that map we were trying to create a sense of busyness so it*

looked like the whole area was going to have something significant going on within it". This point shows the huge anthropogenic influence over the prioritisation of areas under the NIA, which has come out in previous sections, but this quote really makes it very clear. Most of the organisations within the partnership wanted their own ideas and areas represented, and so trying to please many of them would have been a big part of the decision making process, which was discussed by several interviewees, linking back to the factor of having different individuals and organisations involved. The idea of including different projects and sites was also covered in the case study and biodiversity sections, along with using conservation legacy. This links to the challenge of deciding which areas to exclude from the NIA, with discussions centred on the exclusion of certain parts of the very productive agricultural land, which was also identified in the HHL NIA boundary case study. NT1 discussed the tensions between agricultural production and biodiversity, and the value of the land, which was also mentioned in the Business Plan (HHL Partnership, 2011).

6.3.4.4 The role of funding and other resources in the HHL NIA decision making

In the interviews discussions the role of funding was seen as extremely influential because it was often the reason that any work even occurred in the first place. In fact, the HHL Business Plan says that without the NIA or any other funding the partners would just revert back to "mostly minimal maintenance work" (HHL Partnership, 2011). However, they have found it difficult to make sure "*that that direction of travel matches the funding programme*" (IB1), which meant that attempting to meet the funding programme appears to have influenced the decision making process.

There were issues with the amount of funding that the NIA scheme provided; both NE1 and NE2 discussed how small the amount was and therefore limited what could be achieved, as restoration is an expensive activity. This theme was discussed in a meeting on the 15.02.13, where the short termness of the funding (3 years) meant they would not be at a financially sustainable point by the end and so would struggle to continue without more outside funding. In fact, spending was a common theme in many meetings, but with often a focus on how to use

underspent budgets, which was frequently about where was ready to go rather than any other reasons. Therefore, it appears that money influenced the decision making in both a positive and negative way, in some cases providing more money but also practically tempering what they wanted to achieve. This point also helps to explain the importance of why conservation legacy is involved, there were not enough funds to start new projects from scratch, and so more can be achieved if they built on existing projects.

There are also other resources, such as time and data availability, that had a central role in the decision making process. Time was discussed by many of the interviewees, and in meetings, such as the 08.11.13, where it was said that actions were based on whatever time people had. RB2 makes a very valid point that "*the NIA was created out of a partnership of people who already had full time jobs*" and so people could not give too much time just to the NIA. One interviewee discussed only getting involved if there was a problem, which could have really affected decision making. The other issue with time was there were only three years and so there were restrictions to what could be achieved, a big consideration in the decision making.

Often there was not the data or scientific evidence to back decisions up. Many of the interviewees also believed that there was very little guidance provided for the programme from Defra, making it difficult to know where to start and what they were required to achieve. For example, this was brought up in a meeting on the 26.02.13 where they discussed the lack of guidance on Monitoring and Evaluation indicators. The need to work with the records centre was indicated in the HHL Business Plan (HHL Partnership, 2011), but it appears that this was difficult for reasons that were not made clear. Therefore, there were issues with time, scientific evidence and guidance, but NE1 stated that partnerships are actually the most effective way of working, as they allow collaboration of knowledge and resources, to overcome these issues in decision making.

6.3.4.5 The role of practicality in the HHL NIA decision making

There were many ideals of what they hoped to achieve in the HHL, but an element of practicality always needed to be applied to avoid plans being developed which were not achievable. This was also discussed in the boundary case study, where

in the interviews they used the word “pragmatism”, but by definition really meant practicality. It appears from the interviews that this factor could have played a large role, as it was discussed by many interviewees. JB1 suggests that if a plan was written that did not involve this factor then the NIA would not look the way it does now. The idea of practicality came through in discussions over the other factors involved and the NIA boundary, such as funding and the size limit, and is important to consider when trying understanding real-world decisions. This factor was not really mentioned in the documents and meetings, but as it is more about common sense it is not surprising that it had a role in decision making. In fact, it seems like an umbrella term for many of the other factors, and so influencing many decisions.

6.3.4.6 Summary

There were many factors involved in the HHL decision making process alongside biodiversity and ecosystem services, some of which were covered in the previous sections, which mostly seem to come under the idea of practicality. The range of factors include those explicitly titled in each section, such as the partnership management, the partners and organisations, projects and sites, funding and resources and practicality. There were also other important factors that came out of the discussions, including conservation legacy, lack of knowledge and agriculture. All these tempered the role of biodiversity and the concept of ecosystem services in the decision making process, as it was rare that decisions were ever made on individual factors, it was more in a combination. Some may say that these factors tempered the idealistic adoption of biodiversity and the concept of ecosystem services, but actually these practical issues are very important for attempting to make realistic and successful decisions. This was the same pattern as was found in the boundary case study with other factors, and so can be argued that there is a general pattern of many different factors being involved and the idealistic role of biodiversity and ecosystem services being very difficult to achieve, if not impossible, because it is impractical in the real-world.

6.4 Discussion and conclusion

The objective of this chapter was to identify the decision making processes used in the HHL NIA, in particular the role of biodiversity and the concept of ecosystem services, using interviews, documents and meetings. It was found that the decision making process NIA involved many factors alongside biodiversity and ecosystem services, which was not a surprise considering the literature (e.g. Maltby, 2010; Cotter et al., 2014). Also, each decision made did not appear to have a particular schedule to follow, each process varied hugely, with often the available knowledge or existing work defining what occurred. Drawing on some of the literature discussed at the beginning of the chapter, the decision making process will be summarised, before discussing what can be learnt from this work. As also discussed at the beginning of the chapter, the different sources of information will have had different levels of reliability and so it is hard to produce definite conclusions here. What conclusions are drawn are not just based on one piece of evidence, ideas come from various sources to be more plausible, but comparing, not critiquing the process.

6.4.1 The role of biodiversity in the HHL NIA decision making process

Biodiversity appeared to be very important in numerous decisions, but there was much variation in how interviewees felt the concept was used. Decisions often started off with the main focus on biodiversity, which would then get tempered by more practical factors. Biodiversity principles were integrated in a number of different ways. Principles of connectivity and ecological networks were embedded, through increasing the size of the core areas and creating corridors. Previously much of biodiversity decision making was around more traditional approaches, with a focus on particular species and habitats (Carpenter et al., 2006; Fisher and Brown, 2014). Although this was occurring with the HHL NIA, there is also evidence of landscape scale approaches. There was a lot of evidence for the desire to involve areas of high biodiversity in the landscape, which appeared to include areas where work was already occurring, such as the NNRs and visitor centres. This idea of using the landscape's conservation legacy came out of several sections in the chapter. Many of these areas were originally selected based on biodiversity ideas (e.g. BOAs) and so biodiversity was partly

involved in these decisions, but with many other factors, as will be further discussed in Section 6.4.3.

Many of the organisations within the HHL Partnership are environmental NGOs and so biodiversity is one of the main factors that drives their work. The interviewees had varied opinions about the role of biodiversity, but the evidence from the documents, meetings and boundary case study shows a much more consistent picture of biodiversity having a fairly big role, but mostly alongside other factors, such as conservation legacy. It may have been that the different interviewees had different expectations about biodiversity and so some were happy with the HHL NIA process and some were not. Regen et al. (2005) discussed that there is uncertainty in conservation decisions and so it is not a surprise that this was evident in the way the HHL NIA used the concept of biodiversity. Overall, the role of biodiversity in decisions did appear to change over time with some movement towards landscape scale working, but there is also a lot of evidence of the use of the traditional well-established practices, such as site focused activities.

6.4.2 The role of ecosystem services in the HHL NIA decision making process

In the case of the concept of ecosystem services the role in decisions was not as clear as for biodiversity, partly as it is still in its infancy with many unanswered questions (Sitas et al., 2013; Wu, 2013; Pan et al., 2013; Albert et al., 2014a; Pagella and Sinclair, 2014). There were three different ways it was used in the HHL NIA decision making process: (1) talking about the concept but then not really using it, (2) talking about it and using it and (3) using the services in decisions but not recognising them as ecosystem services. In cases where the concept was used it was almost an afterthought, often fitted in to get ecosystem services into the Business Plan, probably because they felt they had to include it due to the NIA regulations and other policy, even though they were not confident. The partners considered the concept to be very new, and therefore still not yet fully understood, especially in terms of how to deliver it. They especially felt they had not been provided with the right information from either Defra or academics to deliver it, which is an issue recognised in the literature (e.g. Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015).

Where the Partnership did use the concept it was often with a service that had always been central to their decisions, e.g. recreation, but they did not see all the services they previously considered as actually part of the concept. Services that they saw as being part of the concept included flood mitigation, carbon storage and cultural services (although not all the time), and those that they did not include food production and tourism. In the future the concept of ecosystem services could have much more of a role in decision making as it becomes increasingly popular in policy (UK National Ecosystem Assessment, 2011; Crossman et al., 2013). However, when decisions were made about the HHL NIA the concept was not embedded enough to be used effectively in the process as the actual concept. In many of the interviews the partners thought they were struggling to deal with the concept of ecosystem services, but there were examples where they are using it implicitly, which is what they need to learn from and use in the future. However, the role of the concept in decision making did increase over the duration of the study; it was not mentioned within the boundary case study, then there was some evidence of superficial use, but by the end some new ideas had been built into the HHL NIA using the concept.

6.4.3 Other factors in the HHL NIA decision making process

There were many other factors besides biodiversity and ecosystem services that were involved in the HHL NIA, including the partnership, partners and organisations, existing projects, funding, resources and practicality, many of which are also discussed in the literature (e.g. Maltby, 2010; Cotter et al., 2014). Funding had a big influence on decision making. Many interviewees believed that the NIA funding was small and their ambitions had to be tempered to make lower cost decisions. A lack of other resources, such as time, data and Defra support meant that again the decisions made were not the ones necessarily wanted, but collaborating in a partnership did allow some of these to be overcome. Practicality had a huge role in decision making, potentially involved in many decisions. With people thinking realistically about what could actually be achieved in the landscape, decisions were made with a practical outlook, such as knowledge held and land already under their control, in order to be successful. The role of the

Partnership also appeared to be a key other factor, but this will be discussed further in the following section.

Conservation legacy came up many times, including in relation to biodiversity, and it therefore appears to have been a very important factor in decision making. There are several potential reasons for its inclusion, and it is probably a combination of all of them, as they interrelate. These are due to organisational priorities, as a starting point for work, because they do not want to waste their previous work, infrastructure and funding or because they do not know how else to do it, as well as having historical biodiversity reasoning. It could also be possible that these areas of conservation legacy were also included to allow partners to have their pet projects, to increase buy-in to the partnership.

One factor that was discussed more in the HHL NIA boundary section was the role of agriculture, as this made a big difference to decisions on where to include within the NIA, based on areas where they had control, what could fit into the size restriction of the NIA and the profitable nature of agriculture in the landscape. However, if they want to create ecological networks and move outside the traditional protected areas approach agriculture needs to be involved. Rey Benayas and Bullock (2012) state that “agriculture and conservation are in permanent conflict”, which was evident in the HHL NIA process. Agriculture was a major point of conflict with nature conservation and landscape scale working, but it does not appear that the Partnership really engaged with the NFU to overcome this. This factor is framed by the Partnership as a competing land use, but it might be better framed under the concept of ecosystem services to try and overcome the tensions.

