A Re-wetted Land Use Capability Assessment for the North West of England

By

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Abstract:

In the wake of the Millennium Ecosystem Assessment, there has been an increased agenda surrounding the provision of ecosystem services within the UK's landscapes, highlighting the need for sustainable forms of agriculture. This coincides with recent changes to how the Department for Environment, Food and Rural Affairs (Defra) allocate funding for artificial land drainage; focusing on the protection and people and property, instead of sparsely-populated rural land. With the proposed withdrawal from current drainage practices, large areas of low-lying land will likely experience re-wetting, creating conditions unsuitable for conventional agriculture. This provides a unique opportunity for decision-makers to implement novel and innovative forms of management strategies that tackle the challenges faced by land uses dependent on drainage.

The aim of this study is to adopt an ecosystem service-based approach in creating a decision-support framework that quantifies the current provision of these services, signifying the potential benefits and implications associated with a change in the *status quo*. Two similar, yet subtly different study catchments were analysed within the context of re-wetting; the Alt Crossens (Merseyside) and Lyth Valley (Kendal). The Land Utilisation and Capability model (LUCI) was adopted to generate informative outputs that represent the provision of seven different ecosystem services. Paludiculture, the productive use of wet peatlands, was examined to determine the feasibility in providing an inclusive solution to stakeholders in areas anticipated to experience re-wetting. It was identified that the return of wetland conditions, combined with the transition away from conventional agriculture, there was an improvement in the provision of flood mitigation, carbon sequestration and storage, as well as decreased levels of nitrogen and phosphorus loading in both areas. Given the low economic value associated with improved grasslands, there was a net increase in the economic return where paludiculture was present. Whereas in the Alt Crossens, an area dominated by arable farming, there was a clear net loss in the economic value of the land. However, the creation of a phased-approach in limiting the withdrawal of land drainage and subsequent extent of land drainage, was shown to limit the economic and agricultural losses, whilst allowing land users to realise the full potential of multiple ecosystem service provision.

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1 Introduction

For the past 300 years in the UK, low-lying wetlands have been targeted by humans to provide fertile agricultural land through various drainage practices (Verhoeven and Setter, 2009). As a consequence of this, alongside the advancement of farming technologies, there are opportunities for humankind to sustain ever-increasing agricultural yields, however these place an increased strain upon the stability these wetlands areas.

As ‘drainage for agriculture’ practices have become more embedded within forms of agriculture in the UK, land users (principally farmers) have become fundamentally dependent on the ability to grow crops in fertile, peaty soils where there is a reduced risk of surface flooding and waterlogging. The prioritisation of agriculture in this sense has come at great expense to the functioning of natural wetlands, and subsequent loss of the benefits previously offered to humans. This can be observed for example through methods of cultivation, such as the tillage of soils, which causes a serious level of peat and soil degradation (Gregory et al., 2015).

Land drainage in the United Kingdom requires a significant capital investment, the majority of which is publicly-funded via the Department for Environment, Food and Rural Affairs (Defra). This funding is directed towards the Environment Agency (EA) for its role in overseeing the management and operation of reducing the risk of flooding in England (LCC, 2014). However, in 2011, the EA published a proposal to withdraw from non-essential forms of land drainage in line with updated policies established by the Secretary of State for Defra (LCC, 2014). This restructuring has prompted a planned withdrawal from the present levels of funding for water management in many agricultural areas.

In the absence of alternative funding to cover the costs and management of drainage practices, these low-lying rural areas will likely experience re-wetting, the re-establishment of the natural water table, producing less favoured growing conditions for conventional agriculture. Given this, there is the requirement for a change in how these areas are utilised; a transition back towards a landscape with no publicly-funded methods of drainage will lead to increased water tables as the land begins to gradually re-wet. Therefore, it is vital to find alternative land use solutions for these areas which are more suited to wetter soil conditions.

The concept of ecosystem services (ES), the goods and services from ecosystems that benefit humans, have been touched upon in within many academic studies in the past few decades (Costanza et al., 1997). However, arguably since the publication of the Millennium Ecosystem

Assessment (MEA) in 2005, there has been an ever-increasing awareness surrounding the sustainability of ecosystem services. These services are now widely recognised as the benefits that ecosystems and the environment contribute towards the wellbeing of humans (MEA, 2005, Haines-Young and Potschin, 2011). From an economic perspective, the value of the recognised benefits to humans account for, on average, $33 trillion per year (Costanza et al., 1997). The importance of these services was further highlighted in the MEA (2005), which reported that more than 60% of global ecosystem services are being unsustainably utilised or degraded by humans. This has been largely influenced by the growing demand and preferential provision of a specific ES, such as agricultural production, at the expense of other low-priority benefits.

This preference towards singular ES provision in satisfying the demand for increased agricultural production has largely underpinned decision-making within politics. However, a focal point in the MEA was the importance of the condition of different ecosystem services and their ability to contribute towards promoting an agenda for environmental sustainability within legislation (Carpenter et al., 2009). Research has identified how there is significant scope for the improvement for all ecosystem services, specifically 'regulating' services which have previously been under-valued within environmental legislation (Jackson et al., 2013). At present, agri-environment schemes provide subsidies to land managers for examples of environmental stewardship in agriculture as a method of delivering environmental benefits (Natural England, 2013). However, the success of these schemes have been predominantly limited biodiversity, water quality and soil protection benefits (Boatman et al., 2013). This stimulates discussion surrounding the effectiveness of present agri-environment schemes, signifying the importance of a 'combined approach' in ensuring the provision of various ecosystem services for future generations.

There has been a clear emergence of ecosystem service-based models, developed to represent and quantify the provision and status of different services since the MEA in 2005 (Pandeya et al., 2016). These have been largely utilised to help represent the complex interdependences in the fluxes and stores of ES as a method of informing decision makers about how best to tackle the challenge of implementing environmental sustainability. These models have been developed to be used on a variety of spatial scales for both singular and multiple services (Bagstad et al., 2013). Each model possesses unique differences in their functionality and ability to represent services at varying levels of technical detail, highlighting the importance of selecting models with the appropriate characteristics for each application.

The Land Utilisation and Capability Indicator (LUCI) model was adopted for this study to calculate the status and ability for trade-offs between different ecosystem services, working at both a field and catchment scale (Bagstad et al., 2013). LUCI generated outputs for seven different ecosystem services (carbon, erosion, flood mitigation, agricultural productivity, nitrogen, phosphorus and habitat suitability). A variety of scenarios were generated, indicative of the anticipated condition of the landscape, based on the proposed transition away from present land drainage practices. This provided a series of informative outputs illustrating the influence that changes in land use and water management have upon the status different ecosystem services.

This study assessed the current state of ecosystem services for two catchments in the North West of England (*Figure 1*); the Alt Crossens near Merseyside, and Lyth Valley in the Lake District. A re-wetted land use capability assessment (ReLUCA) was conducted to determine how both sites would be influenced by varying degrees of re-wetting, granting a unique ability to test alternative land management strategies which could provide innovative solutions to the current challenges faced by wetland areas. This involved a multi-faceted approach, whereby the potential for the landscape to return to its natural water table level was examined, before assessing the potential benefits and losses in terms of specific ecosystem services. This study examines the capacity for alternative land management strategies to be adopted, benefiting multiple stakeholders alongside the sustainability of the environment. This presents a unique opportunity to critically analyse the potential for alternative land management strategies, better-suited to future environmental conditions, to be introduced within the context of the North West England.

Figure 1: *The location of the Alt Crossens and Lyth Valley within North West England.*

1.1 Overarching goals

The aim of this thesis is to assess the current state of ecosystem services for two low-lying coastal catchments in the North West of England; each having previously been wetland areas, which have since been altered for agricultural purposes.

1.1.1 Research objectives

a. To identify the extent and influence of re-wetting, based on the desired movement away from non-essential land drainage and change in landscape conditions.

b. Adopt an ecosystem service-based modelling framework capable of assessing the provision of different services under a variety of drainage and LULC conditions.

c. Assess the provision of multiple ecosystem services for each scenario, signifying where trade-offs and co-benefits between modelled services could be implemented.

d. Introduce alternative forms of land use to candidate areas where present forms of land use would be unfeasible under re-wetted conditions, whilst simultaneously improving the provision of ecosystem services.

e. Quantify the net gains and losses in the provision of different ecosystem services, based on a change in the land management strategies between baseline (drained) and future (re-wetted) scenarios.

The focus of this research is to adopt an ecosystem services approach in evaluating the landscape under the current (drained) and future (re-wetted) conditions that can provide objective evidence to be utilised by stakeholders and implemented on a larger scale across the UK.

2 Literature review

This literature review examines how land is utilised for the benefit of humans and how ecosystem services are perceived within society. Focus will be made with regards to the importance of specific habitats which provide a number of these services and how landscapes have been modified to benefit agricultural systems. The provision of certain ecosystem services will be assessed within the context of environmental policies to identify how the provision of these benefits could be enhanced through alternative land management, and the ability to represent this to inform decision-makers.

2.1 What are ecosystems and how are they important?

Between 2001 and 2005, the Millennium Ecosystem Assessment was conducted to analyse the effects of ecosystem change upon human wellbeing (MEA, 2005). The principle aim of the MEA was to draw attention to the importance of different services, creating a database of scientific evidence to promote efforts in reversing the current declines in these services (Guerry et al., 2015). Whilst it is acknowledged that the concept of ecosystem services has existed for many decades, the focus of the MEA was to bring together information surrounding this topic and present it in an accessible and usable form for a wide variety of stakeholders, including members of the public, ecologists, economists and policy makers.

By extension, ecosystem services (ES) are the benefits and contributions that ecosystems contribute to human wellbeing (MEA, 2005, Haines-Young and Potschin, 2011). They are formed by the complex interactions between biotic (living) and abiotic (chemical and physical) processes which create outputs that can be utilised or consumed by humans (Haines-Young and Potschin, 2011). These services are so diverse and widespread within society that humans are fundamentally dependent upon their flows and interactions. However, it is important to acknowledge that in certain contexts, the provision of one particular service may be deemed more important than others (McMichael et al., 2005). However, since the MEA there has been a notable increase in the number of national governments, international organisations, businesses and nongovernmental organisations (NGOs) which now include information surrounding ecosystem services and natural capital (the store of these ES) within decision-making (Guerry et al., 2015).

2.2 Classifying ecosystem services

Subsequently, ecosystem services have been categorised by the MEA into four categories:

- **Provisioning services**. The products that are obtained from ecosystems, including: food, water, timber, fibre, natural medicines and fuel.
- **Regulating services**. The benefits provided by the regulation of processes within ecosystems, such as: climate regulation, flood mitigation, carbon sequestration, water quality maintenance, waste treatment and erosion control.
- **Cultural services**. The non-physical benefits that humans receive from ecosystems in the form of recreational activities, aesthetic enjoyment, and spiritual enrichment. Cultural services also provide cultural diversity, religious values, educational systems, social relations, cultural heritage and tourism (JNCC, 2014).
- **Supporting services**. The services which are fundamental for the continued production of all other ecosystem services, such as: soil formation, oxygen production, pollination, and nutrient cycling.

Conversely, Haines-Young and Potschin (2011) propose a hierarchical structure, whereby three 'service themes' are at the highest level (provisioning, regulation and maintenance, and cultural). These are followed by nine fundamental classes of service (*Table 1*) which relate to the grouping of ecosystem services highlighted in The Economics of Ecosystems and Biodiversity (TEEB). However, this excludes the provision of supporting services due to lack of direct utilisation or consumption, instead covering said services under a smaller classification. Similar approaches have been adopted by Small et al (2017), who highlight the growing the importance of differentiating between ecosystem use and ecosystem service use, meaning that cultural and recreational benefits would be classified under 'non-material benefits' of ecosystem use.

Table 1: *CICES Basic Structure and Relationship of Classes to TEEB Classification (Haines-Young and Potschin, 2011).*

CIES Theme	CIES Class	TEEB Categories			
Provisioning	Nutrition	Food	Water		
	Materials	Raw Materials	Genetic resources	Medicinal resources	Ornamental resources
	Energy				
Regulating and Maintenance	Regulation of wastes	Air purification	Waste treatment		
	Flow regulation	Disturbance prevention or moderation	Regulation of water flows	Erosion prevention	
	Regulation of physical environment	Climate regulation	Maintaining soil fertility		
	Regulation of biotic environment	Gene pool protection	Lifecycle maintenance	Pollination	Biological control
Cultural	Symbolic	Information for cognitive development			
	Intellectual and Experimental	Aestheic information	Inspiration for culture, art and design	Spiritual experience	Recreation and tourism

For the purposes of this research the classifications proposed by the MEA will be adopted throughout, due to the widespread utilisation of these categories within academic literature and inclusion of supporting services.

2.3 Human manipulation of services

Humans have historically heavily managed the surrounding environment to maximise particular ecosystem services at the direct and indirect expense of other services (Foley et al., 2005; McMichael et al., 2005). This is particularly noticeable since World War II, with an unprecedented expansion in agricultural systems to satisfy the ever-increasing food demand (Stoate et al., 2017). Despite the substantial net gains for human well-being and economic growth, there has been an observed rise in environmental cost, with the effects of degradation beginning to become apparent throughout the supply of ecosystem services on both a national and international scale. The MEA note that 15 out of the 24 recognised ecosystem services have been shown to be degraded or unsustainably used on a global scale, with only four being identified to have been enhanced over the last 50 years, three of which are directly associated with the production of food (MEA, 2005).

This, combined with the anticipated negative implications associated with present and future climate change, has raised awareness surrounding the unsustainable use of natural resources. The magnitude of which is beginning to be realised by the influence that environmental deterioration has upon the capabilities for the deliverance of ecosystem services, notably within agricultural production (Gregory et al., 2015). This has stimulated the discussion between academics as to whether environmental systems have been altered to such an extent that restoration back to their natural state would be both impractical and unfeasible (Hobbs, 2016). Instead, could an intermediate phase prove more achievable, providing an alternative solution that satisfies the requirement for continued long-term agricultural production, whilst replenishing natural capital stocks for future generations to utilise.

2.4 Wetland ecosystems

Wetlands have been targeted by humans to maximise agricultural yields for centuries. Wetlands are areas where high groundwater levels exist throughout the year, characterised by shallow water bodies or areas which frequently experience temporary inundation (Dawson et al., 2003). Covering almost 10% of the terrestrial land cover in Britain and the Republic of Ireland, wetlands are one of the most important natural resources on Earth, presenting a diverse array of habitats that support a significant share of the world's biodiversity, whilst also providing a wide range of ES (Dawson et al., 2003; Natural England, 2010).

Wetland habitats play an important role in the function and provision of wider benefits, being intrinsically tied to the flow and stock of natural capital (*Figure 2*) (Kadykalo and Findlay, 2015). More specifically, low-lying wetlands play an essential role in the natural regulation of water regimes and flood risk prevention. Within this, the rate of surface runoff is decreased by a larger field storage capacity and higher rates of friction, which improve interception and water retention when compared to areas of conventional agriculture (Acreman and Holden, 2003; Dawson et al., 2003). However, one of the most important services in wetlands is the development of peat soils, covering 3% of the global land area, peatlands store 600 gigatonnes of carbon (Yu et al., 2011). This signifies the disproportionate influence that peatlands have within the carbon cycle.

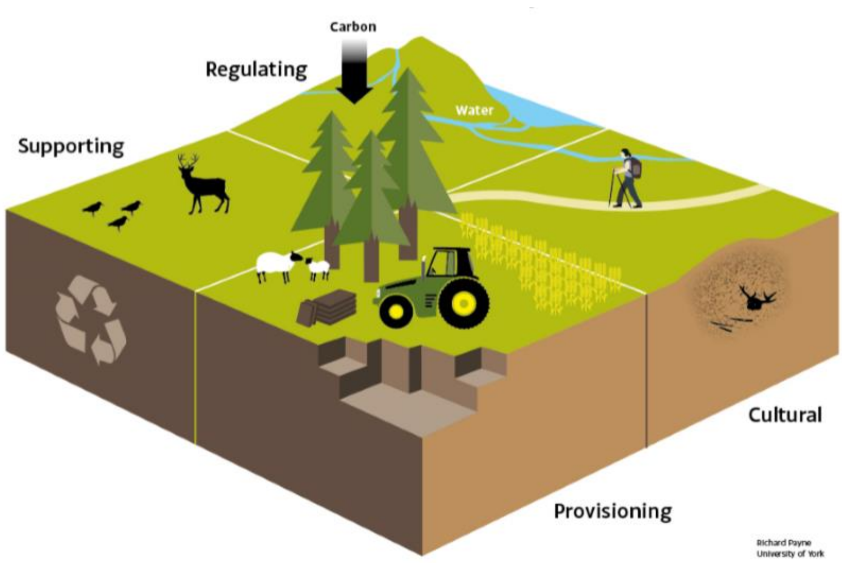


__Figure 2:__ Key forms of ecosystem services provided by wetlands (Payne and Jessop, 2018).

2.4.1 Types of wetland

Wetlands are characterised by the conditions in which they form; bogs experience waterlogged conditions through direct rainfall creating acidic, nutrient-poor conditions, whereas fens are influenced by groundwater and enriched by mineral soils (IUCN, 2014). The waterlogged conditions present in both habitats cause the incomplete breakdown of organic material, predominantly from the plant genus, *Sphagnum*, which accumulates to form peat, an organically rich deposit of soil (Price et al., 2003). There is no singular definition for peat or peatland, instead a variety of descriptions have been proposed by different interest groups based on the minimum depth of the organically-rich soil and mineral content (Holden et al., 2004; IUCN, 2014).