Many of the other factors are examples of where practical considerations get in the way of biodiversity and ecosystem services, but actually they need to be considered alongside them, rather than seen as a hindrance (Maltby, 2010; Faith, 2012). The literature states that decisions are more human driven processes and are influenced by a range of non-scientific factors (Maltby, 2010; Faith, 2012), which was observed in this case. However, these other factors need to be acknowledged more as part of decisions, because they always occur and it might

lead to decisions that are more relevant for real-world implementation (Nhancale and Smith, 2011; Metcalfe et al., 2015).

From these discussions it appears that the three main other factors that were involved in the HHL NIA decision making was conservation legacy, agriculture and resources (including funding). These first two are obvious; conservation legacy came through in many of the discussions presenting a consistent story and even though agriculture was only mentioned in the boundary case study, it takes up such a large area of the landscape that it had to have a big role in terms of cost and land availability. However, as identified previously, rather than almost immediately dismissing it, agriculture needed to be integrated in order to truly achieve landscape scale work with the HHL NIA. There was less discussion over the resources factor, but this appears to be the ultimate practicality factor, because without money, time and people nothing would happen, and as the HHL NIA only had a restricted amount of all three this really tempered what they would ideally have liked to achieve in the landscape.

6.4.4 What can be learnt? And whom can it benefit?

It is hoped that any restoration project that occurs on a landscape scale with potentially a partnership making the decisions can learn from this work. Important points that can be drawn out include that there is much knowledge and passion for biodiversity that is involved in decisions, but often it is opportunistic, based on the staff already in place and the particular specialist knowledge they have. Rather than biodiversity decisions based on opportunity more forward planning should be carried out, first setting out what they want to achieve, then looking for the knowledge to support it. Previous chapters have demonstrated that there are data available to inform these decisions, but this is not being utilised, which is a key lesson from this chapter.

With the concept of ecosystem services, it is not known how to deliver many of the individual services in the landscape and how to deal with trade-offs between services (Wu, 2013). The reasoning behind the lack of understanding is that firstly, there is not enough scientific research and what does exist was not perceived to

be disseminated well from the academic community to the practitioners. The second reason was that not enough information was provided by Defra about how they expected ecosystem services to be dealt with within the NIAs, an issue with the programme in general. Many of the interviewees had worked on ecosystem services for many years, but not under the current concept format and felt that much more knowledge is needed before the concept could be successfully delivered. So there was some buy-in to the use of the concept, those that wanted to use it, but needed more information. There may also have been those who were using it as an excuse to not go towards a new way of working. However, because they have been using some services already it is a starting point, the concept just needs to be made clearer. In order to overcome the unknowns around using the concept and to highlight more of its benefits, a more holistic view is needed.

With the literature at the beginning of the chapter it was discussed that science (such as biodiversity and ecosystem services) should be in decisions, but at present this is often not the case (Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015). It was found in this work that these concepts did have a role, but biodiversity more than ecosystem services. The literature identified many reasons why the science is not utilised, which is mirrored here, especially with decision makers feeling they did not have access to research and that they do not always go looking for evidence (Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015). It was also found that other factors are needed as well, because without them decisions would not be made that are suitable for the real-world, which was also found in much of the literature (e.g. Maltby, 2010; Faith, 2012; Pullin et al., 2013; Dicks et al., 2014b; Cook et al., 2014). Therefore, these other factors need to be better acknowledged (Albert et al., 2014b).

A key part of the investigations here was looking at landscape scale work, because in conservation there has been a paradigm shift to working in this way and the NIA Programme was an attempt by the government for this to happen. This is difficult to investigate because that would involve stating whether the HHL NIA was successful, which is not the point of this work. However, for this project landscape scale work appears to have been carried out through conservation legacy, using previous work, investment and infrastructure to make decisions

about wetland restoration, either because they do not want to waste it or because they do not know else to do it. Conservation legacy was a key theme in the decision making process, both in terms of approaches to conservation (e.g. a preference for habitat and species focused conservation rather than the concept of ecosystem services) and the incorporation of specific projects, which is a big learning point here. There are many calls in literature and policy for more work at the landscape scale (e.g. Botequilha Leitão and Ahern, 2002; Lawton et al., 2010; McKenzie et al., 2013), but it appears that more guidance is needed on the ground for how this should be done.

One of the main problems the HHL Partnership had with the NIA Programme was the lack of information provided by Defra on what was expected and how to go about different actions on the ground. Also, they felt that three years was too short an amount of time and the money was not sufficient to properly deliver. Another issue was that they struggled to keep hold of a Programme Manager, due to a range of reasons. There were mixed opinions over whether working in a partnership made decisions better, some believing the increase in knowledge and buy-in was useful, whereas others suggesting it meant more compromises were made and decisions were more difficult and slower.

The HHL provides an insightful case study for understanding the role that partnerships have in decision making processes, because much of the literature believes they are a good thing (e.g. RSPB, 2001; Hume et al., 2008; Lems et al., 2011; Ruiz et al., 2011). In the HHL NIA decisions were made that affected the landscape at a range of different scales and without the partnership many of these decisions would still have occurred, but the ones on a larger scale would have been difficult. There would not have been the wide variety of expertise and experience to bring together in order to make the best decisions possible (RSPB, 2001; Hume et al., 2008; Lems et al., 2011). Another important question to ask is do partnerships make decisions better? They certainly bring more knowledge and expertise to a decision, but this is based more on the knowledge in the room, rather than going looking for evidence (which can be quite hard for them to find and interpret), and having more people involved increased buy-in. Pullin et al. (2004) believe that actually decisions are based on anecdotal and personal experiences of the people involved, which is backed up by what was found here.

They suggest this occurs because it is not in a format readily available and they do not have time to go looking for it in their jobs, not because they do not want to use scientific evidence (Pullin et al., 2004).

Many interviewees believed that the partnership led to compromises with decisions in order to reach a consensus, which was to the disadvantage of the project and the landscape. It was discussed at the beginning of the chapter that complexities and controversies occur due to organisational and personal differences (Jasonoff, 2003; Lems et al., 2011) and so it is not surprising that the interview responses showed evidence of that. However, Lems et al. (2011) do believe that by working with more partners, relationships can grow that allow more widely developed discussions, and therefore a more acceptable Business Plan, which was also displayed in the interview process.

Leading on, another important question is do partnerships add value to the process? From looking at the interviewee responses it is difficult to draw out an answer, although the Business Plan suggests that it does. Some interviewees do believe that value was added, with more knowledge involved, and also that it gives an increased chance of success because of enhanced experience and buy-in. However, several interviewees questioned whether the partnership would continue to be so successful at the end of the NIA Programme when the funding stopped. They were suggesting that there would be little incentive for partners to continue to be involved without funding, as it would be much harder to achieve anything. Therefore, partnerships may only work when there is funding (i.e. incentives) to get people involved and keep them there. Overall, there was a positive response to working in partnerships, with just a few issues, and many people believed that it did add something to the project, but the incentive to be involved is driven often funding availability.

It is interesting to consider whether the Partnership carry out adaptive management, using the criteria that was set out earlier in the thesis from the literature (Voss and Bornemann, 2011; Susskind et al., 2012; Rychlewski et al., 2014; Westling et al., 2014). The first part of the criteria was recognition that knowledge is incomplete and that learning should continuously occur and be fed into decisions. There was little evidence of this in the HHL NIA process, they

tended to use information they already had and did not go looking for knowledge outside the group. The second part was that stakeholders should participate from a range of backgrounds, including both social and scientific. There was a wide range of stakeholders involved, but few from the social science side. Finally, a group should have goals to measure themselves against. As part of the NIA Programme they did have a series of aims to meet, which by the end they counted themselves as successful against, but these aims did not necessarily measure their actual working. Therefore, the HHL NIA had some elements of adaptive management, but more work on the ground is needed to properly implement this useful concept (Westgate et al., 2013). The government document on NIAs “The Natural Choice Document” (HM Government, 2011) did include many aspects of adaptive management and it seems this might be the way that more policy goes in the future.

6.4.5 Summary

In terms of what factors were used to choose the locations for wetlands, in order to address the objective of this chapter, there was a range that fed into the decisions. Often biodiversity principles provided the theory behind these decisions, but the locations chosen for restoration were often those where work had already been undertaken, where the land was already owned by the Partnership, from knowledge within the Partnership or where organisations had a particular interest. Therefore, these decisions were driven by opportunistic or practical considerations, and the idea of conservation legacy appeared to be key. There was evidence of decisions around landscape scale approaches (which will be covered more in the following chapter), but there was also a lot of evidence of the use of well-established practices, such as species-focused activities. These probably actually provided the basis for the landscape scale work, and so both were important here, including the relationship between them. The role of the concept of ecosystem services was not as prominent as biodiversity, with partners struggling to practically apply the concept. However, there was actually evidence of decisions being made on services in the landscape, especially later on in the process, and so the role appeared to increase over the course of the project. Some of the evidence from the interviews and meetings would suggest

that there were just tokenistic efforts towards biodiversity and ecosystem services. This may be the case with the concept of ecosystem services, but not with biodiversity and actual services.

Overall, the decision making processes of the HHL NIA for wetland restoration involved a wide range of factors for each decision. Much of the literature states that these factors need to be included in addition to the scientific concepts in decision making, and that practical considerations are also important (Maltby, 2010; Faith, 2012). This did occur in the HHL NIA decision making process, but needs to become more accepted in general conservation decision making. In terms of the concepts of biodiversity and ecosystem services, biodiversity had a much more obvious role throughout, but with a big influence from practical considerations, and the use of the concept of ecosystem services appears to be on the rise, but more research and guidance is needed. The ideas of landscape scale work and the concept of ecosystem services are seen to be beneficial to wetland restoration, but more research and on the ground work is needed on their delivery, which could benefit from having adaptive management applied as well. These key themes will be investigated further in the next chapter.

Chapter Seven – Comparison between the identified locations for restoration and the HHL NIA decision making process

7.1 Introduction

The purpose of this chapter is to address the final objective, which is:

To compare the information gathered in the previous objectives to inform wetland restoration, and consider the potential and limitations for biodiversity, the concept of ecosystem services and landscape scale work in wetland restoration.

This objective uses the results from the first three to examine what causes the differences and similarities between the two different types of information and outcomes, the first of which was covered in Chapters Three to Five and the second in Chapter Six. Once these have been summarised and compared, there will be a discussion on the implications of this study for the role of biodiversity, the concept of ecosystem services and landscape scale work in conservation. This is important to do to inform better conservation decision making that has a clearer understanding of how to use biodiversity, the concept of ecosystem services and the other factors in landscape scale wetland restoration. Currently all the various factors are not always clearly considered, in some cases because it is not known how to, and so the benefit of this work is clear.

The two different types of information and outcomes that are referred to here are those covered in Chapters Three to Six. The first type comes from Chapters Three, Four and Five, using biodiversity, the concept of ecosystem services and physical characteristics to identify a range of restoration options for the HHL landscape. Selected metrics of all three were mapped and then used in different combinations under a grid system to explore locations for restoration, taking into account what the current wetland patches in the landscape already deliver. The second type, covered in Chapter Six, used interviews, meeting observations and document analysis to understand the process that the HHL Partnership went through when making decisions for the NIA Programme, especially looking at the role of biodiversity, the concept of ecosystem services and the other factors.

7.2 Summary of the findings

7.2.1. Biodiversity, ecosystem services and physical characteristics

This started with the selection of metrics for biodiversity and ecosystem services, which were mapped and modelled in Chapter Three. In Chapter Four these metrics, alongside the physical characteristics mapped in Chapter Two, were investigated further by identifying which were potentially delivered in the current wetland patches of the HHL. This information, along with key questions identified from the literature, were used to produce a series of restoration options in Chapter Five. This started with two options looking at single biodiversity metrics, which showed limited occurrence of these taxa within the landscape. Most records were located around the core areas, suggesting that restoration should focus on expanding these areas to conserve these taxa. The next two options focused on restoration by targeting single ecosystem services; carbon storage and flood mitigation. Carbon storage focused on the enhancement and expansion of the two core areas of the NNRs, whereas flood mitigation had both some expansion of core areas and some development of connectivity.