2.4.2 Raised bogs

Some peat bogs in the UK have been forming since the end of the last glacial period, around 10,000 years ago. Based on the known rates of peat formation (0.5-1.0mm per year) there are areas in the UK with depths in excess of 12 metre, acting as vast carbon stores. These systems are defined as raised bogs, providing hotspots for a variety of ecosystem services in low-lying areas (Bhatnagar et al., 2018). In the absence of human intervention *sphagnum spp.* continue to grow in wet conditions, increasing the elevation of the bog above the regional water table, forming a gentle-curving dome (Hughes and Barber, 2003; UK BAP, 2008). However, over the last 300 years many of the raised bogs in the UK have been cut and drained to disrupt the hydrological balance of the system, exposing the underlying peat which has been exploited as a source of fuel and, more recently, for agriculture (Hughes and Barber, 2003).

2.4.3 Wetland vegetation

Sphagnum spp. are an integral part of peat formation due to their resistant nature to decay in anaerobic conditions. Active bogs, also known as mires, consist of two layers: the acrotelm, a thin layer of peat-forming surface vegetation, and the catotelm, a thick layer of waterlogged peat soil which is largely dormant (IUCN, 2014). However, many peatlands face increasing disturbances from land use change, drainage regimes and peat removal (Swindles et al., 2016). As found by Price et al (2003), the removal of the acrotelm can result in the surface subsidence at rates of 3.7cm per year and decreases in hydraulic conductivity by over 75%. Exposing the catotelm to the surface environment through vegetation removal or drainage leads to drier soil conditions which allow other plant species to establish, creating future implications within the bog hydrology (IUCN, 2014). Peat wastage, the term associated with the shrinkage, subsidence and oxidation of peatlands is reported to exhibit rates of between 0.44-0.79m per century in pasture fields (*Figure 3*), with losses of up to 5m per century for arable farming and horticultural practices (EA, 2011).

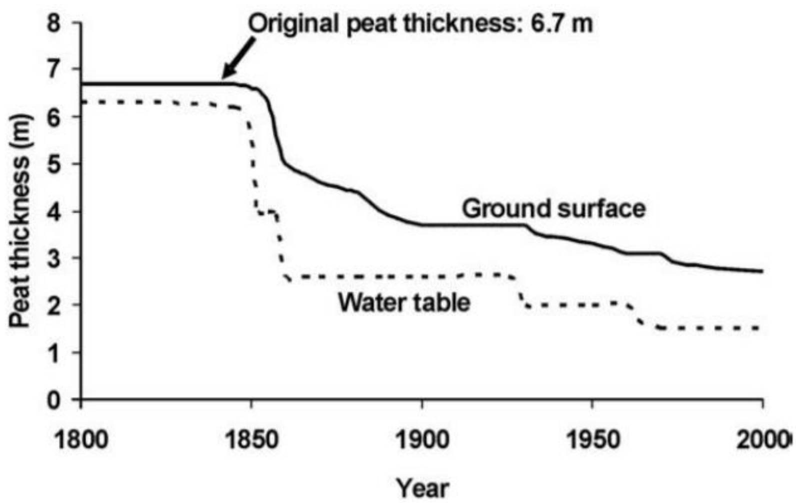


Figure 3: *Stages of peat subsidence after drainage of a peatland at Holme Fen, Cambridgeshire, UK (Dawson et al., 2009).*

2.5 Artificial drainage on peatlands

Peatlands have undergone significant artificial drainage for centuries, driven by the societal requirement for agricultural produce, timber, flood mitigation and the provision of energy (Holden et al., 2004). Artificial drainage is the process of controlling the hydrological function of a landscape, creating conditions where there is no excess water, lowering the relative depth of the water table to improve the conditions of agricultural soils and maximise harvestable yields (Lennartz et al., 2010).

The United Kingdom hosts some of the most extensively drained landscapes in Europe, with artificial drainage in peatlands playing a major role in maintaining present levels of agriculture (Holden et al., 2004). The continuation of these practices, notably the creation of channels that direct water to larger watercourses, lower water tables to expose organically rich soils for high-yield arable farming. This has led to 68% of organic soils in the UK having been drained for agricultural utilisation, most notably low-lying bogs with deep peat reserves (RRR, 2017). Land drainage has become so commonplace in the UK that of the estimated 3 million hectares of peatland, only 22% is found to be in a near-natural condition (Evans et al., 2017). More significantly, over the last 100 years undisturbed lowland raised bogs in the UK have decreased by 94% of their previous extent, with England holding the smallest share at just 500 hectares (UK BAP, 2008).

2.5.1 Implications of land drainage

Peatlands are extremely sensitive to small disturbances in the surrounding environment due to the complex processes and interactions that occur within each ecosystem. As highlighted by the IUCN (2014), one of the biggest misconceptions associated with land drainage in peatlands is that the effects are limited to the margins of the drains, highlighting that the wider effects are often unknown until they present significant challenges. In many cases the poor choice in management of low-lying wetlands has led to the severe degradation of a number of key ecosystem services in the surrounding landscape, notably carbon sequestration, nutrient cycling, soil structure and flood mitigation (Holden et al., 2007a; Evans et al., 2017).

Arable cropland is of particular importance when looking at the negative implications of land modification on peatlands, being associated with the highest greenhouse gas (GHG) emissions per unit area of any other land use (Evans et al., 2017). Despite the increase in the land fertility through the drainage and aeration of soils, rates of peat degradation are 50-100 times greater than peat accumulation, causing the transition of peatlands from a sink to a net source of carbon emissions (Environment Agency, 2011). Furthermore, national GHG inventories for the UK identify that the estimated net emissions from the land use exceed 23 Mt CO_2e yr^{-1}, a 6% share of the UK's total annual GHG emissions (Evans et al., 2017; BIES, 2019).

In a degraded bog the acrotelm, or peat-forming surface of the system, is lost due to a change in the vegetation cover and a reduced water table height. As the catotelm is exposed to aerobic conditions rapid decomposition of the peat follows, releasing carbon from the previously inert peat store (Holden et al., 2007b). This effect is worsened through the additional oxidation from tillage when wetlands are converted to arable cropland. The removal of the acrotelm and drying out of the peat causes significant changes to the soil structure, with even the slightest disruption taking decades to reverse. Together, these two processes act as positive feedbacks for one another, worsening the state of the peatland and reducing the relative thickness of the bog significantly over time.

Rates of peat subsidence and wastage have decreased in the last two decades when compared to the recorded loss over the previous 100 years (Dawson et al., 2009). This is a product of working with farmers alongside agri-environment schemes to increase the overall sustainability of agriculture in peatlands. Despite this, environmental degradation associated with agriculture is still considerable within low-lying peatlands. This signifies how current practices are not enough to prevent the loss of multiple ecosystem services, meaning that alternative methods of land management are required (Wichtmann et al., 2016).

2.6 Ecosystem services and policy

Since the release of the MEA, there has been a shift in the overarching social, economic and environmental responsibilities, which have highlighted a requirement for better examples of sustainable land utilisation (Martínez-López et al., 2019). Land managers must now aim to deliver a range of provisioning, regulating and cultural benefits to enhance the stability of the landscape, should the ability to provide a certain service fail in future (Emmett et al., 2016). This has prompted a renewed interest of long-term agricultural and environmental sustainability within policy, through updating previous agri-environmental schemes and strategies for both the UK and EU.

In order to establish policy or management strategies that maximise the trade-offs and co-benefits of different ecosystem services on a national scale, decision makers require the quantification of the flows and stocks of these benefits in the environment (Emmett et al., 2016; Norton et al., 2018). However, one of the problems faced by integrating ecosystem services within policy is the ability to quantify each service in a way that allows equal comparison. Economic appraisals are typically used to satisfy this when informing policy makers within decision support systems, enabling cost-benefit assessments (Fisher et al., 2008). However, it has been shown that these appraisals undervalue 'non-material' benefits which lack a means of providing a market value. This restricts the ability for these services to be represented in comparison to services that provide already established market values (Carpenter et al., 2006). Given the inclination towards agricultural land uses, there has historically been a clear focus away from cultural, supporting and regulating services within policy, due to the lack of economic analysis (Silva et al., 2013). Furthermore, Small et al (2017) propose that cultural benefits should be redefined and quantified as 'non-material ecosystem use' to bridge the gap when communicating the value of non-material ecosystem services within policy.

2.6.1 Environmental Net Gain

Despite an increased awareness surrounding environmental sustainability, human-induced deterioration has persisted in the UK. Everett et al (2010) link this to the economic decisions of stakeholders; once a certain level of income is achieved then environmental damage is considered and a trade-off is created. As this rate of loss persists, the complex processes that underpin numerous ecosystem services begin to deteriorate, reducing the relative value of the surrounding environment (AECOM, 2017). This highlights the importance of valuing ecosystem services beyond the realms of agriculture, ensuring that the effects from sustainable forms of land use are understood from both an environmental and economic perspective for land users.

Environmental Net Gain (ENG) is a term used to examine the "*measurable improvements for the environment*" and has increased in popularity as many businesses have begun to monitor their environmental contribution, complying with allocated targets (Weissgerber et al., 2019). On a national scale, ENG was recently proposed in the Government's 25-year Environmental Plan as an extension of Biodiversity Net Gain (BNG). Both schemes incorporate the principle that the environment and biodiversity should be left in a better state than before, with the UK

Government's Manifesto pledging to "*become the first generation to leave the environment in a better state that we found it*" (AECOM, 2017).

2.7 Movement away from land drainage

Land drainage in the United Kingdom requires a significant capital investment, the majority of which is publicly-funded via the Department for Environment, Food and Rural Affairs (Defra), suppled to the Environment Agency (EA) to manage the risks of flooding. In 2008, 35% of the Environment Agency's budgeted CO_2 emissions were associated with artificial land drainage (Environment Agency, 2008). As with a change in environmental legislation, company responsibility is a growing influence and the EA is a prime example of this, with their 'e:Mission' acting as a direct response towards meeting targets for mitigating against climate change. Therefore, the EA has committed to withdraw from non-essential drainage works as a method of satisfying carbon emission reductions, as well as saving both natural and economic capital (LCC, 2014). This coincides with updated Defra policy, focusing on the protection of people and properties from flooding, instead of sparsely-populated rural land.

Unless an alternative public body, such as an Internal Drainage Board (IDB), can be established to maintain the operation of present forms of land drainage, water tables will return to their natural levels and re-wetting will ensue. Therefore, it is vital to find innovative solutions so that the land use is better-suited to the anticipated wetter soil conditions, which are not viable for drainage-dependent agriculture. This has stimulated an on-going debate with landowners as reduced crop yields from conventional agriculture are expected (MEA, 2005). However, many benefits are presented by re-wetting, being shown to counteract the deterioration of associated ecosystem services within wetlands and contribute towards Environmental Net Gain (Wichtmann et al., 2016).

2.8 Future land management

Given the choice to discontinue non-essential pumping stations and watercourses management, there is a unique opportunity for the development of novel and innovative land management frameworks to increase the overall sustainability of the environment. However, with this comes a complex challenge in ensuring that the interests of different stakeholders are satisfied whilst meeting environmental objectives (Rawlins and Morris, 2010).

Based on the differing stakeholder requirements, there is no clear, singular management strategy that could accommodate this potential re-wetted landscape. Highlighting how, a multidisciplinary approach in managing the landscape would be better suited to meet the present challenges of re-wetting. Joosten et al (2012) signify how there are three outcomes from a decision support framework within this context: business as usual (maintenance of the *status quo*), conservation and restoration, and alternative management (rewet drained peatlands for other purposes).

2.8.1 Business as usual

Business as usual, or maintenance of the *status quo*, involves continuing current land management practices. There is much debate surrounding the ability to continue agriculture on peatlands without land drainage. It is widely accepted that th`e world needs to double its rate of food production by 2050 to be able to feed the population, and the same arguments can be made for food security within in UK (Ridley and Hill, 2016). Despite only 12% of the agricultural land in Britain being identified as peatlands, there is a disproportionately high level of food production in low-lying raised bogs due to the intensive practices on highly organic, deep soils (Kechavarzi et al., 2007). This begs the question that if these areas are re-wetted, where will a large proportion of the UK's future food production come from?

One possible option is the formation of Internal Drainage Boards (IDBs); assuming a new overseeing body is formed, maintenance of the *status quo* may continue, and intensive arable farming could persist on low-lying rural land. However, agricultural practices would continue to degrade the status of multiple ecosystem services for the benefit of food production in the short term. As the quality and volume of peatlands deteriorate through anthropogenic influences, the conditions for arable farming would worsen (Berge et al., 2017). Land that was once previously highly productive would eventually transition to marginal land of little value for agricultural production (Wichtmann et al., 2016).

2.8.2 Conservation and restoration

Peatland restoration is vital in ensuring that ecosystem services are maintained for future generations, acting as a cost-effective method of mitigating against other environmental challenges such as climate change (Bain et al., 2011). As found by Greifswald Mire Centre

(2015), efforts to better-manage drained wetlands require extensive capital investments, which, dependent on the condition of the peatland area, have varying rates of success. The key component of restoration is to ensure that a positive water balance is restored, creating waterlogged anaerobic conditions and the reintroduction of *sphagnum* species in the area (Bain et al., 2011; IUCN, 2016). A simple, yet effective way to satisfy this is to block drainage channels which decreases flow velocities, entrain sediment, and allow the water table to re-establish (Ramchunder et al., 2009). Thomas et al (2015) identify how the use of on-site peat and vegetation can prevent against large-scale drainage by creating individual cells in wetland areas, maintaining localised water levels. Similarly, it is crucial to remove any vegetation cover that could out-compete or slow hinder the reestablishment of *sphagnum* and other peat-forming species. Areas of woodland are particularly damaging in this instance, as the roots penetrate deep into peat soils, causing aerobic decomposition and peat subsidence (Joosten et al., 2012).

Bain et al (2011) reports how the successful restoration of low-lying bogs can occur as quickly as 5 years, with the full hydrological function of the bog returning over a much longer timescale, dependent on the degree of restoration and the extent of peat degradation. Removing land viable for agriculture is a sensitive issue; without providing viable solutions that address the future challenges posed by a 'wetter' landscape, there could be significant implications for the livelihoods of the land users (farmers) in surrounding areas (Morris et al., 2010). This signifies the challenges faced by large-scale conservation, with the success and timescale of restoration being dependent on the ability to move away from conventional forms of land use and the degree of degradation (IUCN, 2014).

2.9 Alternative management

Re-wetting, the process by which the water table is restored back to its natural level, is a viable option to safeguard against the problems faced by drained peatlands. However, this process prevents further drainage-dependent agriculture to continue (RRR, 2017). One strategy that has been proposed as an alternative is adaptive management; a change in the land use of drained peatlands to deliver wetter methods of arable farming (IUCN, 2018). Paludiculture is an example of this, enabling the production of renewable raw materials under wetter soil conditions whilst simultaneously preserving and replenishing the peat store (Joosten et al., 2012; Wichtmann et al., 2016). This process involves the introduction of alternative farming systems that reduce the negative impacts of agriculture in wetland environments, creating opportunities to maintain the livelihoods of land users in the absence of conventional farming (IUCN, 2018).

2.9.1 Types of paludiculture

The concept of paludiculture comes from the productive use of wetlands; aboveground biomass is harvested for human use, whilst the belowground biomass is left to enhance the peat store (Oehmke et al., 2017). Joosten et al (2012) identify six key themes within paludiculture that should be followed to ensure optimal land usage:

1. Minimising land drainage to reduce or prevent peat oxidation and wastage.
2. The cultivation of crops that are adapted to grow with a high soil moisture content.
3. Minimal tillage, limiting rates of peat oxidation.
4. The cultivation of permanent crops, ensuring complete cover all year round.
5. No land clearage through fire.
6. Limited (if any) application of fertiliser which leeches into watercourses.