Next restoration options were produced based on what is optimal for biodiversity and optimal for ecosystem services. The biodiversity option showed some expansion of core areas and development of connectivity, which was fairly similar to the single biodiversity metric options, but with more restoration suggested in this case to increase biodiversity. The ecosystem services option was fairly complex, with a random scattering of results across the landscape. There was evidence for expansion of the core areas for delivering ecosystem services, especially around the two NNRs, and some development of connectivity. In comparing this to the single services option they all showed the expansion of core areas, but otherwise were fairly different.

Next the biodiversity and ecosystem services options were compared to see if it would be possible to deliver a multi-functional landscape. The comparison suggested this would be very difficult to do, and would result in trade-offs. This information contributes to the low level of literature on this issue, which currently does not fully understand the spatial relationship between biodiversity and ecosystem services (Anderson et al., 2009). Exactly what trade-offs would be

context dependent and maybe informed by a range of human related factors. The biodiversity option was also compared to the two single ecosystem services options, and it was found that there is little association between them, as both single ecosystem services options focused much more on the core areas than the biodiversity option did.

Finally, a wetland potential option was produced using the physical characteristics of wetlands and the landscape. This showed the expansion of core areas again, and some connectivity developed along river corridors. Overall, this work significantly increased understanding of biodiversity and ecosystem services in the HHL, and beyond, as well as providing a useful method that could be potentially used by practitioners, as it is relatively simple, and access to the data and models is not too difficult. Some issues of real-world factors also appeared here, such as data availability.

7.2.2 HHL NIA real-world decision making process

In Chapter Six the factors involved in the decision making process of the HHL NIA were investigated. It was found that biodiversity and ecosystem services did have some role in the process, but that most decisions were an interaction of many factors, as nothing occurs in isolation. It appeared that biodiversity was very important to many parts of the process, with decisions often starting off with ideas of biodiversity, such as ecological networks, but then tempered by other more practical factors. One factor that often appeared alongside biodiversity was conservation legacy, which was partly defined by biodiversity principles originally. Previously, much of biodiversity decision making has been around more traditional approaches, with a focus on particular species and habitats (Carpenter et al., 2006; Fisher and Brown, 2014). Although there was evidence for this with the HHL NIA, there was also evidence of landscape scale approaches, with a move away from static conservation and protected areas towards facilitating ecosystem processes, such as connectivity and dispersal. Overall, the approach towards conserving biodiversity did appear to change over time with the introduction of the landscape scale, but there was also a lot of evidence of the use of the traditional well-established practices, such as site specific habitat work.

The role of the concept of ecosystem services in decision making was less clear than for biodiversity, partly because there are still so many unknowns about the concept and its delivery (Sitas et al., 2013; Wu, 2013; Pan et al., 2013; Albert et al., 2014a; Pagella and Sinclair, 2014). The HHL NIA process used the concept in three different ways: (1) some discussion of the concept but then no use of it, (2) some discussion and then use of it, and (3) using the specific services without recognising it under the concept. It appeared that the concept was often included as an afterthought in decision making, because they felt they were obliged to, but were not provided with the right guidance by Defra for its delivery on the ground. It also appeared there were certain services they could more easily associate with the concept, such as carbon storage, but others which they did not, for example food production. Many of the interviewees thought they were struggling to deal with the concept of ecosystem services, but actually there were examples where they were using it implicitly, which they need to learn from and use in the future. Awareness and integration of the concept in decision making changed over the course of the study and the NIA project. It was not mentioned with the boundary case study at the beginning, then there was some evidence of superficial use, but by the end some new ideas had been built into the HHL NIA using the concept.

There were many other factors that were involved in the decision making process, several of which are also discussed in the literature (e.g. Maltby, 2010; Cotter et al., 2014). In most decisions there was no single factor involved, it was a series of them interacting, often with biodiversity, and sometimes with ecosystem services. Chapter Six identified three factors that exerted particular influence in decision making: (1) conservation legacy, (2) agriculture and (3) resources (including funding). Conservation legacy was consistently identified as a driver for decision making, which may have been partly due to the requirement for NIAs to link existing projects, and its effect on decisions will be discussed in the following paragraph. Even though agriculture was only really mentioned in the boundary case study, it is the dominant land use within the landscape. Therefore, it is important to decision making in terms of the cost and availability of land for restoration, therefore reducing the areas they could work on and the amount of restoration they wanted to achieve. Resources appears to be the ultimate practicality factor, because without money, time and people nothing would occur,

and the restrictions to all three meant this really tempered what they would ideally have liked to achieve in the HHL. The way that resources influenced decision making meant less restoration overall, but in terms of the configuration of where restoration occurred, this was often linked to the idea of conservation legacy, using areas where the infrastructure is already in place, such as Potteric Carr, so that more can be achieved.

An important part of this thesis was to look at the role of landscape scale work, because it has been identified as an approach for achieving better conservation and the HHL NIA provides an example of the practical application of this. For the HHL NIA, the way of doing landscape scale work appears for a large part, to be through conservation legacy, i.e. focusing decisions on wetland restoration around previous work, investment and infrastructure. This was partly due to the NIA requirements to link existing projects, but the Partnership was also motivated by not wanting to waste their work, by not knowing how else to achieve it, by a perception of those areas having the highest biodiversity and the need to include projects run by partner organisations to secure partnership buy-in. Attempting to do landscape scale work through conservation legacy meant that a collection of existing projects were included, some of which were linked up. The idea of ecological networks was also brought up in the Business Plan as a way of doing landscape scale work.

Overall, there were a range of factors that fed into decisions, with biodiversity often providing the theory and ideas, but the other factors making the decisions more practical and realistic in a multi-functional landscape. The concept of ecosystem services did have a role, but not as prominent as biodiversity, and so more work is needed on its practical application. Some evidence from the interviews and meetings suggested that there were just tokenistic efforts towards using biodiversity and ecosystem services, which maybe was the case with the concept of ecosystem services, but not so with biodiversity and actual services (that are often not recognised under the concept). The analysis from Chapters Three, Four and Five demonstrated that the landscape is data rich, but it is clear that this was not used much by the HHL NIA. There was a lot of evidence of well-established practices (such as site specific work) being used to inform decision making in the landscape and little evidence to suggest that the partnership sought

new knowledge or approaches. Much of the literature states that a wider range of factors need to be included in decision making in addition to scientific concepts, and that the practical considerations are also important (Maltby, 2010; Faith, 2012). This certainly occurred in the HHL NIA decision making process, but it needs to become more accepted and occur alongside better integration of scientific knowledge.

7.3 Comparison of the outcomes from the two types of information

In this section the different restoration options developed in Chapter Five will be compared to the anticipated outcomes from the HHL NIA decision making process. These anticipated outcomes were not necessarily what occurred, but what the Partnership wanted to achieve with the decisions they made. Here the HHL NIA anticipated outcomes, which can be seen in Figure 7.1, will be laid over the individual restoration options for biodiversity, ecosystem services and wetland potential, to do a visual assessment of the similarities and differences. Ideally the HHL NIA anticipated outcomes would be laid over all three options together, but visually this would be too difficult to read and it was already concluded in Chapter Five that a multi-functional landscape cannot be easily achieved. The HHL NIA anticipated outcomes have not been converted into the grid system, because of the format of the data and it would lose some of the information. Therefore, each pairing cannot be numerically compared. Also, the HHL NIA anticipated outcomes only cover the NIA area (which was spatially constrained to fulfil the criteria for funding), whilst the restoration options cover the whole NCA, which would influence any numerical comparison. Each comparison will be considered in terms of both extent and location (which will both be obviously effected by the differences in landscape size), and the HHL NIA anticipated outcomes will be looked at as a whole, rather than the individual components, such as “create”.

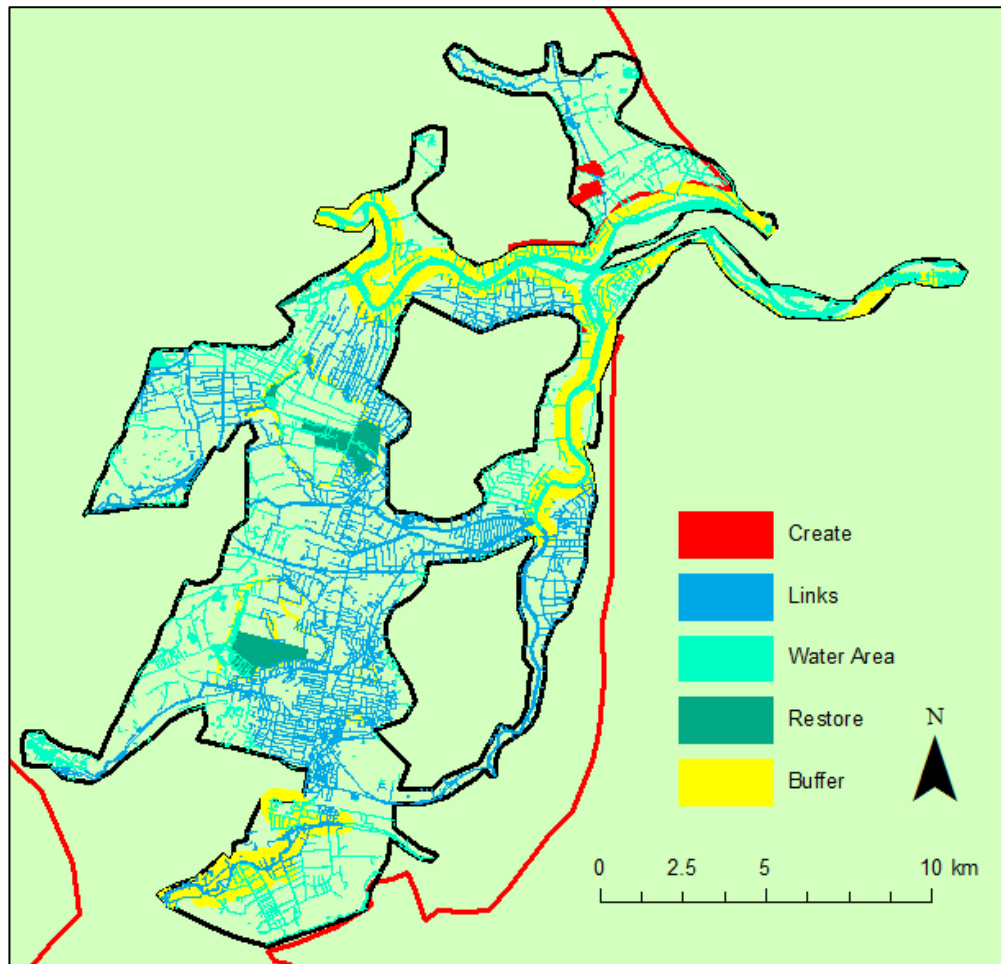


Figure 7.1: HHL NIA ecological restoration priority zones – the anticipated outcomes (obtained from YWT).

Figure 7.2 shows the biodiversity restoration option with the HHL NIA anticipated outcomes laid over. In terms of extent, the biodiversity option covers fewer areas than the anticipated HHL NIA, including within the NIA area. For location, the biodiversity option has some similar areas highlighted within the NIA, such as around the two NNRs, but the HHL NIA map covers much bigger patches, and some of the river corridors. One of the hotspots for the biodiversity option is Potteric Carr Nature Reserve, which only has some inclusion in the HHL NIA. Overall, the biodiversity option appears to show connectivity of current wetland patches, which does work on some of the same principles as the anticipated HHL NIA, but there are big differences between the two outcomes.