Given these set of guidelines/requirements, it is possible to start attributing different crops to certain environmental settings, with a broad spectrum of plants species able to be commercially utilised. Wichtmann et al (2010) details these below in *Table 2*.

Table 2: *Different types of paludiculture, the ideal harvest time and market requirement for quality (**=high, *=medium and 0=low), as developed by Wichtmann et al (2010).*

	Utilisation	Plant growth	Harvest	Demand for quality
Agricultural	Ex situ fodder (hay, silage)	Wet meadows, reeds	Early summer	**
	In situ fodder (grazing)	Wet meadows, reeds	Whole year	**
	Litter	*Carex* meadows, reeds	Summer/autumn	0
	Compost	Wet meadows, reeds	Late summer	0
Industrial	Roofing material	Reeds	Winter	**
	Form-bodies	Wet meadows, reeds	Autumn/winter	*
	Construction (insulation)	*Phragmites* reeds	Winter	**/0
	Paper (plant cellulose)	*Phalaris-Phragmites* reeds	Winter	*
	Basket-ware	Willow shrubs	Autumn	**
	Timber/furniture/veneer	Alder	Frost	**
Energy	Direct combustion and gas	Alder/willow swamps, reeds	Autumn/winter	0
	Fermentation	Wet meadows, reeds	Early summer	*
	Liquid 'sun fuels'	Wet meadows, reeds	Whole year	0
Other	Officinal	Natural mires/plantations	Early summer	**
	Food	Natural mires/plantations	Summer/autumn	**
	Growing media (compost)	Peatmoss stands	Whole year	**

The cultivation and harvesting of crops for biomass appeals to policy makers through a three-pronged approach; not only does re-wetting enable the preservation and accumulation of peat, it also provides a source of income to affected land users whilst creating a source of renewable energy to replace of fossil fuels (Wichtmann and Joosten, 2007). More specifically, reed canary grass (*Phalaris arundinacea*) is favoured as a bioenergy crop because of its high rate of productivity (in excess of 12.7 tonnes of dry weight per hectare) that can be achieved alongside observable levels of peat accumulation (Wichtmann, 1999; Heinsoo et al., 2010).

2.9.2 Paludiculture and policy:

Acting as an inclusive solution, paludiculture could satisfy the requirements of multiple stakeholders simultaneously, ensuring that there is a future for both the livelihoods of farmers and continued provision of ecosystem services for future generations. With the Paris Agreement and Sustainable Development Goals calling for zero net carbon emissions by 2050 alongside no loss of productive land, paludiculture offers a unique and novel method to meet these demands (RRR, 2017). Additionally, paludiculture is the only known sustainable form of land use that is possible in degraded peatlands with marginal agricultural use (Oehmke et al., 2017). However, implementing this on a viable scale requires stakeholder engagement and the support from public incentives to create study and pilot sites that could create a market value for harvestable produce (IUCN, 2011; RRR, 2017).

This extends into the economic efficiency of paludiculture; being able to create an income from the utilisation and sale of the alternative produce alongside the economic value of the additional ecosystem services (Wichtmann et al., 2016). As stated in Paludiculture UK (2017), the management strategy should be self-supporting, but there will be a lag between initial investment and return. One solution is the sourcing of additional incomes through eco-innovation grants, and the added valued of ecosystem services that wetlands provide. Placing a value on these natural capital assets will aid the ability for decision-makers to recognise the benefits that paludiculture offers in both an environmental and economic sense.

As the UK's departure from the European Union expected is expected to be fulfilled in 2019/2020, Brexit offers a unique opportunity for the UK to restructure the funding available for practices which are currently poorly accounted for by the Common Agricultural Policy (CAP) and EU Agricultural Policy (Paludiculture UK, 2017). However, with Brexit comes an element of uncertainty in ascertaining future funding, with novel management strategies like paludiculture presenting an element of risk compared to 'ground-proofed' management methods. Because of this, the challenge of its large-scale adoption partially lies in the ability to gain support from farmers, land owners, politicians and businesses to ensure that paludiculture is both economically and environmentally viable on a commercial scale in the UK (Wichtmann et al., 2016; Paludiculture UK, 2017).

2.10 Ecosystem service modelling:

Technocentric valuation tools are designed to create innovative solutions to environmental challenges of land management, signifying their importance within a decision-making framework (Everard, 2013). More specifically, ecosystem service tools produce metrics that assess the quality, quantity, and economic value of different forms of land use (Guerry et al., 2012).

Numerous ecosystem service models have been developed to evaluate the past, present and future state of ecosystem services, each ranging in the level of complexity, sophistication and application for different environments (Sharps et al., 2017). Despite the ability to 'bridge the gap' between ecosystem service provision and spatial planning, a number of challenges are faced in the capacity to reliably represent each service, given constraints surrounding the availability of high-resolution data and the over-simplification of environmental processes (Palomo et al., 2018). Gopalakrishnan et al (2016) note how this has knock on effects on the environment by overestimating its capacity to support anthropogenic influences, leading to exacerbated degradation. Therefore, expert opinion and knowledge is always useful when augmenting analysis to ensure the reliability of the simulated outputs. Similarly, the ability to choose the appropriate tool remains difficult, with many of which often being too costly and time consuming to be utilised in landscape planning. Often a trade-off is made between the cost of operating the chosen tool at the expense of the level of detail and reliability of the analysis (Grêt-Regamey et al., 2017).

As more and more of these quantification tools continue to develop, decision-makers now face a new problem; there is now no clear model that can be used in every study, instead many different tools perform a variety functions well (Christin et al., 2016). Individual models may possess higher capabilities for specific services but lack the same level of accuracy for other services, meaning that multiple toolkits are often required for a comprehensive analysis.

This brings to light the wide range of data requirements and processing power prerequisites of models, identifying how a sacrifice in detail is necessary when identifying which model is best suited to the user's decision-making needs (Christin et al., 2016). This highlights the necessity to evaluate the suitability of models prior to conducting research, ensuring an efficient use of time and resources.

Despite the growing social, economic and environmental importance of ecosystem services within decision making, the ability to implement the knowledge of ecosystem services within landscape planning is still in its infancy (Albert et al., 2014). Drawbacks between these two topics arise when integrating the requirements and interests of different stakeholders, often through a multi-criteria decision analysis, within an ecosystem service model framework. Often the type, production and communication of ecosystem service information requires adaptations to be relevant within specific contexts (Albert et al., 2014). Similarly, problems arise when addressing specific environmental issues, and the ability guarantee a link between the outcomes of landscape changes and the objectives of each stakeholder (Albert et al., 2016). This step typically requires an additional monitoring phase, and with the nature of different ecosystem services, positive results may only be observed over long timeframes.

2.10.1 Decision-support tools:

In an attempt to compare the wide array of decision-support tools available for ecosystem service modelling, there have been a number of comparative assessments to determine the pros and cons of different modelling approaches. Bagstad et al (2013) provides an assessment for the performance of 17 different ecosystem service tools to realise their ability for widespread application. Given the context of this paper and its relevance to wetland ecosystem services, three spatially explicit tools were identified for comparison of use within this research: ARIES (ARtificial Intelligence for Ecosystem Services), InVEST (Integrated Valuation of Ecosystem Services and Trade-offs), and LUCI (Land Utilisation and Capability Indicator). Each model varies in its approach and assumptions, providing quantitative outputs at local and national scales, dependent on the resolution of data inputs and user-defined parameters (Sharps et al., 2017).

ARIES was developed as an online platform that supports a variety of ecosystem services, maintaining a simplistic structure that allows outputs biophysical outputs to be created at different spatial scales (Villa et al., 2014). Eight ecosystem service models are implemented within the model, with the possibility of further refinement from context-specific needs (Sharps et al., 2017). One of the benefits of ARIES is the level of generalisation, which enables outputs to be generated in situations when data inputs may be limited compared to other tools with more demanding requirements (Bagstad et al., 2013).

In contrast, InVEST combines land use and land cover (LULC) data with the fluxes and stores of ecosystem services to produce ecosystem service outputs in biophysical and economic terms (Bagstad et al., 2013). Initially the tool was built within ArcGIS, but has since developed into a free, standalone version that includes sixteen different services (Sharp et al., 2018). InVEST also features an ability to represent recreational services, an area that is scarcely touched within ecosystem service frameworks (Sharps et al., 2017). However, the level of detail for a wetland application could be questioned, due to insufficient level of detail required for certain services, such as the absence of information surrounding soil characteristics when calculating carbon fluxes (Bagstad et al., 2013).

LUCI is an extension of the Polyscape framework, which investigates the impacts of land management on ecosystem services at a sub-field level (Trodahl et al., 2017). Land cover maps can be updated to reflect different land management scenarios. LUCI quantifies biophysical conditions of the landscape, further classified into 'traffic light' maps that reflect the overall status of different ecosystem services, and the effect that LULC decisions have within the environment (Jackson et al., 2013). Bagstad et al (2013) identifies how LUCI shares a number of features found within other decision support frameworks, whilst being the only tool within an international review of ecosystem service models suitable for both landscape and site-scale modelling. LUCI only covers seven ecosystem services; however, the tool's requirement for high-resolution data and compatibility with user-defined parameters enables complex analysis within these services.

2.11 Summary:

Based on a review of the literature surrounding wetlands, it can be seen that a large proportion of low-lying rural land in the UK is utilised for agricultural purposes at present and is heavily dependent on artificial drainage. Many of these areas were previously wetlands, offering a wide array of ecosystem services for humans. However, historically anthropogenic influences have led to the loss or degradation of many of these services, focusing instead on maximising agricultural production. Whilst this switch from a natural environment to heavily-managed state has satisfied the ever-increasing demand for food, this transition is inconsistent with the growing importance of environmental sustainability and the UK's ability to meet climate change targets. Updated Defra policies now focus on protecting people and properties through land drainage, meaning that the drainage of sparsely populated rural land no is longer seen as a priority. Under future, re-wetted conditions conventional forms of agriculture are deemed unfeasible on a commercial scale, highlighting the need for alternative methods of land use better suited to future environmental conditions. Stakeholders need to be informed about the influence that these decisions have on the wider environment, including the gains and losses associated with proposed changes. Therefore, an ecosystem service-based assessment will be conducted to represent the biophysical and economic status of both present and future landscapes. Based on the provision of multiple ecosystem services alongside the anticipated spatial extent of re-wetting, candidate areas for future intervention will be identified. At present, there is a lack of land management options that offer an 'inclusive solution' in satisfying the challenges faced by maintaining agricultural productivity and improving environmental sustainability. This research will address this by creating a decision-based framework that supports the application of sustainable management in low-lying rural land, consistent with the present political and societal needs.

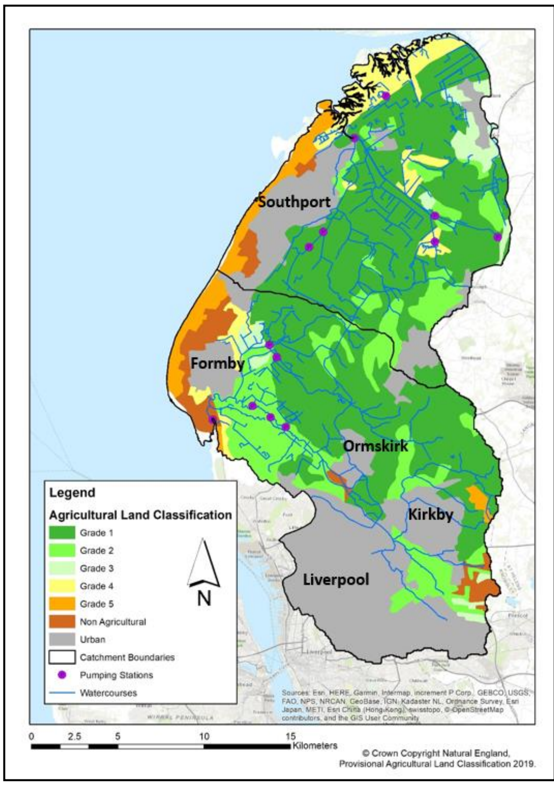
3 Materials and methods

The focus of this research is to develop a strategy for evaluating the landscape and condition of different ecosystem services under a current (drained) and future (re-wetted) soil condition to provide objective evidence that can be utilised by stakeholders. This study assesses the current state of ecosystem services in two catchments by assembling and manipulating nationally available datasets, augmented with localised knowledge, which are then processed by an ecosystem service model. Following the creation of a baseline scenario, a multiple-service analysis will be conducted to identify areas which exhibit poor overall ecosystem service provision, flagging areas suitable for alternative land use strategies. Building upon this, future scenarios will be created to reflect reduced levels of land drainage in these areas. Different land use and land cover (LULC) mosaics will be created based on two principles, establishing a land cover that better-suits the future degree of re-wetting whilst simultaneously ensuring that the overall condition of multiple ecosystem services is improved.

3.1 Site Descriptions

3.1.1 Alt Crossens

The Alt and Crossens (Alt Crossens) catchment is approximately 410km^2 in size, consisting of predominantly low-lying, agricultural land between the Ribble and Mersey Estuaries in South West Lancashire. Around 28% of the area is urbanised whilst 40% contains fertile, peaty soils that support a wide variety of agricultural uses (*Figure 4*). 60% of all Grade 1 and Grade 2 agricultural land in north west England is located in the Alt Crossens region (Environment Agency, 2009), symbolising the area's economic importance and ability to support a wide range of arable crops (*Figure 5*). A large proportion (26%) of the catchment lies less than 5 metres above sea level and is therefore naturally prone to flooding.



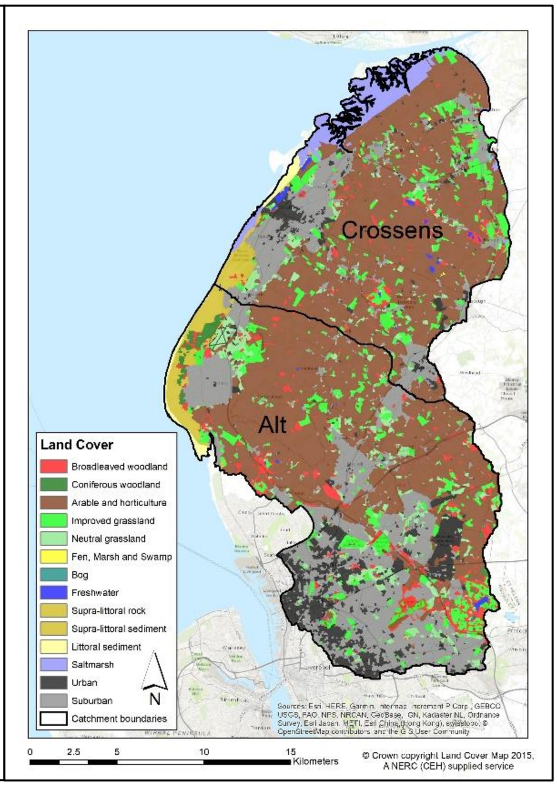


Figure 4: *The agricultural land classification of the Alt Crossens (left), source: Natural England 2019.*

Figure 5: *The land cover of the Alt Crossens (right), source: CEH Land Cover Map 2015.*

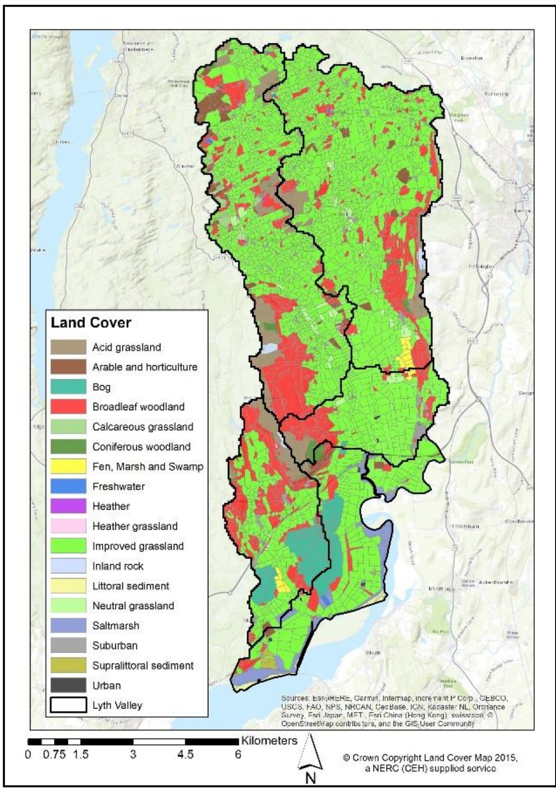
The low-lying nature of the Alt Crossens has led to the formation of wetlands, which historically dominated the land until the 18th Century, when drainage channels and gravity drains were established to allow arable farming to ensue. Over time, through peat wastage and subsidence, the relative elevation of the land has decreased, requiring a higher level of management and the introduction of pumping stations. This is observed at present with an extensive network of drainage channels, two major pumping stations (Altmouth and Crossens) and 11 smaller satellite stations, which collectively cost £3 million a year to operate (Environment Agency, 2010; Environment Agency 2011). Because of the topography of the land alongside previous surface lowering, the pumped drainage catchment is inherently complex. The Lower Alt System draining areas via the River Alt towards Altmouth pumping station, whereas the Crossens Pumped Drainage System provides flood alleviation via a number of drainage ditches towards Crossens pumping station.

The areas benefitting from land drainage in the Alt Crossens, known as Drainage Benefit Areas (DBAs), are of particular importance as they are strongly linked to drainage issues in the study catchments and their geographic location is largely bound to peat soils which facilitate intensive arable farming (EA, 2010; EA, 2011). Given the recent movement away from non-essential land drainage, there is the potential for large-scale changes in land management. However, due to the compartmentalisation of the land, as defined by these DBAs, there are opportunities for flexible local water management, ensuring that a more phased approach could be adopted when moving away from land drainage practices. Given the reported rates of peat wastage and the possibility of a change in the water management regime, the Alt Crossens is a highly important study area in determining the future land management under different environmental conditions and changing policy requirements.

3.1.2 Lyth Valley

Lyth Valley is located south west of Kendal, on the edge of the Lake District National Park in Cumbria. The Valley itself is approximately 88km^2 in size and forms on the lower extent of the River Gilpin catchment, feeding into the River Kent estuary. Much like the Alt Crossens, the topography of this catchment is predominantly low-lying, with the majority of the land lying 4-6 metres above sea level. Historically, this has provided conditions that favour the formation of wetlands and peat reserves within the flat valley floor. However, anthropogenic influences have driven a change in the land use historically, focusing on agriculture and peat extraction (Environment Agency, 2015). Traditional peat-related land practices include peat extraction to act as a fuel source, however given the large implications associated with peat removal and combustion, extraction no longer occurs within the area.

Lyth Valley is sparsely populated, with less than 2% of the land being classified as urban/suburban. The majority of the land is classified as improved grassland (63%), and broadleaf woodland (17%), with the remainder being composed of acid grassland, saltmarsh and peat bog (*Figure 6*). The catchment consists of Grade 3 (35%), Grade 4 (44%) and Grade 5 (21%) agricultural land (*Figure 7*). Because of this, Lyth Valley is dominated by livestock grazing on artificially drained land which ranges from ‘good to moderate’ and ‘poor’ (MAFF, 1988).



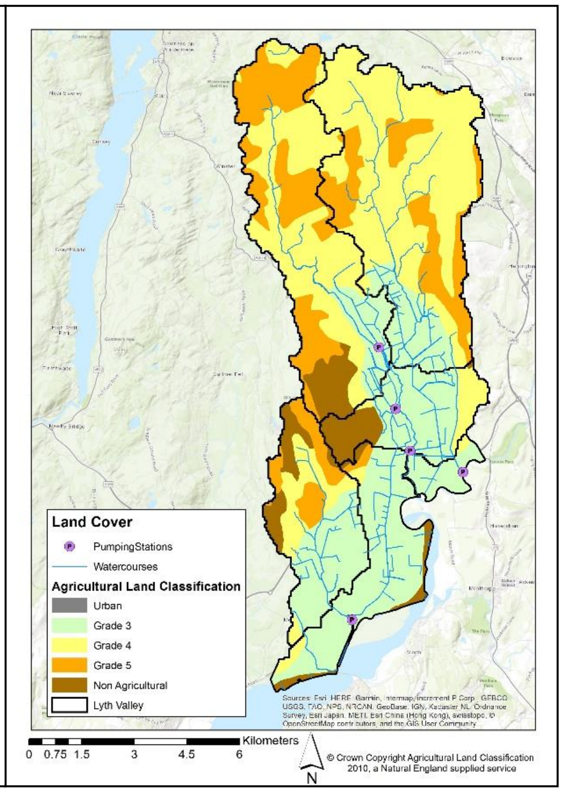


Figure 6: *the land cover of Lyth Valley (left), source: CEH Land Cover Map 2015.*

Figure 7: *The Agricultural Land Classification of Lyth Valley and associated sub catchments (right), source: Natural England (2019).*

Over the last 30-40 years the drainage regime of Lyth Valley has been maintained by the operation of five pumping stations and a complex network of embanked drainage channels. As with the Alt Crossens, given the topographic lowering through peat subsidence and lowering, improved drainage networks are required to prevent the waterlogging of soils to provide the conditions suitable for agriculture to persist (Environment Agency, 2009). The catchment is drained by the 'High Level System' and 'Low Level System', with the pumping discharge feeding into the River Gilpin (High) and River Kent Estuary (Low) (Environment Agency, 2015).

Lyth Valley has many similarities to the Alt Crossens, notably accommodating large areas of sparsely populated low-lying agricultural land, with a significant proportion of the area governed by complex drainage practices that involve a large amount of capital and investment to maintain. However, there are also many characteristics which make the two study areas quite different. Principally, the difference in presence of high-grade (1 and 2), fertile land in the Alt Crossens in comparison to predominantly low-grade (3-5) land in Lyth Valley (MAFF, 1988). There are also significant differences in the ways in which land is utilised in the two sites; arable farming dominates the Alt Crossens, whereas Lyth Valley is dominated by largely marginal livestock grazing.

3.2 Model selection

Technocentric valuation tools are often adopted to address current ecological problems associated with land management (Everard, 2013). Given this, the creation of strategies to produce a framework that enable ecosystem services to be quantified, assessed and simulated within future scenarios holds the key to providing the solutions to these challenges. As stated by Bagstad et al (2013), “to enter widespread use, ecosystem services assessments need to be quantifiable, replicable, credible, flexible, and affordable”. Since the Millennium Ecosystem Assessment in 2005, a number of different ecosystem service toolkits have been developed for a variety of uses, each hosting subtle differences in their ability to represent ecosystem services based on the environmental setting. Therefore, it is crucial to firstly identify a modelling toolkit whose application is relevant and suitable for the chosen study areas. Bagstad et al (2013) conducted an extensive assessment of decision-support tools for ecosystem services and highlighted three different models that could produce outputs for multiple ecosystem services simultaneously. These were ARtificial Intelligence for Ecosystem Services (ARIES), Land Utilisation Capability Indicator (LUCI) and Integrated Valuation of Ecosystem Services and Trade-offs (InVEST). To aid the identification of the most suitable model for this research, a set of criteria were produced (*Table 3*), based on the work of Bullock and Ding (2018). This led to the Land Utilisation and Capability Indicator (LUCI) model being adopted for the purpose of this study.

Table 3: *Guiding criteria to assess the potential application of three ecosystem service models (LUCI, ARIES an InVEST) within this research, taken from Bullock and Ding (2018).*

Steps	Guiding Questions				
Step 1: Determine the policy questions and scope of the research	**What policy questions need to be addressed using an ecosystem service model?**	**How can the questions be framed to produce useful model outputs?**	**What ecosystem services are important for the decision-making process and what is their geographic scale?**	**What formats (biophysical, economic, or simple maps) are required from the model outputs?**	**What level of accuracy is needed for this decision?**
	Which areas should be prioritised for alternative management under a re-wetted condition? What are the net gains and losses in different ecosystem services associated with changes in the land management?	Assess the value of the ecosystem services under alternative management techniques to ensure a sustainable method of land management.	Carbon sequestration, flood mitigation, agricultural production, recreational benefits, habitat creation and water quality.	Output data should be in biophysical and economic formats. Map outputs within GIS will enable policy-makers to illustrate how changes can influence different services.	The level of accuracy should be high, given the relatively small scale of the study sites (catchment size) and subsequent sensitivity of the area.
Step 2: Consider the decision-making context	**What is the intended use of the ecosystem service model results?**	**What data about ecosystem services are available to address the questions?**	**Is there the technology capacity to model results?**	**What is the timeframe for the decision?**	
	To provide objective evidence that can be used to inform decision-makers about the potential benefits or losses associated with a change from the current management.	Digital maps for soil classifications and land cover are available, which can be refined using local knowledge to update any changes since the data was collected.	Yes, there is access to GIS software. So, a model which is compatible with ArcGIS is desired.	Moderate, the project will span 12 months but is an ongoing process so outputs can be continually adopted to and utilised.	
Step 3: Evaluate ecosystem service models in the decision-making context	**Which modelling tool can address the questions and objectives?**	**Does the tool include the ecosystem service that are required to model?**	**In what format are the outputs produced?**	**What is the level of accuracy that the tool can achieve?**	**Can the tool deliver results in the decision-making time frame?**
	InVEST, ARIES and LUCI can project ecosystem service values for baseline and future scenarios.	InVEST, ARIES and LUCI all produce model outputs for most of the required ecosystem services. InVEST provides outputs for cultural benefits alongside a wide array of other services.	Biophysical maps outputs are created by each model to quantify the biophysical aspects of each service tool. LUCI classifies outputs into simple categories to illustrate the overall condition of each ecosystem service. Market values can be added to LUCI and InVEST outputs to provide an economic appraisal.	InVEST and ARIES are typically used at a variety of scales ranging between regional and national. LUCI works at a regional, local and field level.	LUCI, InVEST and ARIES have the capacity to deliver results within the designated time frame. Dependent on the size of the application and level of analysis, InVEST and LUCI applications can take a matter of days.
Step 4: Reassess your data resources and modelling capacity	**Is the quality and quantity of data available for the model?**	**Is there the technical ability to run the preferred tools?**	**If needed, is there access to the resources for external expertise?**		
	Nationally-available datasets are available to be processed by LUCI and InVEST, with the potential for detail to be added by local knowledge, site visits and additional datasets. ARIES automatically collects available datasets.	LUCI, ARIES and InVEST has been shown to provide usable outputs in previous studies. Each model comes with a user guide to provide support for a variety of applications	Yes, with regards to LUCI there is the potential to additional resources from the developers in return for feedback on the model.		
Step 5: Choose the most appropriate model	LUCI was favoured over InVEST and ARIES as LUCI was developed specifically for its application within the UK and New Zealand. Although InVEST offers a wider array of ecosystem service models, the degree of that each tool can be adjusted using additional datasets is somewhat limited and could cause further limitations in its application further down the line. Additionally, LUCI possesses the ability to reclassify outputs based on the condition of each service, creating clear outputs that can be utilised by decision-makers to identify specific areas of weakness in environmental sustainability, based on the built-in trade-off tool.				

LUCI, a spatially-explicit toolkit, explores how changes in landscape characteristics influence the spatial manner of ecosystem services and their interdependencies (Trodahl et al., 2017). These services, governed by different types of land management, are compared against values extracted from future projections to identify candidate areas for intervention, protection or maintenance of the *status quo* (Jackson et al., 2017).

Bagstad et al (2013) note that LUCI shares a number of features with other decision support frameworks, whilst being the only tool that is suitable for both landscape and site-scale modelling. Despite this, the application of LUCI within the UK has been largely limited to Wales, through the GLASTIR Monitoring and Evaluation Programme (GMEP) and has yet to be tested in other parts of the UK (Emmett et al., 2017).

For the purpose of this research, LUCI was primarily run as a toolbox extension out of ESRI™ ArcMap 10.4 to create a number of land-use scenarios in both study catchments. During the research stage, associates from Victoria University, Wellington, granted access to the newly developed server-based version of the tool. Working together, simulations were run through the server-based version to aid the continued development of the toolkit and to trial its application within areas of the UK. However, to ensure consistency within the simulated outputs, only those produced from the more reliable desktop version of LUCI were retained for this study.

3.3 Data requirements

One of the benefits of LUCI is that the model operates on a relatively low set of data requirements in comparison to other ecosystem service models, using readily-available national data that can be supplemented with additional data to improve the precision of simulations (Trodahl et al., 2017). Emmett et al (2016) highlight how the predominant drivers in the complex spatial distribution of ecosystem services are elevation, land use/land cover, precipitation and soil classification, all of which have been previously surveyed to varying degrees of resolution at a national level for the UK. Optional inputs, such as watercourse networks and climatic information can also be included, otherwise LUCI generates these based on the existing Digital Elevation Model (DEM) and national averages for rainfall and evapotranspiration. The datasets used to generate LUCI outputs are summarised in *Table 4* (below), including the basic requirements and optional datasets. The resolution of each dataset was chosen based on recommendations from previous applications of LUCI, alongside restrictions surrounding accessibility and processing power.

Table 4: *Data inputs used for this application of LUCI.*

	Data used:	Format/resolution:	Source:
Topography	Digital Elevation Model	5m Raster	Ordnance Survey
Soils	Agricultural Land Classification	Vector polygon	Natural England
	Soilscapes NATMAP	Vector polygon	Cranfield University
Land use	Land Cover Map 2015	25m Raster	Centre for Ecology and Hydrology
	Crop Map of England 2016	Vector polygon	Rural Payments Agency
	Crop Map of England 2017	Vector polygon	Rural Payments Agency
Rivers	Watercourses	Vector shapefile	Ordnance Survey
	Statutory watercourses	Vector shapefile	Environment Agency

3.3.1 Digital Elevation Model (DEM)

The fundamental input for LUCI is a sufficiently high-resolution digital elevation model (DEM), dependent on the size of the study area and level of detail required. This is important because all outputs will be generated at the same resolution as the DEM, with a $5m^2$ resolution being found sufficient for decision-making outputs at a field scale (Jackson et al., 2013). Elevation data was retrieved from the Ordnance Survey at a 5m resolution to represent the structure of the surrounding landscape. The option to adopt $1m^2$ resolution LiDAR data was considered to provide a better assessment of the landscape. However, the inclusion of this was limited by the size of study sites alongside a number of technological constraints, including: the size and number of intermediate files, increase in processing time, and the lack of complete data at such a resolution.

3.3.2 Land Cover

Land cover was taken from the 2015 Land Cover Map (LCM) provided by the Centre for Ecology and Hydrology. The dataset identifies 22 distinct land cover classes on a national scale at a 25m resolution. Through local expert knowledge and site surveys, the LCM was updated to more accurately represent areas of the landscape which had undergone changes since the dataset was initially captured in 2012. This enabled the LCM to represent the present land cover, avoiding inconsistencies where known, significant changes had occurred.

3.3.3 Soil classifications

The Cranfield University and the Land Information Service (LandIS) provided soil classifications through the National Soil Map of England and Wales (NATMAP). An array of soil characteristics are outlined within this, including: a basic description, underlying geology, dominant soils, associated soils, fertility, texture, drainage, land cover and habitats. NATMAP association Soilscapes was adopted in this study to provide a concise and easily-interpreted classification of 30 different soil conditions. This data enables LUCI to calculate expected soil moisture capacity capabilities of different soil and land cover compositions, linking classifications to look-up tables for the provision of different ecosystem services.

3.3.4 Watercourses

Whilst LUCI possesses the ability to calculate and create a stream network based on the DEM, it was preferred to 'burn in' already established watercourse networks to ensure that the hydrological function of the landscape was reliably represented. Watercourse networks were extracted from the Ordnance Survey and merged with the Statutory Main River Map from the Environment Agency database. This enabled a detailed and complete representation for the locality of both main watercourses and drainage channels in the study areas.

3.4 Research strategy

The aim of this research is to create an approach that can be implemented on a wider scale than the two sites included in this study. To do so, it is important to create a clear, concise and reproduceable methodology. As identified below with *Figure 8*, the process for this research is cyclic, with the potential for multiple iterations and scenarios to be processed and quantified, with each step bring broken down into more detail below.

1. Data inputs are provided to create land cover scenarios for both study areas under the current management condition, representing a “baseline” condition.
2. These inputs are processed within LUCI to create biophysical outputs which quantify the amount that each ecosystem service provides.
3. These biophysical outputs are then summarised into clear, concise maps that group values into five key categories for each service, ranging from: very high, high, moderate, poor and very poor. The thresholds for each classification are defined by the user, for site and policy-specific designation. However, LUCI also provides default values for each classification for the thresholds which would by typically classified within the UK.
4. Utilising these groupings, each class was assigned a weighted value that enables a multiple ecosystem service analysis to be performed, identifying areas where these services are being unsustainably utilised. Doing so, highlights candidate areas where alternative management strategies could be implemented to improve the provision and overall status of multiple ecosystem services.