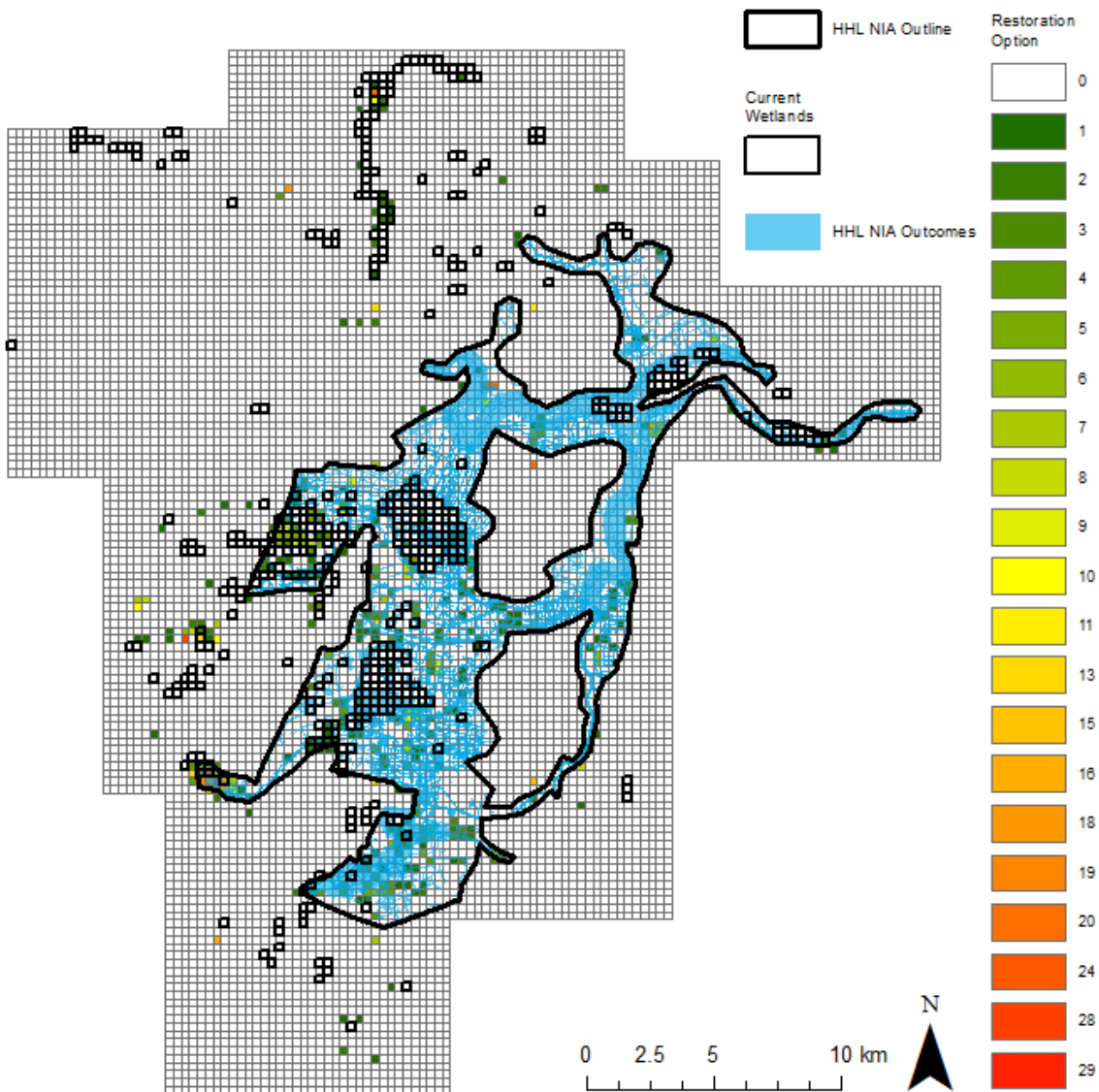


Figure 7.2: HHL NIA anticipated outcomes with the biodiversity restoration option.

Figure 7.3 shows the ecosystem services option with the HHL NIA anticipated outcomes laid over. The restoration option gives much more coverage than the HHL NIA map, even when just considering the area within the NIA. In terms of location, the ecosystem services option covers nearly all of the areas that the HHL NIA map does, however most areas only deliver one to three services, with little coincidence between different ecosystem services. Looking back at the original ecosystem services maps in Chapter Three it can be seen, for example, that areas that deliver flood mitigation and carbon storage appear not to deliver recreation. The hotspots for this option are mainly within and around the two NNRs, which is only partly covered by the HHL NIA map. This option appears to be working around the idea of increasing the core areas, which does not get mentioned with the HHL NIA map (although will occur through the “restore” and

“create” elements), and visually there are many differences between the two outcomes here. In Chapter Five, two individual services were also used to suggest restoration locations, and although these both also focused on the core areas, the patterns elsewhere were very different to each other, the general ecosystem services option and to that of the HHL NIA.

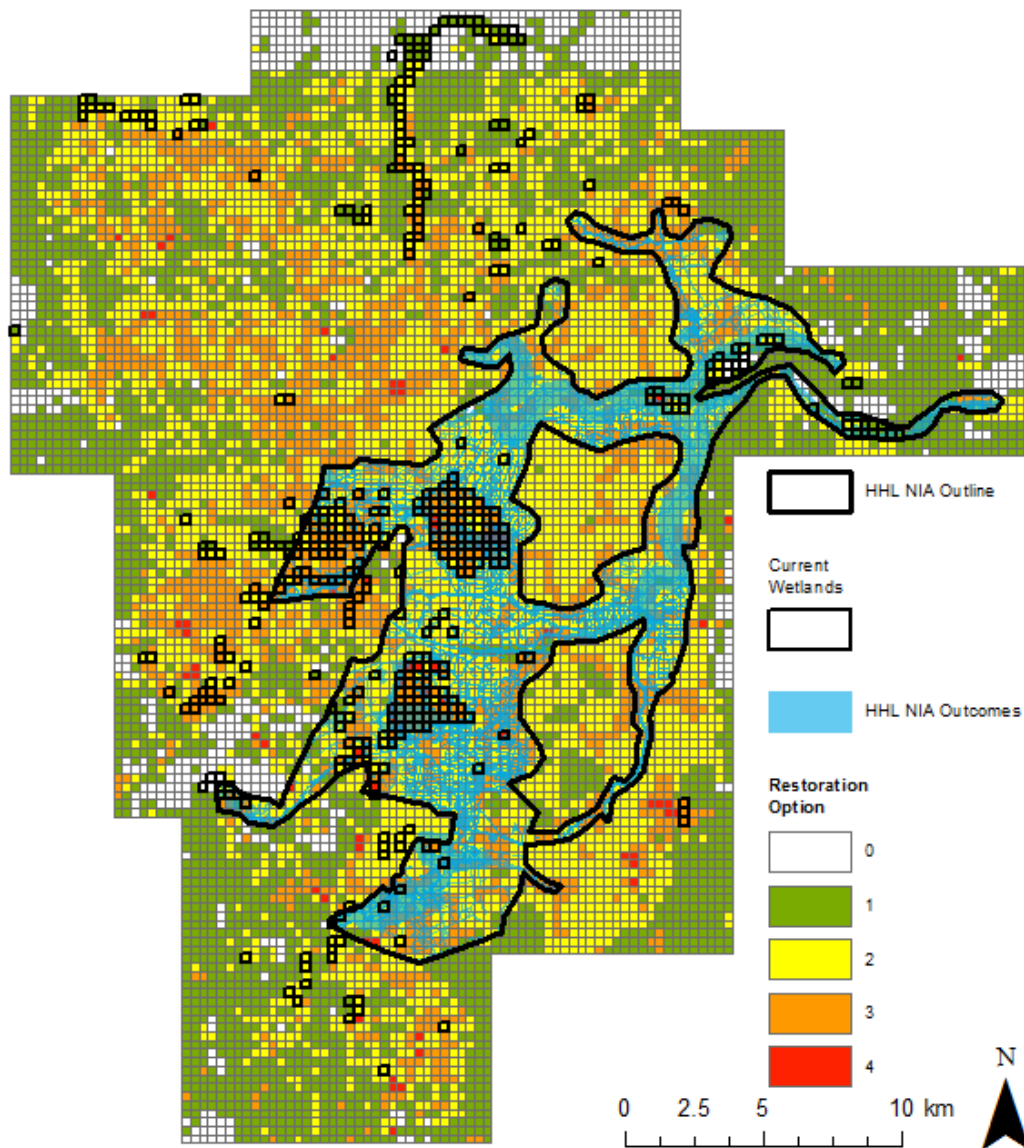


Figure 7.3: HHL NIA anticipated outcomes with the ecosystem services restoration option.

Figure 7.4 shows the wetland potential restoration option with the HHL NIA anticipated outcomes laid over. The wetland potential option shows less coverage than the HHL NIA map, even if only considering the NIA area. In terms of location, the wetland potential option, from a very basic visual comparison, appears to cover mostly different areas from the HHL NIA map. The wetland potential map appears to show some core areas and the expansion of them, which is some of

the same ideas that the anticipated HHL NIA work on, but they still show very different outcomes. These results suggest that the work of the NIA is not necessarily in the areas most physically conducive to restoration.

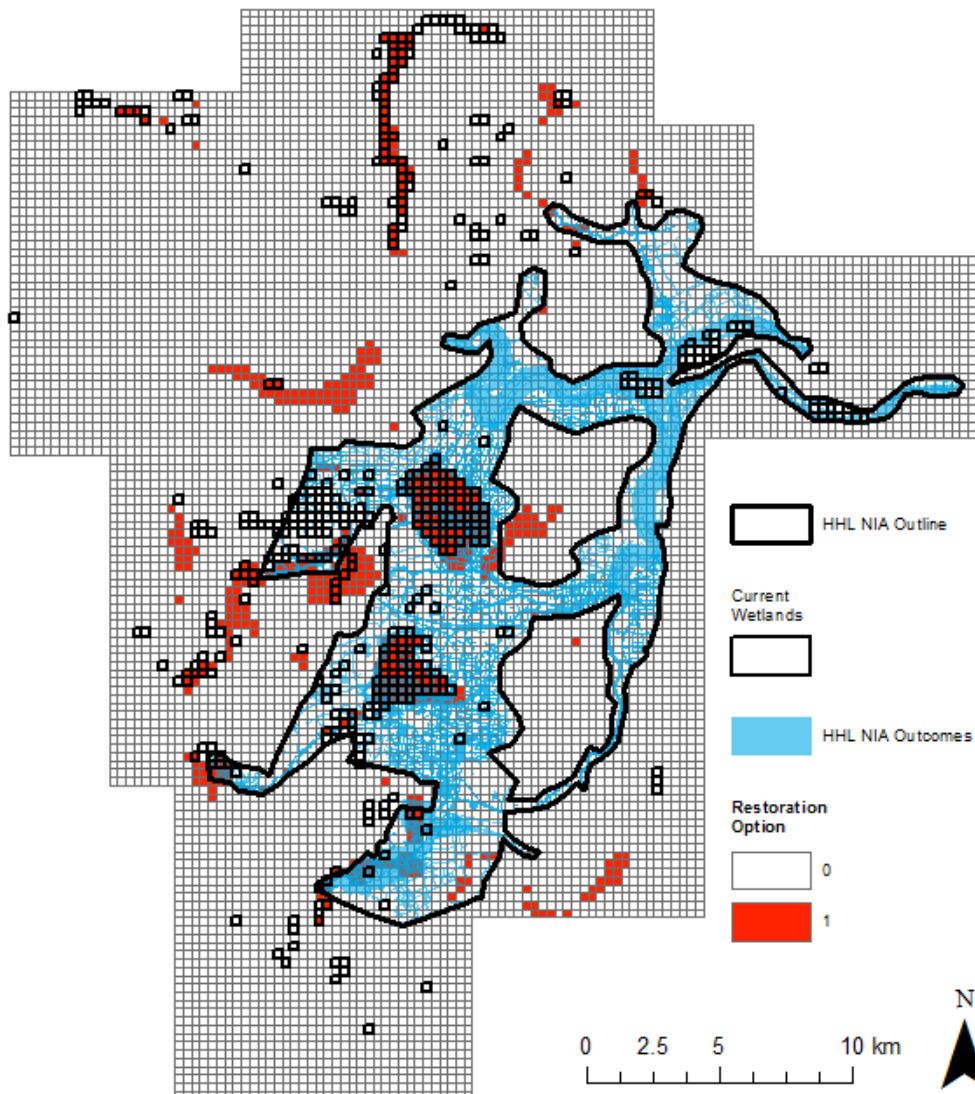


Figure 7.4: HHL NIA anticipated outcomes with the wetland potential restoration option.