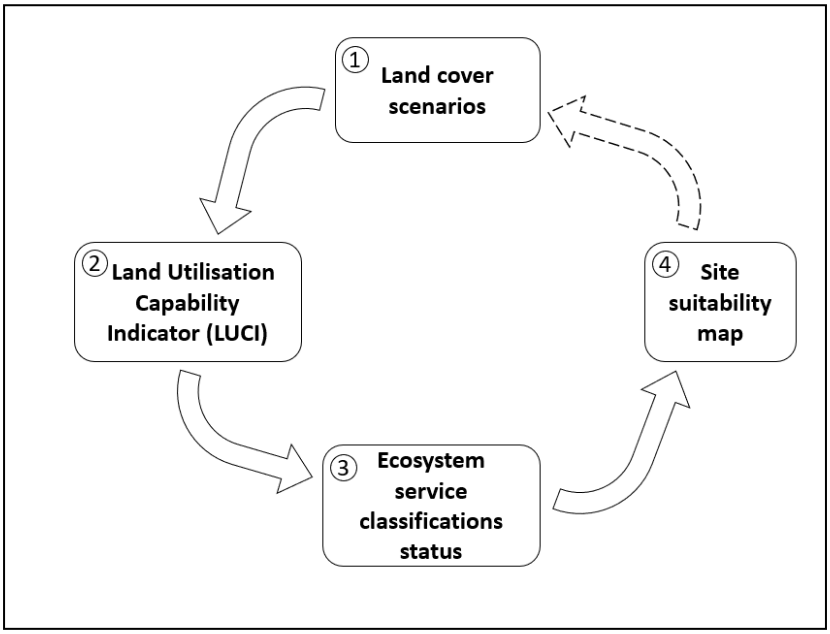


Figure 8: *The cyclic strategy of assessing the status of different ecosystem services within LUCI.*

Ecosystem services chosen for this application of LUCI were based on the context of the two study catchments, as areas of significantly drained wetlands, where the flow of these services are particularly important. Once this has been established, additional information can be included to augment the analysis, such as the extent of re-wetting, to create scenarios that represent the likely future environmental conditions. These future scenarios require data inputs to be updated, specifically the land cover map, and then be re-processed within LUCI to create a set of outputs for future conditions. By comparing the different outputs for a re-wetted environment against the drained baseline scenario, the net gains and losses in ecosystem services are quantified, used to inform decision-makers about the consequences of a change in land management.

Integrating the requirements of stakeholders, land managers and policy-makers helps identify the most suitable areas for LULC change, their spatial relevance in relation to re-wetting, the economic feasibility of decisions and the expected benefits will ensure the potential of each scenario has been fully explored.

3.4.1 Ecosystem service pre-processing

The first step of LUCI is the generation of a hydrologically and topographically consistent DEM, which was used to calculate the hydrological function and routing of water within the landscape. LUCI draws upon the DEM and the mean rainfall data to create a stream network if data are unavailable. However, with the input of a detailed watercourse network, the position of watercourse channels can be 'burnt in' to aid the hydrological routing process. Once this is completed, a land use scenario was created by combining the DEM with information on the soil type and land cover (Jackson et al., 2013). This step also presents the opportunity to update national datasets, where inaccuracies or errors with sampling can be rectified to better represent current conditions. These steps create intermediate files within LUCI that enable the calculation of individual ecosystem services within the study area (*Figure 9*).

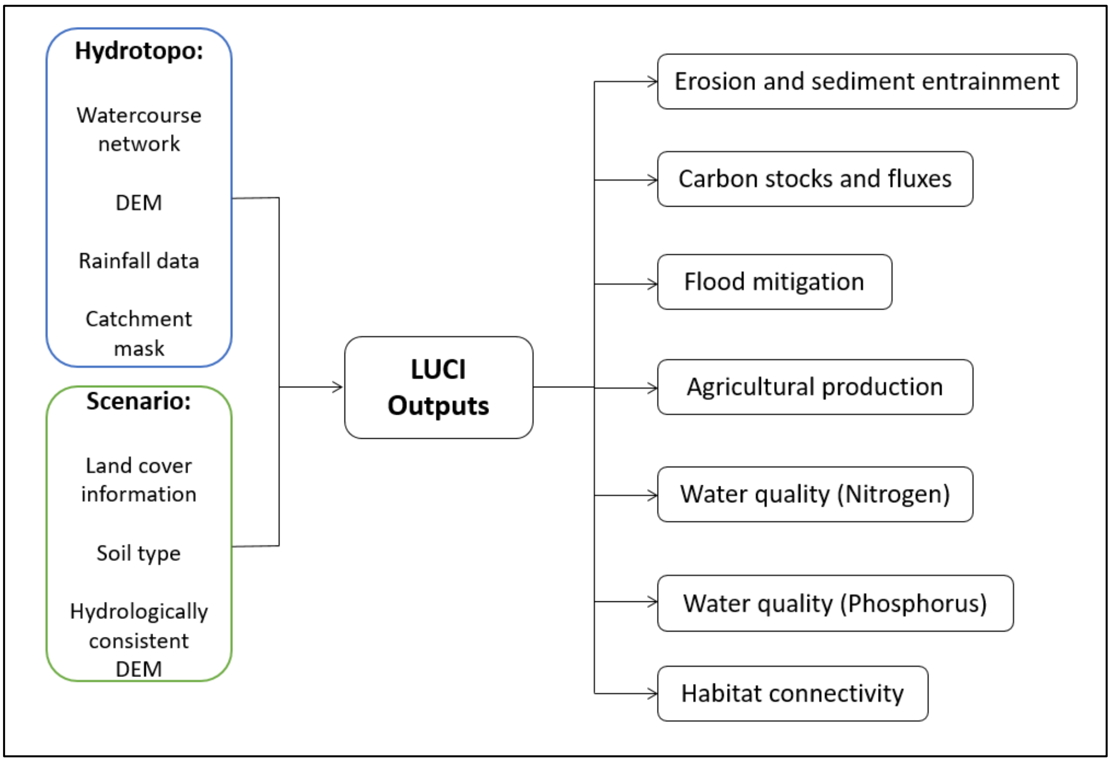


Figure 9: *Flow diagram of the LUCI pre-processing steps, data requirements and model outputs.*

3.5 LUCI outputs

Once the pre-processing steps are completed within LUCI, the tool processes ecosystem services either individually or simultaneously (dependent on the user's requirements) to produce usable outputs for each study area (*Figure 10*). LUCI generates map outputs that summarise the quantitative outputs using a "traffic light approach", categorising service provisions into five different classes from 'very high' to 'very low'. This enables simple and easy analysis when working with stakeholders and decision-makers. Areas which are advised to undergo intervention or cease the current management strategy, due to an existing poor service provision, are highlighted in red. Orange areas highlight where there are opportunities to improve the condition of a service. Areas which currently exhibit a good level of ecosystem service provision and should be preserved, or continue their current management, are highlighted in yellow and green.

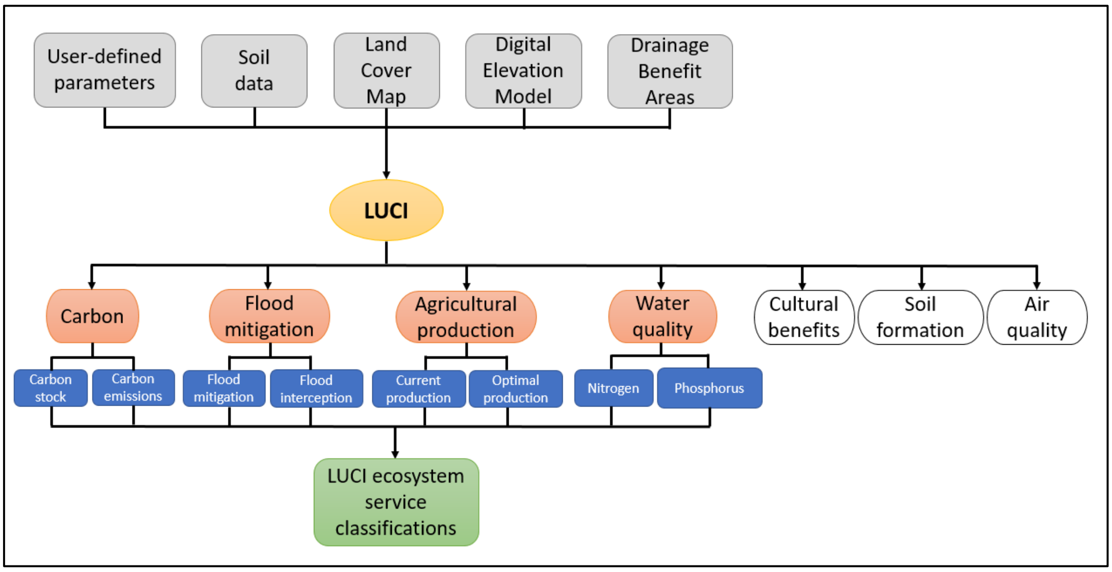


Figure 10: *LUCI data requirements, ecosystem service tools, outputs and relevant services not currently modelled (hollow).*

3.5.1 Carbon stock and fluxes

LUCI calculates carbon stocks and fluxes at steady state values, assuming that the land cover is established within an area and that carbon locked within the soil and biomass is not in a state of flux, i.e. LUCI assumes no initial lag in productivity associated with a change in land management. Land cover scenarios for soil type and land cover are linked to 'look-up' tables which are based on IPCC tier 1 protocols. The model applies values for carbon stored in biomass and soil based on this and calculates the potential for carbon emission or sequestration in the area. Using this as a working basis, LUCI produces a number of outputs which examine different aspects of carbon, including: total carbon store, carbon stock at 30cm and 1m soil depth, carbon emissions, and the overall carbon status. More specifically, the 'carbon status' output considers all aspects of these aspects, highlighting where to improve rates of sequestration and storage, whilst ensuring minimal carbon emissions.

3.5.2 Agricultural Productivity

The agricultural productivity tool within LUCI utilises a number of different inputs and site-specific parameters. Optional variables, notably the fertility and drainage conditions of the landscape, alongside information derived from soil characteristics and the DEM, enable LUCI to represent growing conditions suitable for agriculture within the landscape. These conditions, combined with the current land cover, create an array of productivity classes that reflect the capacity for agriculture within the landscape. User-defined thresholds were set to reflect the optimal level of production based on the topographic influences, including slope and elevation. However, given the low-lying flat nature of both catchments, these were set to default values for consistency purposes.

The levels of agricultural productivity were based on a ranking system for the typical level of production for each of the classes. Based on this, LUCI generates four different outputs for agricultural production. Current production illustrates the expected level of productivity in an area based on the conditions at present. The 'optimal production' tool ignores the known land use and simulates the level of production that LUCI deems optimal for agriculture. Relative production status signifies the difference between the current and optimal production status, highlighting where areas undergo higher or lower levels of production than expected. Following this, the overall agricultural productivity output combines outputs from all three prior outputs to create a map that signifies where land is being over-utilised or under-utilised.

3.5.3 Water quality and nutrient loading

Water quality models within LUCI calculate the total amount of phosphorus and nitrogen that is supplied to watercourses. Nutrients leaching into watercourses are quantified as kg ha^{-1} yr^{-1}, to illustrate the mass of the nutrient contributing towards a poor water quality annually, per unit area (Jackson et al., 2017). LUCI utilises the high-resolution DEM in combination with the average rainfall of the study area to create a map representing the topographic routing of the landscape. A number of outputs are generated by LUCI, including in-stream water quality as well as the overall loading of both nitrogen and phosphorus, indicative of the influence of the known the land cover classifications. Export coefficients were based on land cover alone and do not account for point sources of N or P, such as sewage drains. Therefore, the water quality tool examines the agricultural contribution and supply of excess nutrients, rather than other anthropogenic influences. In this application of LUCI, only nutrient loading will be explored here to represent the direct influence that land cover decisions have upon the loading of nitrogen and phosphorus.

3.5.4 Flood mitigation

Flood mitigation models in LUCI predict where water will accumulate through overland and near surface flow as a product of the topography and land cover of the surrounding areas. This tool informs decision-makers about the potential for water to accumulate in the landscape following a large precipitation event, based on the hydrological routing and presence of mitigating features. By taking information about the storage of different soil categories alongside land cover characteristics, LUCI produces qualitative outputs to illustrate areas at risk of surface flooding. The model also examines the land cover type to determine the presence of flood mitigating features in the landscape, or the relative lack of mitigating features, where intervention should be implemented.

3.5.5 Win-win scenarios and ecosystem service trade-offs

The LUCI model provides the ability to create additional trade-off maps which identify opportunities to improve the overall status of multiple ecosystem services, rather than areas which offer high provision for a singular service (Jackson et al., 2017). To do so, the model identifies areas where the delivery of ecosystem services could be improved, focusing on areas that currently offer singular benefits or where the overall condition of services is poor. When each tool is run, a number of biophysical outputs are created, and are classified into five different bands ranging between very high to very poor. When analysing the potential for 'win-win' scenarios and 'trade-offs' between different services, each service was weighted based on their classification and combined to identify where the provision of multiple ES through 'co-benefits' could be achieved through a revised land cover. For this application six different ecosystem services were analysed – carbon storage, carbon sequestration, agricultural production, nitrogen, phosphorus and flood mitigation. Each service was weighted equally, and cumulative totals were then classified into five different categories to enable the identification of areas which would benefit from alternative management.

3.6 Modelling changes in the water regime

LUCI is a decision support tool and is therefore only part of the answer in establishing solutions to challenges faced by unsustainable land management. Therefore, other approaches are required to cover all aspects of the study. One of the focal points of this research is the potential change in the water management regime and how the land is drained. To achieve this, the findings of Jacobs, JBA and ARUP in their "Lower Alt with Crossens Pumped Drainage Catchment Flood Risk Management Strategic Plan" were utilised (Environment Agency, 2010). Drainage Benefit Areas (DBAs) were extracted from the Water Management Units (WMUs) to illustrate the spatial extent of the areas which benefit from land drainage, due to presence of drainage channels, the relative elevation and the locality of pumping stations (*Figure 11*).

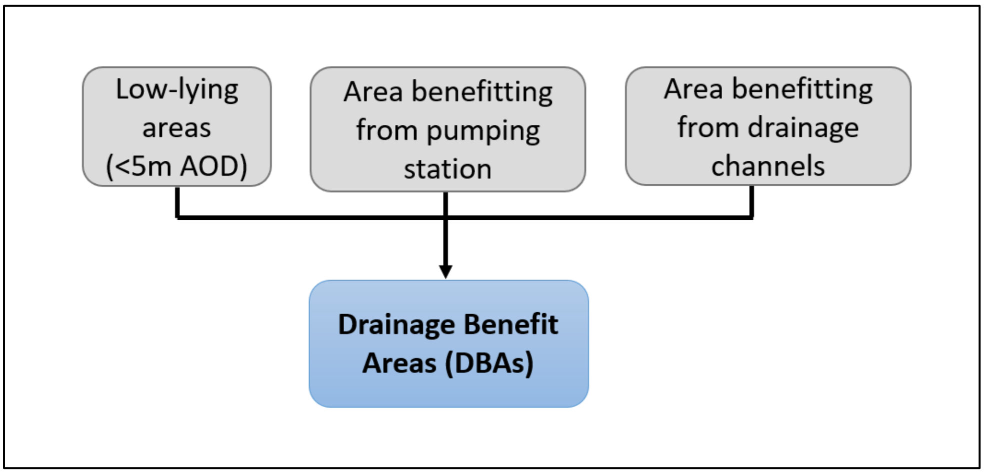


Figure 11: *Basic criteria in determining drainage benefit areas.*

The boundaries of DBAs were defined by the presence of drainage channels, the locality of pumping stations and the extent of low-lying land, typically within 5 metres of the sea level (*Figures 12 and 13*). Given this, a 5m^2 resolution DEM was utilised alongside a 1:25,000 base map of the catchments to parameterise the benefit areas. These were then cross-referenced against the known co-ordinates of the different pumping stations within the Alt Crossens and Lyth Valley to ensure each DBA contained a station or high density of drainage channels. The original Water Management Units, as defined in the JBA report, utilised a 1:250,000 base map, which arguably lacked accuracy when identifying the influence of drains at a field level. Therefore, a recently-published base map with a much higher resolution was utilised to counteract this and provide an enhanced level of precision when identifying the areas which benefit from drainage were within each DBA.

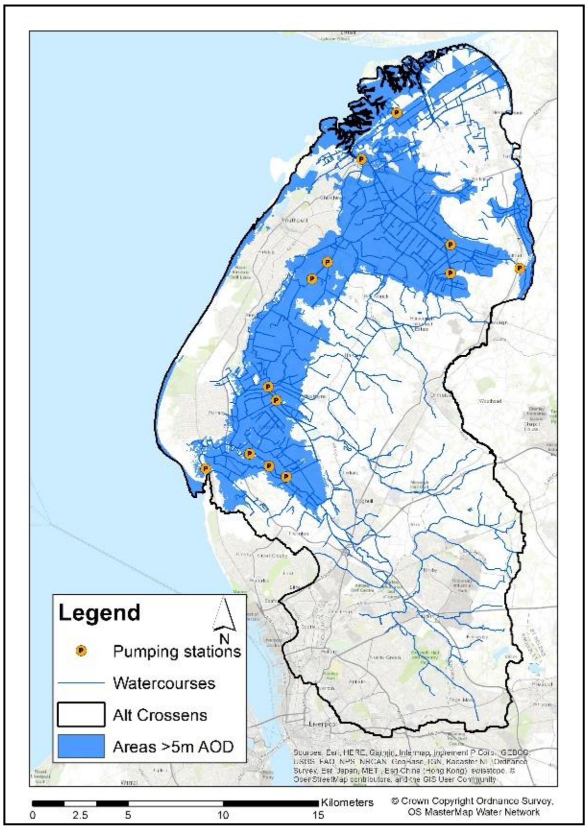


Figure 12: *The location of each of the 13 pumping stations within the Alt Crossens (left), with reference to the presence of low-lying land and watercourses.*

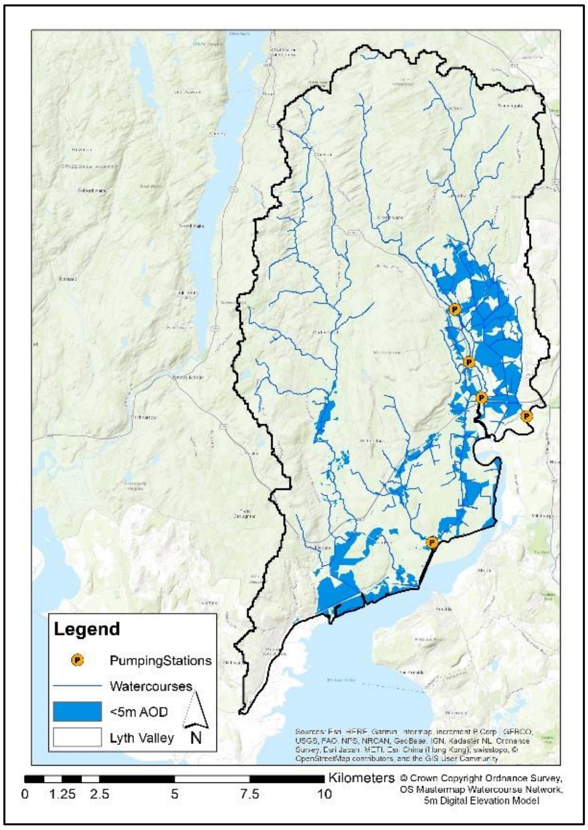


Figure 13: *The location of each of the 5 pumping stations within Lyth Valley (right), with reference to the presence of low-lying land and watercourses.*

For the purpose of satisfying all areas which benefit from the land drainage in the Alt Crossens, the study area was extended east of the Banks Marsh station to account for Water Management Units which crossed the boundaries into the neighbouring Douglas catchment. Given the large spatial extent, a scenario-based approach was adopted to define the likely change in the land drainage regime. This involved creating four separate scenarios ranging in the level and degree of future drainage (*Figure 14*). Each of these was created based on the relative priority of each pumping station, due to the stations being a key influence on the cost associated with water management.

- **Scenario 1:** "Business as usual". All 13 pumping stations and drainage channels remain operational and managed. This assumes that an overseeing body, other than the Environment Agency takes responsibility for the cost of the land drainage.
- **Scenario 2:** "Partial closure". Six of the 'low-priority' pumping stations, which have a high maintenance cost and low drainage capacity, are closed. These include: Kew, Banks Marsh, Ince Blundell, Boundary, Rufford and Clay Brow.
- **Scenario 3:** "Extended closure". 11 out of the possible 13 pumping stations are closed. Both Crossens and Altmouth remain operational due to the EA's legal responsibility to provide coastal protection to urban areas.
- **Scenario 4:** "Full closure". 11 out of the possible 13 pumping stations are closed. Both Crossens and Altmouth remain operational due to the EA's legal responsibility to provide coastal protection to urban areas. This scenario also includes a reduced level of watercourse management in the Back Drain and Sandy Brook areas.

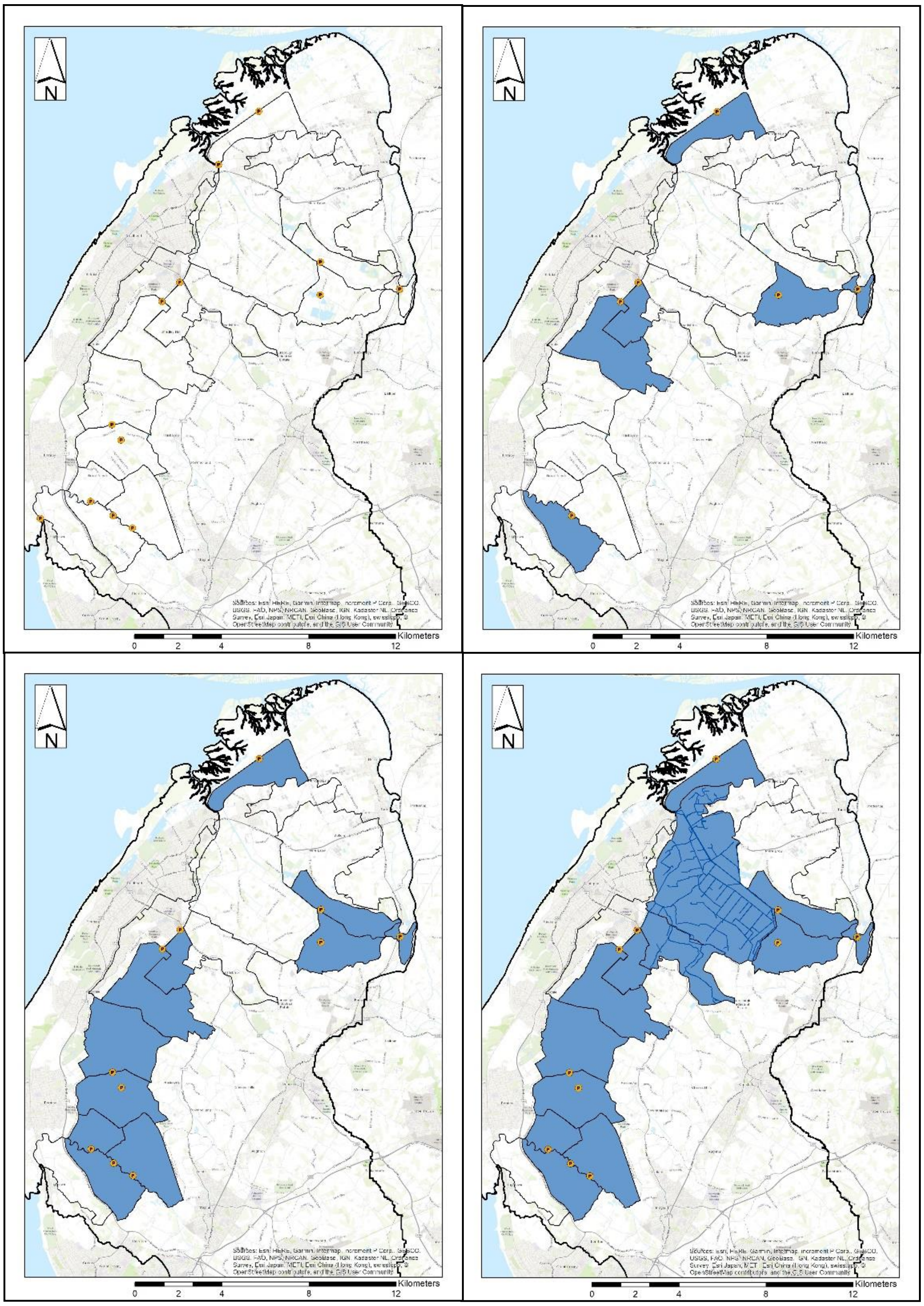


Figure 14: *The different extents of re-wetting in the Alt Crossens, based on a scenario-based approach. Scenario 1 = top left, Scenario 2 = top right, Scenario 3 = bottom left, Scenario 4 = bottom right.*

In the case of Lyth Valley, legal requirements are in place to keep the Levens Catchwater station permanently operational, due to the benefit to urban areas, whereas Ulpha station is contracted to operate until 2023. As there are only four pumping stations which may potentially close in the near-future, there was the reduced opportunity for a scenario-based approach in this application. Instead, the likely option was to execute a full closure on stations with no legal obligation. DBAs for Lyth Valley are shown in *Figure 15*, with the areas likely to experience re-wetting highlighted under a full future closure of non-essential pumping stations.

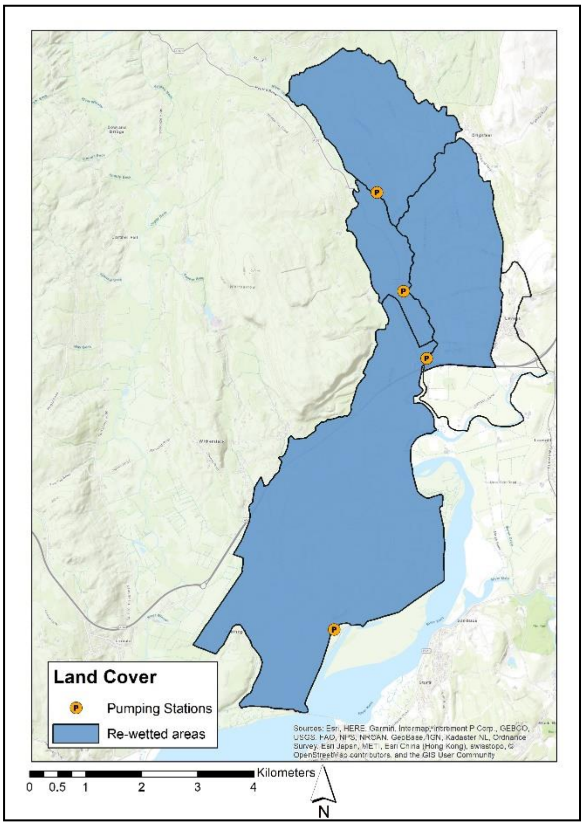


Figure 15: *The expected extent of re-wetting in Lyth Valley, based on the closure of 4 pumping stations.*

3.7 Attributing an economic value to land cover

During the analysis it became apparent that one of the limitations of the agricultural productivity toolkit was that there was no method of quantifying the production beyond the five outputs classes generated by LUCI. As this study specifically examines land cover and how future changes can influence ecosystem services, with one of the major services being agricultural productivity, there was a clear requirement to provide an improved level of quantification for this service. Therefore, the 2015 Land Cover Map was augmented with additional datasets to provide more classifications within the "arable and horticulture" land cover class. Two alternative datasets were examined; the Crop Map of England (CROME) and the Land Cover Plus dataset. The Land Cover Plus dataset was subsequently excluded due to the lack of information surrounding additional land cover types other than farming.

Subsequently, the 2016 and 2017 CROME surveys were utilised; by summarising the spatial extent of each crop within the extent of the land cover map field boundaries. The dominant crop per field was used for the final assessment.

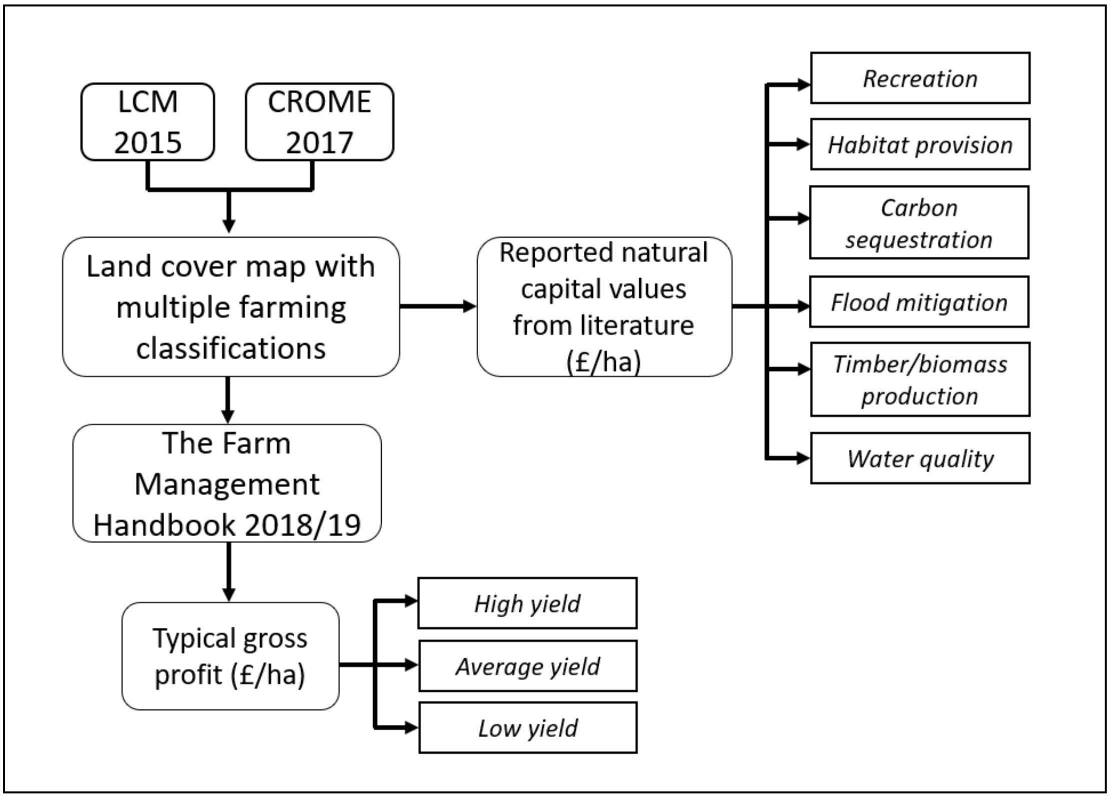


Figure 16: *Strategic flow diagram illustrating how natural capital can be valued based on academic literature and market values.*

Once reclassified, the updated LCM dataset was then subject to an economic appraisal. Using the latest edition of "The Farm Management Handbook 2018/19", it was possible to attach gross market values to each crop defined through the CROME survey (*Table 5*). Three different yield values were provided, relative to the degree of management and farmer effort for each form of farming (*Figure 16*). Given this, values from the high yield data were extracted to reflect the highly-fertile Grade 1 arable land within the Alt Crossens. Whereas, the largely Grade 3 agricultural land in Lyth Valley was assigned values based on average value yields.

These were extracted in £ per hectare and assigned to each classification to create a value of each field, based on the calculated area.

Table 5: *The gross margins for different types of crops and agriculture, based on Craig (2018).*

Crop type	Gross economic returns (£/ha)
Mixed vegetation/unknown	1,000
Spring barley	996
Beet	1,011
Carrot	4,070
Maize	824
Spring oats	1,397
Spring wheat	1,318
Spring oilseed	781
Potato	4,448
Winter barley	1,161
Winter wheat	1,440
Winter linseed	570
Winter oats	1,325
Winter oilseed	1,259
Spring field beans	938
Green beans	443
Spring peas	837
Winter field beans	619
Grass	371

These principles were also adopted for the other land cover classifications, outside agriculture, to create a comparative study between the economic value of agriculture and the benefits provided by other ecosystem services (*Figure 16*). Based on the ecosystem services available for modelling within LUCI, an analysis was made into the value of natural capital associated with alternative land management practices and a cumulative value for each classification assigned (*Table 6*).

Table 6: The economic value of natural capital and associated benefits from different land covers.

Land cover type	Value (£/ha)	Source:
Peatland	£411	Remme et al., 2010
Pastureland	£371	Remme et al., 2010
Heathland	£383	Remme et al., 2010
Freshwater	£428	Connors and Phillips, 2017
Farmland	£1,000	Connors and Phillips, 2017
Woodland	£738	Connors and Phillips, 2017

Using these values, the total economic return of land was calculated to help quantify the net gains or losses in agriculture under a future management scenario. By comparing both the economic value and LUCI-generated biophysical outputs, it was possible to compare the baseline scenario and alternative land management scenarios to quantify the potential for change in economic terms for each study area.

3.8 Future management strategy

Given the *status quo* in how land is valued, maintaining agricultural productivity is key, however additional income from environmental sustainability is crucial to attach a feasible income to agriculture on marginal land. With the potential departure of the UK from the EU, the loss of agricultural subsidies through the Common Agriculture Policy (CAP) could significantly influence how land managers continue to operate. With this relative degree of uncertainty, it is crucial to satisfy all areas to ensure that a multi-faceted approach to the land use is adopted, to be able to adapt within a potential change in how the landscape is valued.