As it has been concluded previously that these three options do not fit together and so a multi-functional landscape would be difficult to achieve; in order to base restoration decisions on these, trade-offs would need to occur. However, as the different options show different elements of ecological networks, by having a combination of all three it could lead to better landscape scale conservation, as with the HHL NIA. It is also interesting to note that many of the areas for the wetland potential and ecosystem services options sit outside the NIA boundary, which demonstrates how real-world decision making has influenced restoration at the landscape scale. The boundary was clearly defined by real-world priorities

and the restoration options shows that this may not have been the optimal boundary. The comparisons were difficult because of the different formats of the two approaches and the lack of numerical comparison. However, it is obvious that the two approaches have different outcomes. The reasoning behind this will be investigated in the following sections, along with the implications for future decision making on landscape scale restoration.

7.4 Similarities between the two types of information

There are very few similarities between the two, in terms of factors involved and the outcomes, which is not surprising considering the way they were carried out and the results in the previous section. One similarity is that biodiversity and ecosystem services did have some role in both, but this varied between the two, which will be discussed further in the following sections. Another similarity is that both outcomes focus restoration around the two NNRs, obviously a key part of the HHL landscape. Also, both apply principals of ecological networks, although the different options display different elements. The biodiversity option shows that many smaller patches in the landscape could be connected (stepping stones), the ecosystem services option requires working on core areas and the wetland potential option necessitates buffering existing wetlands and creating new areas. The HHL landscape under the NIA plan has more detail of specific restoration actions, identifying areas to create, buffer, link and restore. These actions all facilitate the creation of ecological networks, although there appears to be little coincidence of the networks between the two. Therefore, with both, and not always deliberately, the idea of ecological networks is important, which has already been identified as key to landscape scale work. This will be discussed further in Section 7.7.

7.5 Differences between the two types of information

The differences are obvious, both in terms of the outcomes and the information. The differences between the outcomes are covered in Section 7.3, but can be summarised by stating that there were big differences in both location and extent between them. Some of these obvious differences will be caused by the fact that

the restoration options are over a larger continuous landscape, whereas the HHL NIA is an oddly shaped, smaller landscape, constrained in size and design by real-world factors. However, it is the differences in information where the most can be learnt for more successful future landscape scale wetland restoration.

The major difference, and the point of understanding the decision making process of the HHL NIA, is the factors that are involved in the process of wetland restoration decision making. With the first type, the restoration options were all based on biodiversity, ecosystem services and/or physical characteristics of the HHL landscape. The second had many other factors involved alongside these. The main three were identified in Chapter Six as conservation legacy, agriculture and resources (including funding). There are a number of causes for why there are differences (outside that of purely the way the outcomes were produced), including a lack of research, a lack of access to research, a lack of people looking for evidence and that actually in the real-world we need to consider these factors, which will be discussed further, alongside the implications, in Sections 7.6 and 7.7. It would appear that the differences were more caused by the other factors, rather than a lack of knowledge, but both were important.

7.6 Implications for biodiversity, the concept of ecosystem services and real-world factors

All three of these aspects are important for answering the objective and the overall aim of the thesis, and so the important learning points for all three will be discussed, as well as touching on partnership working. Biodiversity obviously had a significant role in the first outcomes, and it was involved in the second, but mostly alongside other factors. Although the inclusion of these other factors has already been identified as important, and will be discussed further later in this section, there is also a need to prioritise biodiversity, to keep it at the forefront of conservation and to ensure evidence based decision making. It was seen through the interviewing process that there was passion for biodiversity, but the use of it was often opportunistic and based on existing knowledge of the partnership members. They did not appear to look for knowledge outside the expertise of the partnership membership. It was also identified that they used many well-

established practices for biodiversity, such as site specific work, and again did not really look for new approaches, other than some thinking towards ecological networks and landscape scale work (see Section 7.7).

Decision making for biodiversity needs to be stop being centred around opportunity and current knowledge and requires more forward planning. Conservation partnerships, such as the HHL, must first define clear objectives, before looking for the knowledge to support it and goals to measure it by. Therefore, there needs to be more considered integration of biodiversity in decisions. Previous chapters have demonstrated that there is the data available for such as approach, but that it is really not used, which is a key lesson. The literature states that it is difficult to deal with biodiversity when there is poor data (Smith et al., 2008) and that historically where decision makers do use data it tends to be very basic (Knight et al., 2011b). Therefore, this work can be used as an example of where the much needed evidence based decisions can be made. However, even if the information was provided there is no guarantee that decisions would improve and the range of other factors would have been involved regardless, which will be covered in Section 7.8.

A problem identified from the literature is that biodiversity is often protected for humans, rather than for its own intrinsic value (Wu, 2006; Zhang et al., 2010; Reyers et al., 2012b). However, there was little evidence of that in this case. Actually the partners were fairly wary about using the more anthropocentric concept of ecosystem services, partly because they saw it as a move away from biodiversity, which is what they were used to doing. There have been problems with conserving biodiversity, and the concept of ecosystem services has been put forward as a way to overcome this (Farber et al., 2002; Mace et al., 2012). However, there was evidence here that practitioners are not keen on the concept and so either it needs to be branded differently, as something very separate to biodiversity, or more education is needed on how the concept could be useful for biodiversity conservation.

The concept of ecosystem services had a clear role with the first outcomes, but was less utilised in the second compared to biodiversity. There are still many unknowns about the delivery of ecosystem services, as identified from the

literature (e.g. Anderson et al., 2009; Pan et al., 2013), but the work done in this thesis, spatially mapping a range of ecosystem services, contributes evidence to answering these questions. One question asks if it is possible to deliver multiple services at the same time (Pan et al., 2013), but the ecosystem services restoration option showed that most parts of the landscape only delivered one to three of the five evaluated. This restoration option was also very different in configuration to the biodiversity option, which will make it even more difficult to convince those already working on biodiversity to also work on this concept. This is a big limitation to overcome. However, some of the ecosystem services used are a product of human intervention, such as woodland in active management, and so they are not separate from socio-ecological issues.

There were many reasons identified for the lack of understanding of the concept of ecosystem services. These were also found in the literature, and include: the complexity of the process, lack of data, lack of research on practical applications or tools, problems with how the environmental and social issues are connected and that there is barely any precedence for its management (Carpenter et al., 2006; Casado-Arzuaga et al., 2013; Pullin et al., 2013; Hatton MacDonald et al., 2014; Fletcher et al., 2014; Pagella and Sinclair, 2014; Dicks et al., 2014b; Walsh et al., 2015; Rucklehaus et al., 2015). There was some buy-in to the concept, as partners had been using it in some form for many years, but they felt that more needed to be known before it could be successfully delivered. Therefore, this is another major learning point that there was lack of understanding of the concept, which led to it not being used appropriately. The Partnership felt this was because there was not good enough dissemination of research down from academics, those on the ground did not have time to go looking for evidence and that not enough information was given by Defra on how to deal with the concept as part of the NIA. This lack of guidance was not just an issue with this concept, but the NIA programme in general, which needs to be addressed for future work.

The issue of lack of data was addressed in previous chapters of this work, with some ecosystem services modelled and mapped across the HHL. If this information was used then more evidence based decisions could be made, which may also overcome some of the problems with understanding the concept. The HHL NIA showed evidence of increasing use of the concept over time, but the

issues discussed need to be overcome and the concept needs to become more embedded in conservation decision making to be more effective. The improved use of the concept could have large benefits. Firstly, it could be of a big advantage to landscape scale management (Sitas et al., 2013; Albert et al., 2014a; Pagella and Sinclair, 2014), which will be discussed in Section 7.7. It also could be used as a framework to overcome tensions and issues within a landscape, such as those that exist between conservation and agriculture in the HHL (Rey Benayas and Bullock, 2012). There are numerous other benefits, such as: a communication tool to stakeholders as way of showing different conservation options, as a way to look at the synergies and trade-offs between economic, social and environmental interests, to enhance the acceptance of planning proposals, to act as a starting point for discussions amongst stakeholders and currently as a good way of securing funding (Goldman et al., 2008; Albert et al., 2014a, b; Hatton MacDonald et al., 2014).

Overall, the best ways to overcome the current problems with the application of the concept of ecosystem services appears to be to have more of a connection between research and work on the ground, more robust methods for delivery on the ground (Crossman et al., 2013), more guidance from organisations such as Defra and a more holistic view that highlights all the benefits. This work on biodiversity and the concept of ecosystem services links to a bigger question of whether science in general has a role in conservation decision making and what other factors are also involved in the real-world. For the current role of science there is plenty of literature that suggests that for a range of reasons the role is not good enough (e.g. Pullin et al., 2013; Dicks et al., 2014b; Walsh et al., 2015), as mirrored by this work. However, we need to have more “evidence-informed conservation” (Dicks et al., 2014b) and the science makes the work less ad-hoc, and more planned and structured, which is needed.

There were many other factors involved in the HHL NIA and more needs to be learnt from them in order to fully understand and use the real-world process. Many of these other factors, such as economic, social, political and institutional issues, were also found in the literature (Knight et al., 2011a). However, they are often poorly acknowledged and need to become accepted as part of the process, to attempt to have more successful decisions and reduce conflict (Nhancale and

Smith, 2011; Pullin et al., 2013; Albert et al., 2014b; Cook et al., 2014; Dicks et al., 2014b; Metcalfe et al., 2015). These factors include subjective values (Pullin et al., 2013; Walsh et al., 2015), as was found with the HHL NIA, which are important, but they cannot be the only source of information, evidence is needed as well. Local knowledge often understands more about what is occurring now on the ground, whereas research provides something bigger and helps predict future change. Both are therefore important, but the process needs to become more open and transparent (Jasonoff, 2003). Overall, Sutherland et al. (2012) has identified that we need to learn from the experiences of those working on the ground, and also provide more evidence of successful decisions and methods (Bottrill and Pressey, 2012).

Partnership working was one of the main points of the NIA Programme, and though it was not one of the main points of interest in this work, there are many issues that can be learnt that link to the other aspects of this section. In general, the partners were positive about partnership working and considered it a useful approach for delivering landscape scale conservation, a viewpoint shared in the literature (e.g. RSPB, 2001; Hume et al., 2008; Ruiz et al., 2011; Lems et al., 2011). Many of the partners believed that the increase in knowledge and buy-in was useful, leading to more widely developed and accepted discussions, which supports other studies on the subject (RSPB, 2001; Hume et al., 2008; Lems et al., 2011). Also, without the partnership, many of the decisions on a larger scale would have been very difficult, which demonstrates the importance of partnerships for work at a larger scale. However, it also presented a number of challenges. Other partners suggested that it meant more compromises and that decisions were more difficult and slower, and it seemed to make them less likely to look for knowledge elsewhere. Therefore, there are both positives and negatives to be learnt about partnerships, but further research is required.

Overall, learning points have been gained from considering biodiversity, the concept of ecosystem services, the other real-world factors and partnership working. From biodiversity more data needs to be used in decisions to make them more evidence-based. This is also the case with the concept of ecosystem services, but much more work is also needed about how to practically apply this concept on the ground and its integration into existing conservation practice,

which is currently focussed towards biodiversity. The other factors need to be accepted more as part of the decision making process; that they explicitly needed to make decisions more realistic. For partnership working there are both positives and negatives, but more research is required. Decisions need to have clear objectives that cover all of these aspects (Piper et al., 2006; Arts and Buzier, 2009; Jasonoff, 2010), and include clear and rigorous science with increased knowledge (Maltby, 2010; Retford et al., 2012), which this work has shown is possible.

7.7 Implications for landscape scale restoration

Another key part of this work was to investigate what role landscape scale work had and what can be learnt. As has been shown, it is not feasible to restore continuous large areas of wetlands, and so we need to enhance what already exists, and integrate wetland patches into the larger landscape, ensuring connectivity between patches. There are many calls in literature and policy for work at the landscape scale (e.g. Botequilha Leitão and Ahern, 2002; Lawton et al., 2010; English Biodiversity Group, 2011; McKenzie et al., 2013), and there has been a rise in its use (Jones et al., 2013). There has been a paradigm shift from small protected areas and species work to the landscape scale. However, it appears that much more research (as has been done here) and guidance is needed for delivery on the ground. It was identified in the previous section that the HHL NIA had a problem with the guidance provided from Defra for the programme, which includes how to deliver landscape scale restoration, both practically and applying the concepts proposed by Lawton et al. (2010). Therefore, more information needs to be produced to achieve multiple benefits to wildlife and people at the landscape scale (Lawton et al., 2010; English Biodiversity Group, 2011; UK National Ecosystem Assessment, 2011).

A major consideration for landscape scale that came out of this work was the idea of conservation legacy. The HHL NIA used areas of previous work, investment and infrastructure to decide on locations for restoration. The reasoning behind this includes that it was requirement of the NIA Programme, but it also appears that they did not want to waste what they had already done, they were not sure

how else to do landscape scale work, these were perceived as the greatest biodiversity assets and any work done was more likely to be effective. Conservation legacy appears to be a key theme in much of the decision making process both in terms of approaches to conservation (e.g. a preference for habitat and species focused conservation rather than the concept of ecosystem services) and to the incorporation of specific projects.

The other aspect of landscape scale that has come out is the use of ecological networks and the Lawton principles (Lawton et al., 2010). Both the HHL NIA anticipated outcomes and the restoration options show various elements of ecological networks. The restoration options all show different elements, and so by using a combination of all three, a multi-functional landscape may well be delivered, as found also in the literature (e.g. Linder Mayer and Nix, 1993). Also, the biodiversity option showed areas of interest outside the protected areas, and so to connect up this up they would need to use Lawton principles, again showing how important they are for a landscape.

From looking at the maps at the beginning of the chapter it can be seen that many of the hotspots for the three restoration options, especially for ecosystem services and wetland potential, are actually outside the HHL NIA boundary. In the previous chapter the boundary decision making process was used as an example of real-world decision making, and the fact that the options have strong restoration suggestions outside the HHL NIA boundary demonstrates this. So for example, perhaps not much work has occurred in the east of the landscape, even though there were suggestions from the ecosystem services and wetland potential options, because of the real-world factors.

Overall, the benefit and need for working at the landscape scale is obvious, but there is still more research needed into how to deliver it on the ground. In this project the main two ways of achieving landscape scale restoration appears to be through using conservation legacy and ecological networks. These should be used to inform future work on the ground, alongside the other issues discussed, to achieve the delivery of multi-functional landscapes.

7.8 How can these issues be overcome?

It is also important to think about how some of these issues and implications can be overcome, which have been touched on in the previous sections. In terms of biodiversity, this work has found that more information is needed to give evidence of how to deliver landscape scale biodiversity, with a more considered integration of biodiversity in the process. There is also a need to think more about the specific goals and how these would be measured. This all needs to happen alongside the concept of ecosystem services, which is a useful concept, but not to take over from biodiversity. For ecosystem services there was also a call for guidance on its delivery, even more than for biodiversity. There is also a need for a more holistic view of the concept, so it can be used to overcome more issues, such as the conflict between conservation and agriculture. This will be much more difficult to overcome, but perhaps more education over what the concept can do and steering clear of economic valuation (which only benefits certain services) will help.

A number of other factors were identified as integral to real-world decision making for wetland restoration. These need to be more clearly involved to make decisions more realistic and more likely to be successful, and take into account the important contextual information of stakeholders. It was shown in this work, for example, with the boundary case study, that they had a big role in the process, especially conservation legacy, but the partners seemed not to really acknowledge their involvement, even though it was obviously there. Therefore, as with biodiversity and ecosystem services, more guidance is needed as how to involve them more clearly in the decision making process. However, these factors should not take away from biodiversity and ecosystem services; it needs to be alongside them. This is linked to how to deliver work at the landscape scale, which this work discussed through conservation legacy and ecological networks, but again more guidance is needed.

Most of the recommendations for how to overcome these issues require more evidence and guidance for “on the ground” delivery. However, the extent to which such information would actually get used is questionable, because this work has shown that the Partnership did not use much of the data available. If the HHL Partnership had used the information from Chapter Three the HHL NIA may have

looked different, but it is difficult to really answer this question. However, it would probably have not influenced the involvement of the other factors in the process, they are very important in the real-world context, as has already been discussed, and would still have rightly influenced decisions.

The next point is would they have even used the information if they had it. We would like to think they would do so, because just providing more evidence and guidance would be a relatively simple solution to delivering better landscape scale restoration. However, again it is not possible to really answer this question. Information on ecosystem services would probably be used as there were many calls in Chapter Six for this. However, more information on biodiversity may not, as they did not really think that they had much of a problem with it. Therefore, providing more guidance could improve decisions, but would not necessarily do so. It is hard not to make the assumption that more guidance will automatically make decisions better, as covered in a paper by Fernandez (2016), but this work is setting a precedent by not doing so.

This is a very difficult issue to overcome, but just acknowledging it in this work is a step forward. The best way to overcome it is to think about how to provide guidance, which is probably through workshops, where the information is provided through short points and maps to enable stakeholders to discuss it together and ask questions. Thus, sharing easy to digest information, that is about exchanging knowledge rather than just providing it. This would then hopefully better enable decisions. Overall, this chapter has found some interesting points in comparing the two different types of information and the outcomes, that could help inform better landscape scale wetland restoration in the future, which will be summarised in the following chapter.

Chapter Eight – Conclusions

The central theme of this thesis was to investigate the roles that biodiversity, the concept of ecosystem services and other factors had in wetland restoration at a landscape scale in the HHL through the NIA Programme. The sections that follow summarise the main conclusions of each objective, before addressing the aim, looking at the benefits beyond the HHL case study and future work.

8.1 Objective one

To map the main types of ecological knowledge that could potentially inform wetland restoration, as well as the spatial structure of current wetland patches in the HHL area. This ecological knowledge will take the following forms:

(a) Major biodiversity patterns

(b) Indicators of ecosystem service provision.

This objective was discussed in Chapter Three, where the main aim was to select a range of metrics of biodiversity and ecosystem services, and map these across the HHL landscape in preparation for the second objective, to provide information for wetland restoration. There is a great deal of literature and policy documents that consider which metrics to use. Selections were made based on these recommendations, the context of the landscape, the end point of the work, data availability and using the models available. Data availability, as well the ecosystem services models, is what defined the actual mapping process, as has been the case with other studies (Kandziora et al, 2013), and is a point to be aware of for any future studies. The two models chosen were InVEST and LUCI, because of the context, the services available in each model and to be able to compare the difference in results between different models.

The publicly available data and models used in this chapter are useful for spatially understanding biodiversity and ecosystem services, and certainly add far more information than was known before. Also, they provide a method that could better inform landscape scale wetland restoration. In order to get more of this type of information into decision making, methods are needed that are easy to use,

transparent and robust (Albert et al., 2014b). This chapter provides a way to do so, as maps can act as powerful communication tools and provide information about trade-offs and synergies for decisions (Jiang et al., 2013; Buckhard et al., 2013; Vorstitis and Spray, 2015).

8.2 Objective two

To use the information from Objective One to address:

(a) What biodiversity and ecosystem services are delivered by the current wetland patches in the HHL landscape?

(b) Identify a range of potential locations for wetland restoration for the HHL.

This objective was considered in Chapter Four for (a) and Chapter Five for (b), with the conclusions for (b) also discussed in Chapter Seven. For (a) metrics of biodiversity, ecosystem services and physical characteristics (mapped in Objective One) were compared to the current wetland patches to improve understanding of wetlands, to then be used to inform wetland restoration. A grid system was used at two scales to see what metrics had strong associations, which would suggest they are delivered by current wetland patches in the HHL. At both scales there were very similar results for the physical characteristics that were delivered by current wetland patches. These results can be used to build up a physical description of the current wetland patches, a useful starting point for understanding biodiversity and ecosystem service delivery. Distinct from the rest of the landscape these areas are low lying, have lower levels of certain pollutants, whilst the main land use and soil types are bog related.

In terms of biodiversity, there was again similar results at both scales that can be used to build up a picture of biodiversity in current wetlands relative to non-wetland patches. These areas tend to be protected by various designations, are within the wetland type BAP habitats and support higher levels of some of the groups and focal species. Wetlands are extremely important for beta and gamma diversity (UK National Ecosystem Assessment, 2011; Maltby and Acreman, 2011), and so by finding out what current wetland patches are delivering it is useful for learning how to protect certain aspects of biodiversity to halt the decline,

especially at this relatively unknown landscape scale (Feld et al., 2009; Cui et al., 2012). The delivery of ecosystem services also had many similarities between the two scales. These results show that the current wetlands do deliver some ecosystem services, such as habitat quality, flood mitigation, the provisioning service of managed woodland and carbon storage. By understanding what ecosystem services are delivered by current wetlands it adds evidence to some of the unanswered questions around ecosystem services such as “where are ecosystems producing benefits” (Bagstad et al., 2013b), what are the most important ecosystem services for a particular landscape (Peh et al., 2013) and can multiple services be delivered (Pan et al., 2013).

Two different scales (500 m and 1000 m grid systems) were used for this analysis to see what effect this had. In general, the results were fairly similar, but there were a couple of occasions where they were different, which could influence how a landscape is interpreted, as was also found by Holt et al. (2015). It is not possible to say here what the optimal scale for this method and the landscape is, as that would require much more detailed testing, but it is important to note for any similar projects that the scale chosen will make some difference to the results and therefore the interpretation (Holt et al., 2015). Care was taken in choosing appropriate scales, in accordance with the literature and original data format (Van Horssen et al., 1999; Engler et al., 2004).