A variety of different management strategies were drawn upon based on the degree and nature of the ecosystem services present within the landscape, as identified by the 'Multiple Ecosystem Service Assessment'. However, a set of guidelines were required to ensure that this research could be applied on a larger scale across the UK.

- A clear overall movement away from a management system that benefits a single ecosystem service, instead focusing on multiple services for an improved stability.
- No new areas of woodland on existing layers of peatland, with a preference of felling existing stands on areas of deep peat.
- Any areas within 3m of relative sea level (AOD) within Drainage Benefit Areas should move away from conventional flood-sensitive agriculture.
- Any areas flagged to exhibit a poor economic productivity should aim to pursue improved income through alternative management practices, such as paludiculture.
- An effort to increase the presence and extent of habitat corridors and buffer zones around sensitive areas, such as conventional farming or raised bogs.

4 Results and analysis

This section of the report will analyse the findings of the modelling framework chosen for this project; both the Alt Crossens and Lyth Valley study areas were examined to assess the net gains or losses in ecosystem services between the present baseline and future re-wetted scenarios. Each study site was analysed independently, with the same values and thresholds adopted to illustrate how this approach can be implemented on a wider scale.

4.1 Ecosystem services - Alt Crossens

Four different scenarios were created for the Alt Crossens, reflective of a transition away from intensive land drainage and the potential introduction of alternative land management practices. To reflect a change in the water management regime and subsequent water table, four different scenarios were produced. Additionally, the analysis for Scenario 1 (baseline) included an additional parameter which ignores the influence of waterlogging conditions upon each ecosystem service, whereas Scenarios 2-4 assumed a more naturalised regime which included the influence of waterlogged conditions in the areas designated for re-wetting. *Table 7* and *Figure 17* represent the change in land cover classifications through Scenarios 1-4, with the focal point being a large-scale movement away from land use for arable and horticultural practices. Instead, these areas were replaced by alternative, non-intensive land uses, reflective of paludiculture and the sustainable provision of additional ecosystem services. Between the baseline scenario (Scenario 1) and the future re-wetted scenarios (Scenarios 2-4), a 6,434 hectare reduction in arable and horticultural land use was observed, with neutral grasslands for paludiculture (3,649 ha), broadleaf woodland (837 ha), improved grassland (837 ha) and restored peat bogs (771 ha) accounting for this change.

Table 7: Spatial extent of different land covers for Scenarios 1-4.

Land Cover Map Broad Habitat	Scenario 1 area (ha)	Scenario 2 area (ha)	Scenario 3 area (ha)	Scenario 4 area (ha)
Arable and horticulture	10,083	8,157	6,002	3,649
Bog	0	0	185	771
Broadleaf woodland	488	1,097	1,548	1,935
Coniferous woodland	7	7	7	5
Fen, marsh and swamp	7	7	7	7
Freshwater	56	56	56	56
Improved grassland	931	1,384	1,596	1,768
Neutral grassland	365	1,228	2,534	3,744
Saltmarsh	9	9	9	9
Suburban	922	922	922	922
Supralittoral sediment	2	2	2	2
Urban	149	149	149	149

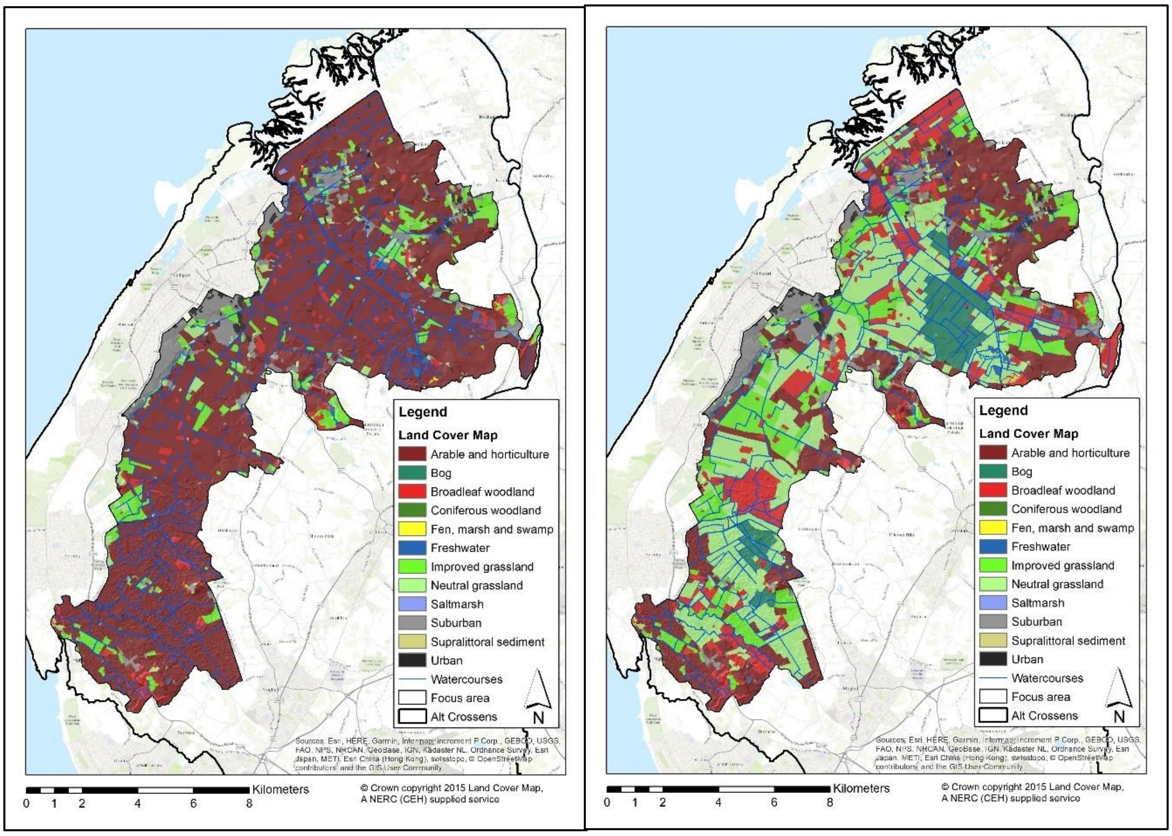


Figure 17: Spatial extent of different land covers for Scenario 1 (baseline) and Scenario 4 (full closure).

4.1.1 Agricultural production

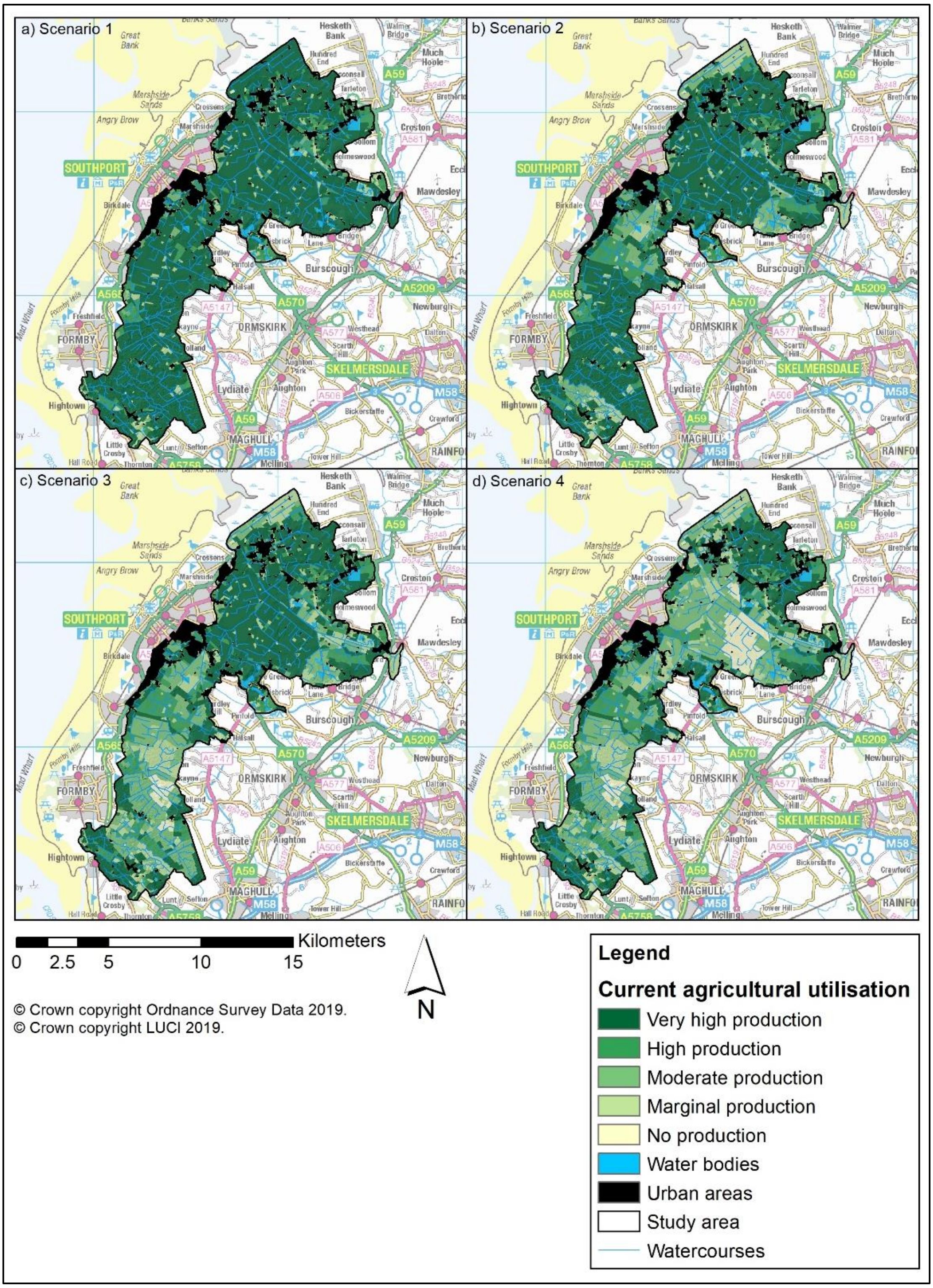


Figure 18: *Estimated levels of agricultural production under varying levels of land management.*

The current agricultural utilisation for the Alt Crossens (*Figure 18*) was calculated within the LUCI toolkit to represent current levels of production, ranking the provided land cover data based on the estimated agricultural productivity. Scenarios 1-4 illustrate the change in the spatial extent and degree of agricultural productivity, reflective of the changes to the land use, based on differing drainage regimes. From this, it was established that as the scenarios move away from a heavily-drained landscape dominated by conventional agriculture, towards future scenarios with minimised drainage and alternative management, there is a clear decrease in the levels of agricultural productivity. This is particularly apparent when quantifying the spatial extent of land with a ‘very high productivity’, falling from 76.5% in Scenario 1 to 27.4% in Scenario 4. Similarly, there was a clear growth in the coverage of land with a ‘moderate’ and ‘marginal’ productivity class, increasing by 25.5% and 10.9%.

These changes were attributed to the large-scale transition away from a landscape heavily-dependent on the operation of drainage channels and pumping stations, towards future landscapes which function under wetter conditions and possess the ability to continue agricultural production through a different means. To add perspective on this seemingly large decrease in production, another simulation was run using the ‘Optimum Agricultural Production’ tool in LUCI, predicting ideal levels of agricultural utilisation, based on the environmental conditions of the landscape. These are identified below in *Figure 19*, representing the optimum degree and location of agricultural production in the study area under current (drained) and future (re-wetted) conditions.

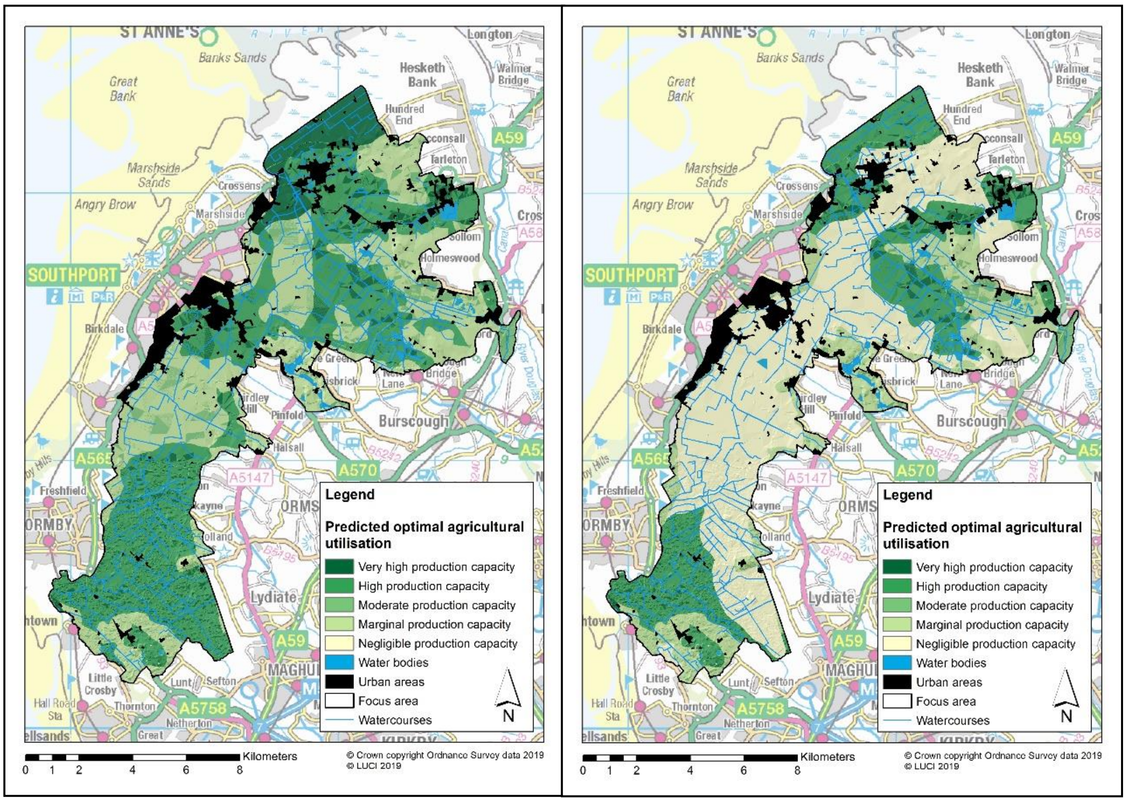


Figure 19: *Predicted optimal rates of agricultural utilisation under drained (left) and re-wetted (right) conditions.*

Using this tool, LUCI estimates that an optimum scenario would include 12.8% of land with a very high production and 42.6% with a high production under drained conditions. Conversely, for a re-wetted scenario, these values decrease to 3.9% and 23.4%. From this, it can be established that under re-wetted conditions, there is a significantly lower capacity for conventional agriculture to persist in the landscape, i.e. if the drainage channels and pumping stations were to stop functioning, large amounts of land would be marginally productive. This signifies the importance of Scenarios 2-4; if current drainage practices were abandoned and the land use kept the same, there would be significant losses in agricultural productivity. Whereas, Scenarios 2-4 represent the opportunity for agriculture to persist in the Alt Crossens under wetter conditions.

4.1.2 Economic value of the land

One of the notable changes in the landscape was the economic value of the land as the scenarios moved away from conventional agriculture towards alternative land uses. An assessment for the economic value of the land was required to augment the analysis, calculated utilising the results from the CROME analysis and estimate land cover values. Based on these results (*Figure 20* below), it is possible to begin assessing the scale of these economic gains and losses under alternative management strategies.