The information from Objective One and Objective Two(a), along with some key questions identified from the literature, were used to create a series of wetland restoration options for the HHL landscape, based on the idealistic use of biodiversity, ecosystem services and physical characteristics. The most important question to answer was whether a multi-functional landscape could be delivered and in this case it was found that this would be extremely difficult. In the literature (e.g. Anderson et al., 2009), there is no clear answer to this question, and so this case adds evidence towards answering it. Trade-offs would be needed to deliver both biodiversity and ecosystem services in the HHL, which would be local context dependent and informed by human related factors (Buckhard et al., 2012; Rey Benayas and Bullock, 2012). Other important conclusions that were drawn for general conservation include that whether making decisions based on biodiversity, ecosystem services, wetland potential or a combination, there would

be mostly expansion of core areas and some development of connectivity, which is important for landscape scale work.

Whilst recognising the challenges and limitations of integrating many disparate data sets at the landscape scale in this way, the results of these first two objectives represent the best and widest ranging data collection that has been achieved for the HHL. It also provides a sufficiently robust picture of the relationships amongst physical characteristics, biodiversity and ecosystem services to allow meaningful interpretation, and improving our understanding of the HHL and other similar wetland environments. It could provide a useful method for those on the ground, to inform their decisions about wetland restoration, as it is relatively simple and access to the data and models is possible in most cases. Previously it was identified that in order to aid landscape restoration, clear and rigorous science is needed (Maltby, 2010; Retford et al., 2012), which was attempted to be collected and used here.

8.3 Objective three

To identify the actual decision making processes used in the NIA project, focusing especially on the factors used to choose the location of intended wetland restoration.

This objective was covered in Chapter Six, with conclusions also discussed in Chapter Seven. It appeared that biodiversity and the concept of ecosystem services did have some role, but that most decisions were an interaction of multiple factors. Biodiversity appeared to be important to many parts of the decision making, with decisions often starting off with ideas of biodiversity, such as particular habitats, but then tempered by more practical factors. Previously, biodiversity decision making has mostly been around more traditional approaches, with a focus on particular species and habitats (Carpenter et al., 2006; Fisher and Brown, 2014). Although this did occur with the HHL NIA, there was also evidence of landscape scale approaches, such as ecological networks.

With the concept of ecosystem services the HHL Partnership used it in several ways, which is a big learning point of this work, as there is currently many

unknowns about the concept and its delivery (Sitas et al., 2013; Wu, 2013; Pan et al., 2013; Albert et al., 2014a; Pagella and Sinclair, 2014). There was uncertainty and inconsistency in how the concept was integrated in decision making, with some ecosystem services better understood than others and some being actively managed within the NIA but not recognised as ecosystem services. There were cases where they were using the concept implicitly, even they thought they struggled to use it. There was an evolution of the use of the concept over time; it was not included in earlier decisions, then there was some evidence of superficial use, however by the end some new ideas had been built into the HHL NIA using the concept.

There were many other factors that were involved in the decision making process, many of which were also identified in other studies in the literature (e.g. Maltby, 2010; Cotter et al., 2014). From discussions in the previous chapter there were three very influential other factors identified: conservation legacy, agriculture and resources (including funding), which all had big roles in the decision making process, and are therefore, factors that other similar projects should be aware of. This is especially the case for conservation legacy; a lot can be learnt from the use of this factor for delivering landscape scale projects in the future. Using conservation legacy allowed the partnership to join up much of their previous work and infrastructure so that more could be achieved and they were moving away from the traditional site by site scale. Using previous sites was a regulation of the NIA criteria, but far more should be made of it.

Overall, there was a range of factors that fed into decisions, with biodiversity often providing the theory and ideas, with other factors making the decisions more realistic. The concept of ecosystem services did have a role, but not as prominent as biodiversity. Some evidence suggests that these were tokenistic efforts with biodiversity and ecosystem services, which maybe the case with the concept of ecosystem services, but not so much with biodiversity and the actual services. The previous objective showed that the landscape is data rich, which needs to be used in decision making. Much of the literature states that a wider range of factors need to be included in addition to the scientific concepts, and that practical considerations are important (Maltby, 2010; Faith, 2012), which occurred in the HHL NIA decision making, but needs to become more accepted.

8.4 Objective four

To compare the information gathered in the previous objectives to inform wetland restoration, and consider the potential and limitations for biodiversity, the concept of ecosystem services and landscape scale work in wetland restoration.

This objective was covered in Chapter Seven. There were very few similarities between the two types of information and their outcomes; however, an important one to note was that they both involved elements of ecological networks, particularly expansion of core areas, and so both were attempting landscape scale restoration. The main disparity was the factors involved, and the level of involvement of each. With the first, both biodiversity and the concept of ecosystem services had significant roles, with only physical characteristics otherwise involved. Both biodiversity and the concept of ecosystem services also had a role in the second type, although not as significant, especially with latter. There were many other factors involved in the second outcome, which has been identified as a positive, as a variety of factors make decisions more realistic.

There are important lessons from this work that will have implications for similar projects, and points that also need to be overcome. For biodiversity, ecosystem services and landscape scale work it has been found here there more evidence and guidance is needed about how to deliver them on the ground. However, there is also literature that says that even if this information exists it does not necessarily make decisions better, because it is not always used (Fernandez, 2016). This is very difficult to overcome, but the suggestion was made in this work that the information is best provided through workshops, to allow discussions to occur amongst stakeholders. In terms of landscape scale wetland restoration it was also discussed that conservation legacy is a good approach. The other real-world factors need to become a more accepted part of the decision making process, because they attempt to make decisions more successful and attempt to achieve conservation in a multi-functional landscape.

8.5 Aim

The aim of this project is to: explore the potential and limitations of biodiversity, ecosystem services and “real-world” management in enhancing decision making for landscape scale wetland restoration.

It has been identified that all three of these ideas should and do have an influence in shaping landscape scale wetland restoration. However, in order to achieve more successful restoration, they could all be used more effectively and explicitly than it appears that they do currently. In Objective Four it was concluded that even though there are many calls for more information to improve the use of all these factors, actually more information does not necessarily mean better decisions, as the information is often not used. In order to improve this it has been suggested here that guidance should be delivered by workshops that allow for discussion between stakeholders.

Any conservation decisions need to have clear objectives that cover all of these aspects discussed (Piper et al., 2006; Arts and Buzier, 2009; Jasonoff, 2010; Maltby, 2010; Retford et al., 2012), and this work has shown it is possible to utilise data and models to have more rigorous evidence for decisions, as well as considering all of the more human driven factors. This work has substantially increased the understanding of biodiversity, the concept of ecosystem services and the real-world factors that are all needed for effective landscape scale wetland restoration, as well as the exploring the interactions and constraints between each, which has academic merit as well as a key tool for management. However, the work has been about providing information, not dictating values. It has also addressed the critical gap in understanding the combined role of the different factors in shaping landscape scale wetland restoration, as well as providing a basis for a framework for better using environmental data to inform decision making that could be used in a wide range of other situations.

8.6 Implications and beyond the HHL case study

Many of the ideas discussed, can be taken beyond the HHL case study and applied with the many real-world landscape scale conservation projects across

the UK, for example in the Somerset Levels, and perhaps even extend further beyond. All of the conclusions on the roles of biodiversity, the concept of ecosystem, the other real-world factors and landscape scale work could be applied in order to make other projects more successful, and so that we continue to learn from projects on the ground (Sutherland et al., 2012), to build up an even bigger evidence base. It is also hoped that this work will have implications for the way governmental organisations provide guidance to conservation projects and perhaps even effect policy.

8.7 Future work

There are several directions that further work could be taken, as has already been identified throughout the thesis. With the spatial mapping, a wider range of metrics of biodiversity, ecosystem services and physical characteristics could be mapped in order to make the restoration suggestions even more detailed. This could involve the collection of primary data on the ground and of a wider range of modelling techniques. Also, it would be useful to have data at much smaller scales than the grid squares used for the restoration options, in order to target restoration suggestions even more specifically. It has been identified in this work that the models used were useful, but working to make these more user friendly and transferrable across different landscape types would have big improvements. The approach taken for targeting restoration locations in this work was to build on what we already have regarding biodiversity and ecosystem services. This is just one approach, and so there could be scope for more work using different approaches, such as looking at what areas have the potential to deliver ecosystem services.

With the roles of biodiversity, the concept of ecosystem services and the real-world factors, it would be valuable to further investigate this with other similar projects to see if they are also getting the same levels of use of the concepts and the same other factors being involved. If more research is done it could help provide better guidance for making conservation decisions on the ground and for what format this guidance should be in. The need to for guidance of best practice has been identified from this work. Particularly, more needs to be done with the

concept of ecosystem services, because it has already been identified that the concept could be of more benefit to doing landscape scale restoration and in overcoming conflicts within landscapes, such as those that exist between agriculture and conservation.

Finally, something that was touched on in this thesis is partnership working, but this could be investigated in much more detail to see whether the benefits of working in one outweighs the negatives. Therefore, this work opens up many more avenues of interesting and useful research to contribute to this research area of landscape scale conservation, which could be vital in the future with the increasing threats that our natural environment faces.

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Appendices

Appendix A: Protected Areas Designations

Information was provided by Natural England.

A:1 SSSIs

Site Name	Reason for designation
Barn Hill Meadows	The site is important for its herb-rich, unimproved, neutral grassland (including important grass species), a habitat now uncommon in the intensively farmed landscape of Humberside and in lowland England generally.
Barrow Hills Sandpit	Comprises a fine example of species-rich grassland and scrub developed on freely-draining unconsolidated sands of glacial origin.
Belshaw	Important because it supports a colony of the greater yellow-rattle plant, which occurs within a remnant of neutral grassland with grasses, herbs and other plant species.
Brighton Meadows	Notified for its nationally and internationally important alluvial flood meadow plant community (dominated by grasses) and its outstanding assemblage of breeding birds (such as waders) associated with lowland damp grasslands.
Burr Closes Selby	A small area of damp alluvial meadowland, agriculturally unimproved, rich in flowering plant species (grasses and herbs) and noted locality for the Forester moth.
Chesterfield Canal	Supports a nationally uncommon aquatic plant community characteristic of the brackish, eutrophic (nutrient-rich) water. The flora includes a number of nationally scarce species.
Crowle Barrow Pits	Includes a variety of habitats including alder carr, scrub, fen and open water in which several locally uncommon plant species occur.
Derwent Ings	A series of neutral alluvial flood meadows, fen and swamp communities and freshwater habitats, with many plants, breeding wetland birds and invertebrates, which includes up to 16 species of damselflies and dragonflies.
Epworth Turbarry	An area of relict peat vegetation in low-lying carr land, which is partially covered by blown sand, and in the past was extensively cut. The plant communities represented are birch woodland, heathland and fen. Breeding birds include teal, snipe and long-eared owl.
Eskamhorn Meadows	A nationally important site for species-rich neutral grassland.
Hatfield Chase Ditches	A large area of former marsh and wetland, which has been extensively drained for agriculture, and split by a complex network of ditches that contain a rich assemblage of aquatic and emergent plants, with invertebrates and water voles.
Hatfield Moors	A once extensive lowland raised bog with a system of drainage ditches within adjacent agricultural land. The lowland peat bogs are a nationally rare habitat. The site has also been notified because of its important breeding bird and insect populations; a restricted representative flora and fauna persists within a mosaic of mire and dry heath habitats beneath birch scrub.
Haxey Grange & Fen	Principally a fen site with complementary areas of unimproved neutral grassland, willow scrub and woodland. It is the best example of primary fen habitat known in South Humberside and is particularly important for invertebrates.
Haxey Turbarry	This is a relict bog which was formerly extensively dug for peat, and is now largely colonised by birch, grasses, mosses, fen species and breeding birds.
Hewsons's Field	One of the few remaining fragments of neutral unimproved grassland, with many grass and herb species.
Humber Estuary	There is the estuary (with its component habitats of intertidal mudflats and sandflats and coastal saltmarsh) and the associated saline lagoons, sand dunes and standing waters, with wild fowl, waders and seals.
Kirkby Wharfe	Low-lying land adjacent to the dyke supporting a rich marshland flora, with drier neutral grassland at the high margins. Dominated by sedges, rushes and herbs.

Laughton Common	Nationally notable areas of lowland acid grassland, inland dune grassland and lowland heath which are scarce in the county and across England.
Mattersey Hill Marsh	The site comprises one of the best examples of mixed marsh in Nottinghamshire and is representative of marsh communities in Central and Eastern England.
Misson Line Bank	Contains some of the best remaining examples of eutrophic open water, marsh and base-poor fen communities in Nottinghamshire.
Misson Training Area	One of the largest remaining tracts of fenland; supports a diverse range of semi-natural habitats, including standing open water, tall-herb fen, unimproved neutral and acidic grassland, dry oak woodland and nationally restricted wet woodland types. There is a breeding bird community and a rich invertebrate fauna.
Mother Drain, Misterton	A drainage channel running parallel to the River Idle, supporting an exceptionally rich invertebrate fauna, which includes notable assemblages of dragonflies, water beetles, and a rare moth.
Owston Hay Meadows	This is the second most important example of neutral grassland, hay meadow habitat in South Yorkshire. A rich assemblage of plants has been retained with over 90 species, where grasses and herbs are dominant.
Potteric Carr	A mosaic of open water, reed bed, wet grassland and carr habitats, which now represents the largest and most diverse wetland of its type in the county, supporting abundant aquatic flora, invertebrates and birds.
River Derwent	Considered to represent one of the best British examples of the classic river profile, supports diverse communities of aquatic flora and fauna (invertebrates, dragonflies, fish and otters), many elements of which are nationally significant.
River Idle Washlands	The site comprises good examples of wet grassland plant communities, and attracts large numbers of wintering and passage waterfowl.
Rush Furlong	Relict of the Isle of Axholme strip-farming system, with many grasses and herbs.
Sandall Beat	The woodland is the product of the afforestation of former pasture, fen and heath, which now represents the largest predominantly deciduous woodland in lowland South Yorkshire, and supports diverse woodland flora and fauna.
Scotton & Laughton Forest Ponds	Peaty heathland pools associated with open acid grassland, birch woodland and a distinctive marginal wetland vegetation. It represents the county's largest resource of this nationally scarce plant community, with many grasses and mosses.
Scrooby Top Quarry	A series of medium-coarse pebbly river-laid sandstones with cross-bedding structures typical of transverse burial bars, which provides an important contribution to our overall understanding of ancient river systems, as well as an insight into the palaeogeography of this region during the Triassic period.
Shirley Pool	The site contains excellent examples of wetland habitats including open water, reed swamp, tall fen, wet neutral grassland and carr which grades into birch-oak woodland. It is the most natural wetland of this type in South Yorkshire.
Skipworth Common	An extensive tract of heathland on a spur of glacial sands; the vegetation includes both dry and wet heath, poor fen, reed swamp, bracken, open water and birch woodland, with moths, dragonflies and breeding birds.
South Cliffe Common	Consists of a mixture of heathland and acidic grassland, it is important as a remnant of once much more widespread habitats, now substantially reduced by agricultural improvement and conifer planting, with many invertebrates and birds.
Styrrup Quarry	A succession of accreted sand bodies in a section approximately at right angles to the palaeocurrent direction. Study of these structures provides a valuable insight into the nature of British Triassic palaeoenvironments.
Sutton & Lound Gravel Pits	Extensive areas of open water and margins which support an exceptionally rich assemblage of breeding wetland birds and a nationally important population of wintering gadwall.
Thorne, Crowle & Goole Moor	These moors form the largest extent of lowland raised mire in England, even though much modified by peat cutting. Remnants of the original flora and fauna, while fen habits, containing rich assemblages of species, have also developed.
Tuetoos Hills	Supports an important mosaic of dry acid grassland vegetation including an inland example of acid dune grassland dominated by sand sedge.

Went Ings Meadows	The meadows are developed upon silty clay alluvial soils which are subject in parts to waterlogging and seasonal flooding, and constitute the best example of unimproved neutral grassland known in South Yorkshire.
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A:2 Ramsar

Site Name	Reason for designation
Lower Derwent Valley	The site represents one of the most important examples of traditionally managed species-rich alluvial flood meadow habitat remaining in the UK, with a rich assemblage of wetland invertebrates, including 16 species of dragonfly and damselfly, and is a staging post for passage birds in spring.
Humber Estuary	A representative example of a near-natural estuary with the following component habitats: dune systems and humid dune slacks, estuarine waters, intertidal mud and sand flats, saltmarshes, and coastal brackish/saline lagoons, each with different dominant plant communities. There is also a colony of grey seals and a breeding site for natterjack toads.

A:3 NNRs

Site Name	Reason for designation
All	NNRs contain examples of some of the most important natural and semi-natural terrestrial and coastal ecosystems. They are managed to conserve habitats or to provide special opportunities for scientific study of the habitats communities and species represented within them. In addition they may be managed to provide public recreation and natural heritage interests.
Humberhead Peatlands	The boggy, lowland mire that makes up the reserve is one of the country's rarest and most threatened habitats.
Lower Derwent Valley	Made up of a series of flood meadows, pastures and woodlands. The reserve supports a rich diversity of plant species and outstanding populations of breeding and wintering birds.
Skipwith Common	One of the last remaining areas of northern lowland heath in England. An incredible variety of plants and animals depend on the Common for their survival.

A:4 LNRs

Site Name	Reason for designation
Axholme Line	Neutral calcareous grassland.
Barlow Common	The site has a mosaic of woodland, wetland, reedbeds and four large ponds. 140 species of birds have been recorded on site, over 200 species of plants, 21 species of butterfly, water voles and roe deer.
Daneshill	The dominant habitat is the lakes, associated wetland, scrub and grassland, which are important for over-wintering waterfowl, amphibians, dragonflies, butterflies, invertebrates, mosses and liverwort communities.
Eastrington Ponds	The large pond has an artificial island and is good for ducks, geese, great crested grebes, insects, bats and water voles. Wildflowers include common spotted, twayblade and bee orchids.
Hatchell Wood	The definitive character of the area is its wooded appearance from the many oak trees and other remnants of Hatchell Wood and Whin Hill Plantation, within a residential area, with lots of hedge and shrub planting.
Howden Marsh	Registered common. The site is an old fenland marsh much of which has never been drained. It is particularly rich in water beetles and water voles.
Mayfield & Broom Park	Orchid population on Broom Park site.

Owlet	Birch, oak and pine areas are interspersed among more open heath while scattered throughout the area find mature oak trees. Remnant heath vegetation occurs on more open areas and is home to a wealth of butterflies.
Owston Ferry Common	The meadow habitat attracts many birds and butterflies. A section of the site is a Scheduled Ancient Monument dating back to the 11th Century.
Retford Cemetery	Victorian cemetery believed to be the only working cemetery declared a Local Nature Reserve. There is a variety of native and non-native mature trees and grassland with wildflowers. The site is of county importance for bats. Plants include spring beauty, meadow saxifrage and bluebells.
Sandall Beat	Woodland, planted on open land in early 19 Century, small area of fen and a little heath
Sugar Mills Pond	Former brickworks and sugar refinery next to the Aire and Calder Navigation canal with two small lakes. Mature trees enclose the site from the surrounding farmland and open meadows. About 70 species of resident birds have been recorded, with also water voles and grass snakes.

A:5 SPAs

Site Name	Reason for designation
Lower Derwent Valley	An internationally important assemblage of birds. Over winter the area regularly supports waterfowl.
Humber Estuary	An internationally important assemblage of birds. Over winter the area regularly supports waterfowl.
Thorne and Hatfield Moors	During the breeding season the area regularly supports: Caprimulgus europaeus, nightjar caprimulgus and others.

A:6: SACs

Site Name	Reason for designation
Skipwith Common	Certain habitats are the primary reason for selection of this site: Northern Atlantic wet heaths the most extensive of its type in the north of England. The wet heath is part of transitions from open water, fen, reed and swap to European dry heaths and other habitats. The site has nearly 80 species of birds recorded.
River Derwent	Water courses of plain to montane levels. The primary reason for selection of this site are the river lamprey populations and the Otters.
Lower Derwent Valley	Contains a greater area of high-quality examples of lowland hay meadows than any other UK site and encompasses the majority of this habitat type occurring in the Vale of York. The abundance of the rare narrow-leaved water-dropwort <i>Oenanthe silaifolia</i> is a notable feature.
Hatfield Moor	Degraded raised bogs still capable of natural regeneration Like Thorne Moors, Hatfield Moors is a remnant of the once-extensive bog and fen peatlands within the Humberhead Levels, and is still the second-largest area of extant lowland raised bog peat in England. Refugia of vegetation have survived as rather dry heathland and as birch <i>Betula</i> woodland. The bog is also notable for its invertebrate fauna, which includes the mire pill beetle.
Thorne Moor	Degraded raised bogs still capable of natural regeneration Thorne Moor is England's largest area of raised bog. The restored secondary surface is rich in species of Active raised bogs with bog-mosses.
Humber Estuary	The Humber is the second-largest coastal plain estuary in the UK, and the largest coastal plain estuary on the east coast of Britain. Habitats within the Humber Estuary include Atlantic salt meadows and a range of sand dune types in the outer estuary, together with subtidal sandbanks, extensive intertidal mudflats, glasswort beds and coastal lagoon., with many fish species and seals.

Appendix B: Ethics information for the interview process

Information Sheet for Informed Consent

Ecological priorities and real-world governance in the restoration of wetlands in the Humberhead Levels landscape: Do they differ and how can this be overcome?

Funding: NERC Biodiversity and Ecosystem Services Sustainability Programme

Researcher: Kate Orgill, PhD Student, Department of Geography, University of Sheffield

Lead supervisor:

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Co-supervisors:

Professor Lorraine Maltby, Animal and Plant Science, University of Sheffield

Professor Philip Warren, Animal and Plant Science, University of Sheffield

Dr Liz Sharp, Geographical and Environmental Sciences, University of Bradford

Aim

The PhD project will address the question: *what are the relative influences of ecological principles of biodiversity and other ecosystem service benefits and 'real-world governance' issues in shaping landscape-scale habitat restoration?*

Objectives

1. To identify the form and spatial structure of current wetland patches in HHL and map this onto known information on major biodiversity patterns, indicators of ecosystem service provision and metrics of ecological functionality.
2. To use the information from objective one to see the level of agreement between the current wetlands and the wide range of chosen indicators, and then also compare this to the areas identified to be restored.
3. To identify the decision making process used to choose the location and form of the NIA restoration programme.
4. To compare the two different approaches from objectives one and three, and study what causes the disparity between results.

5. To assess the potential for information exchange between myself and the HHL NIA in order to attempt to resolve the differences between ecological and societal restoration preferences.

Anticipated Outcomes

- Increase the understanding of biological outcomes and ecosystem services for wetland research.
- Provide valuable information for the HHL NIA.
- A framework for other landscape-scale restoration projects in the future.

Confidentiality and Data Storage

The interviews will not be carried out confidentially but there will be a chance for each participant to view the transcript before any analysis is carried out. At this point any sections that the participant wishes to become anonymised or removed will be dealt with. If the participant is under agreement the interview will be audio recorded. The information will be stored on a computer and hard drive with password protection that only Kate Orgill will have access to.

Contact Details

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