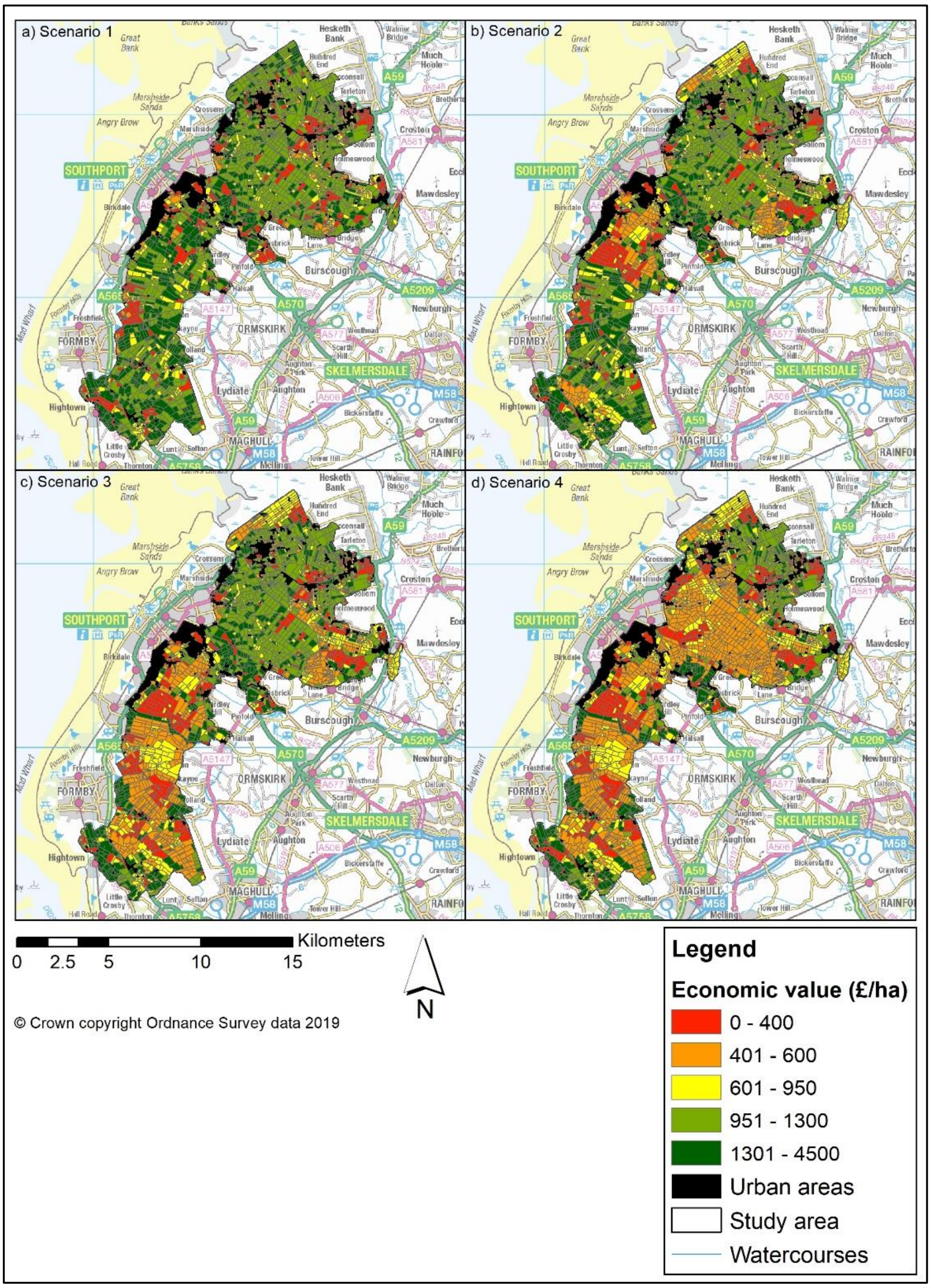


Figure 20: *Calculated economic value of land, based on the present and future (re-wetted) land cover conditions.*

The calculated economic gain of the land under the baseline scenario was £14.7 million per year, with the mean economic value of the land being £827 ha^{-1} $year^{-1}$. Under Scenario 2 this decreased to £13.4 million a year, a £1.3 million shortfall, whilst simultaneously providing a lower mean land value of £784 ha^{-1} $year^{-1}$. There was a further decrease to £11.4 million per year in Scenario 3, a £3.3 million shortfall with a mean land value of £706 ha^{-1} $year^{-1}$. However, the lowest economic returns were identified within Scenario 4, reducing to £9.7 million per year with a mean economic value of the landscape being £617 ha^{-1} $year^{-1}$, accounting for an economic loss of £5 million per year relative to current practices.

4.1.3 Carbon stock

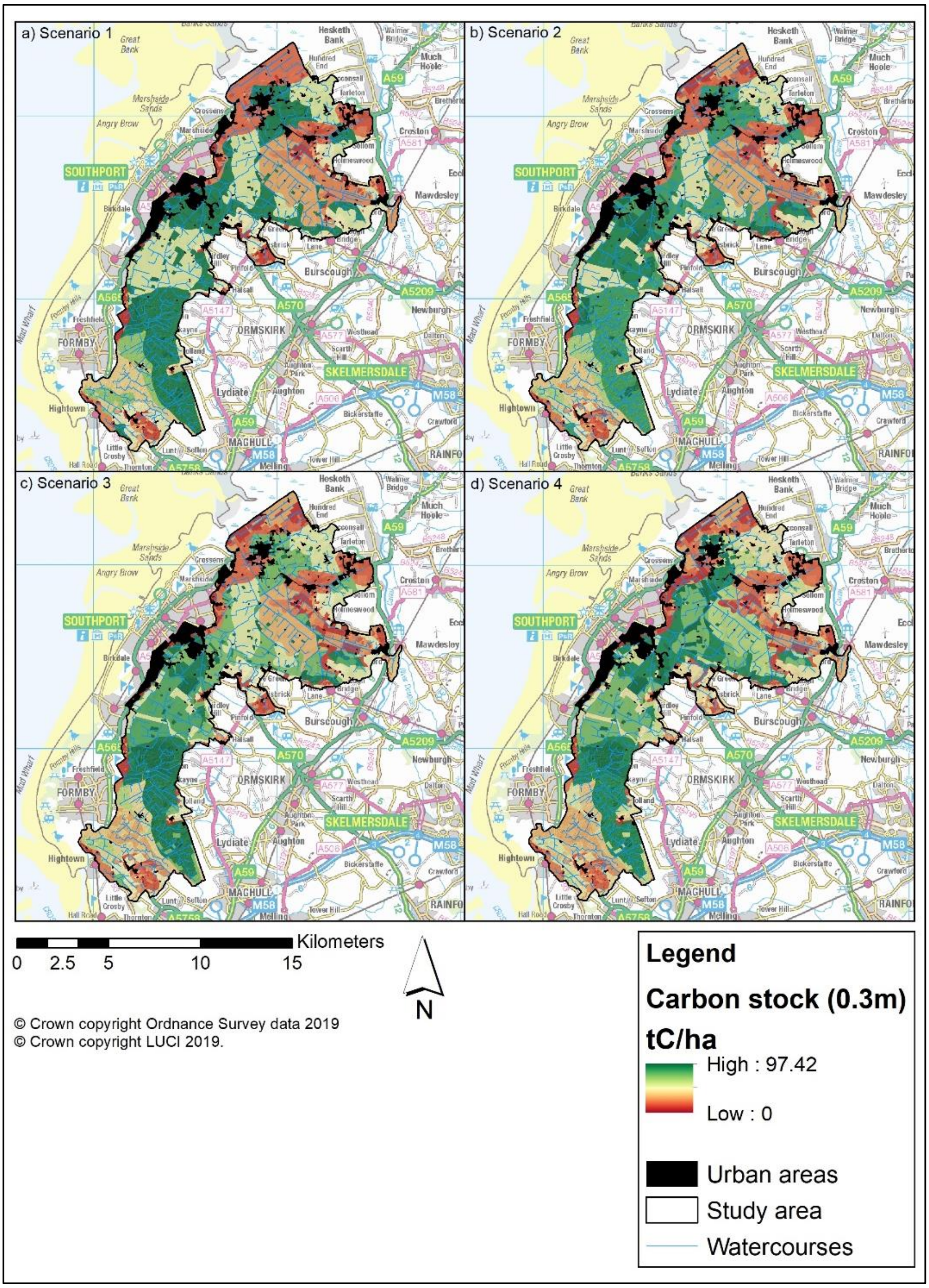


Figure 21: *Calculated carbon stock of above and belowground biomass at a soil depth of 0.3m.*

The carbon stock of both the aboveground and belowground biomass, at a 0.3m soil depth, was calculated for the baseline and each of the re-wetted scenarios. From *Figure 21* (above), there is a clear improvement in the ability to store additional carbon within the landscape, observed by the conversion of land with a low carbon storage to that with a higher capacity. These areas with a low carbon storage potential were associated with sites where land drainage remained operational and where the influence of flooding was minimal, enabling conventional agriculture to persist.

A mean carbon stock of 21.1 tC ha^{-1}, equating to 274,900 tonnes of carbon storage, was found for the baseline scenario. In Scenario 2, the mean storage increased to 25.1 tC ha^{-1}, accounting for 326,700 tonnes of carbon storage. This increased further under Scenario 3, with a mean carbon stock of 30.9 tC ha^{-1} equating to 402,616 tonnes of carbon. Scenario 4 demonstrated the largest increase in carbon storage, with an increase in the mean storage to 39.1 tC ha^{-1} and the potential to store 509,225 tonnes of carbon in the landscape, almost double that of the baseline scenario.

4.1.4 Classified carbon emissions

LUCI generates an output indicative of the potential for the landscape to sequester additional carbon based on soil and land cover combinations, linked to IPCC tier 1 protocols (Jackson et al., 2017). This potential was calculated based on the maximum soil carbon store, associated biomass carbon and current levels of carbon storage. A "space for time" substitution was utilised to calculate the potential for additional carbon sequestration in soils, based on data from sites with established land cover and soil combinations. Using this, the potential for additional carbon sequestration were classified, as seen in *Figure 22* (below).

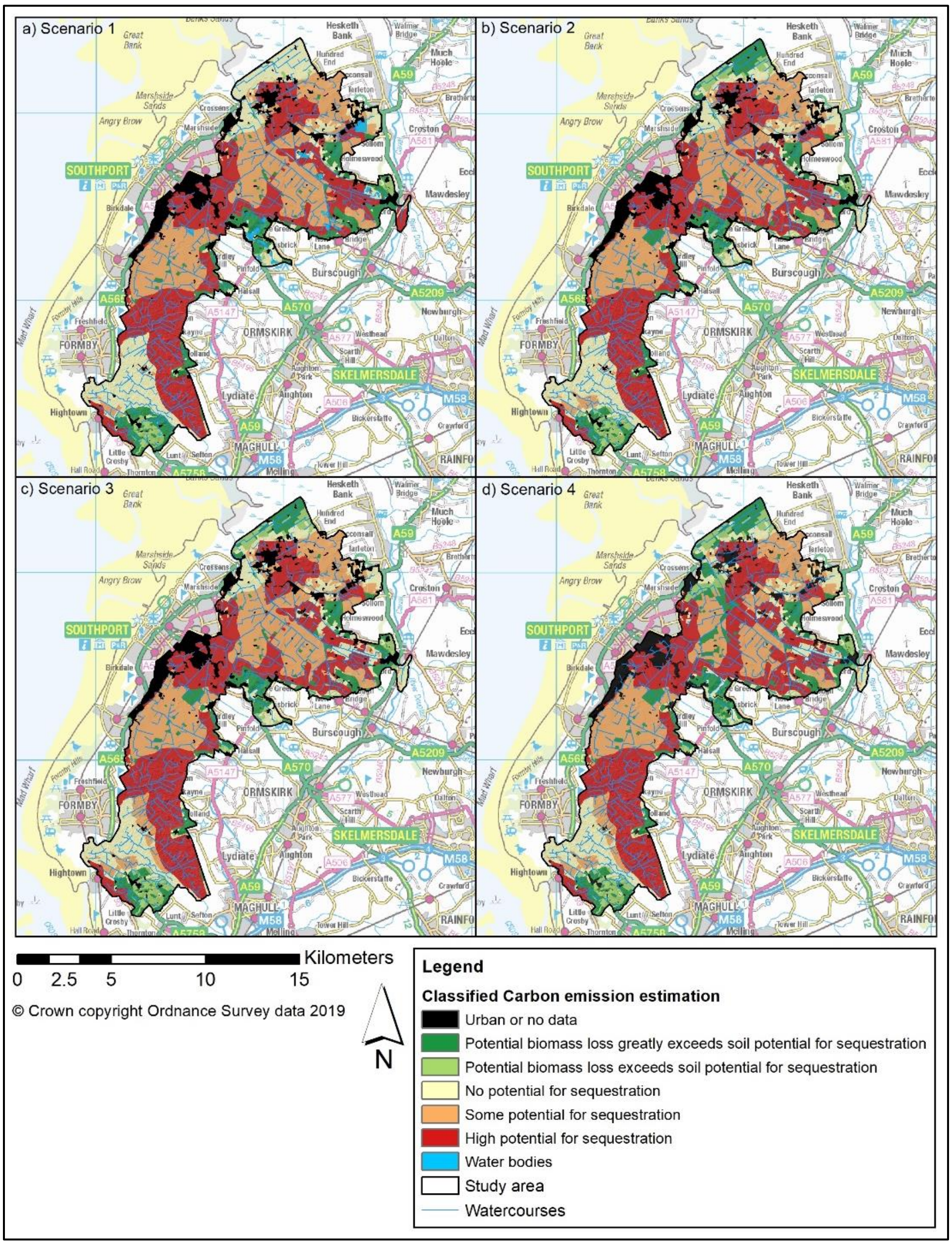


Figure 22: *Classified rates of carbon emission and sequestration, based on four different scenarios.*

Figure 22 illustrates a slight increase in the presence of land between Scenarios 1-4 whereby a change in land cover could lead to biomass losses 'greatly' exceeding the potential for additional soil sequestration, rising from 9.7% to 12.2% of the total area (13,017 ha). This was also observed for land with a where biomass losses 'slightly' exceed soil sequestration, increasing from 2.2% to 5.2%. A small decline was observed for land with no potential for additional sequestration (2.5%), alongside a 2.0% decrease in land with some additional potential for CO_2 sequestration, contributing towards a 1.7% increase in land with a high potential for additional CO_2 sequestration (*Figure 23*).

Within this, it can be observed that despite the minor shift in the landscape being shown to have a higher potential to store carbon in soils under a different land cover between the baseline and future scenarios, the reduction and change in biomass could offset any immediate benefits to sequestration and storage if these areas were to experience future land modification. Based on these findings, there was limited evidence for increased carbon sequestration linked to higher levels of re-wetting, despite the relatively positive findings for carbon storage (Section 4.1.3). Reasoning for which may be due to the way in which LUCI represents the potential for carbon sequestration/emissions, with much smaller margins in thresholds for each classification, compared to other means of representing carbon fluxes which have a much larger range in values (tC ha^{-1} $year^{-1}$).

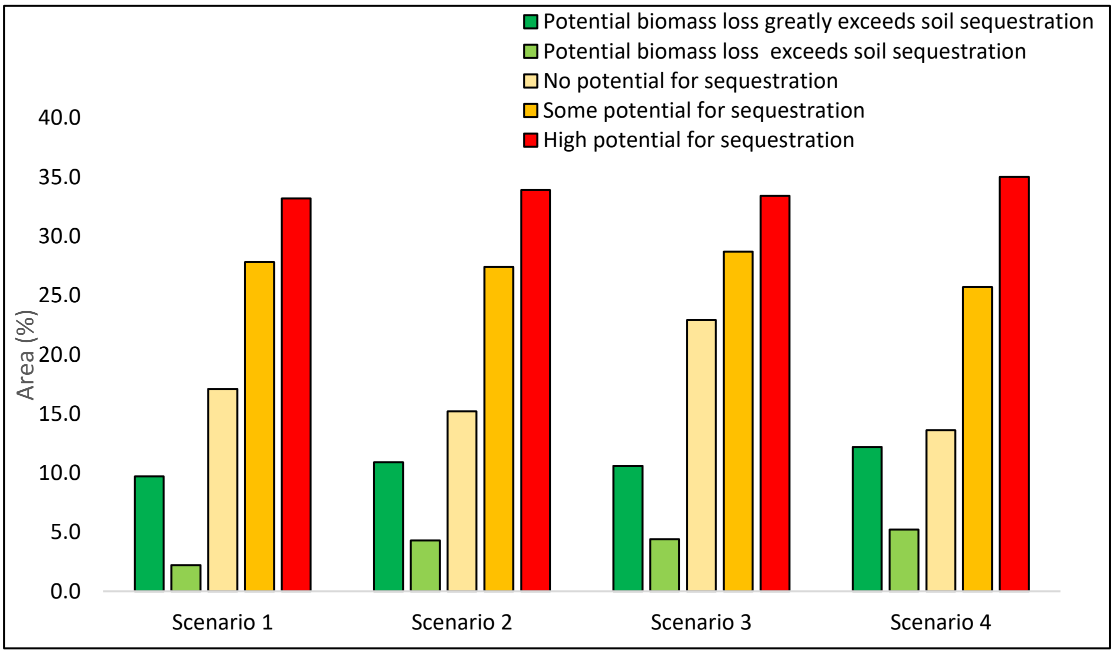


Figure 23: *Classified carbon sequestration potential in the Alt Crossens for Scenarios 1-4.*

4.1.5 Flood mitigation

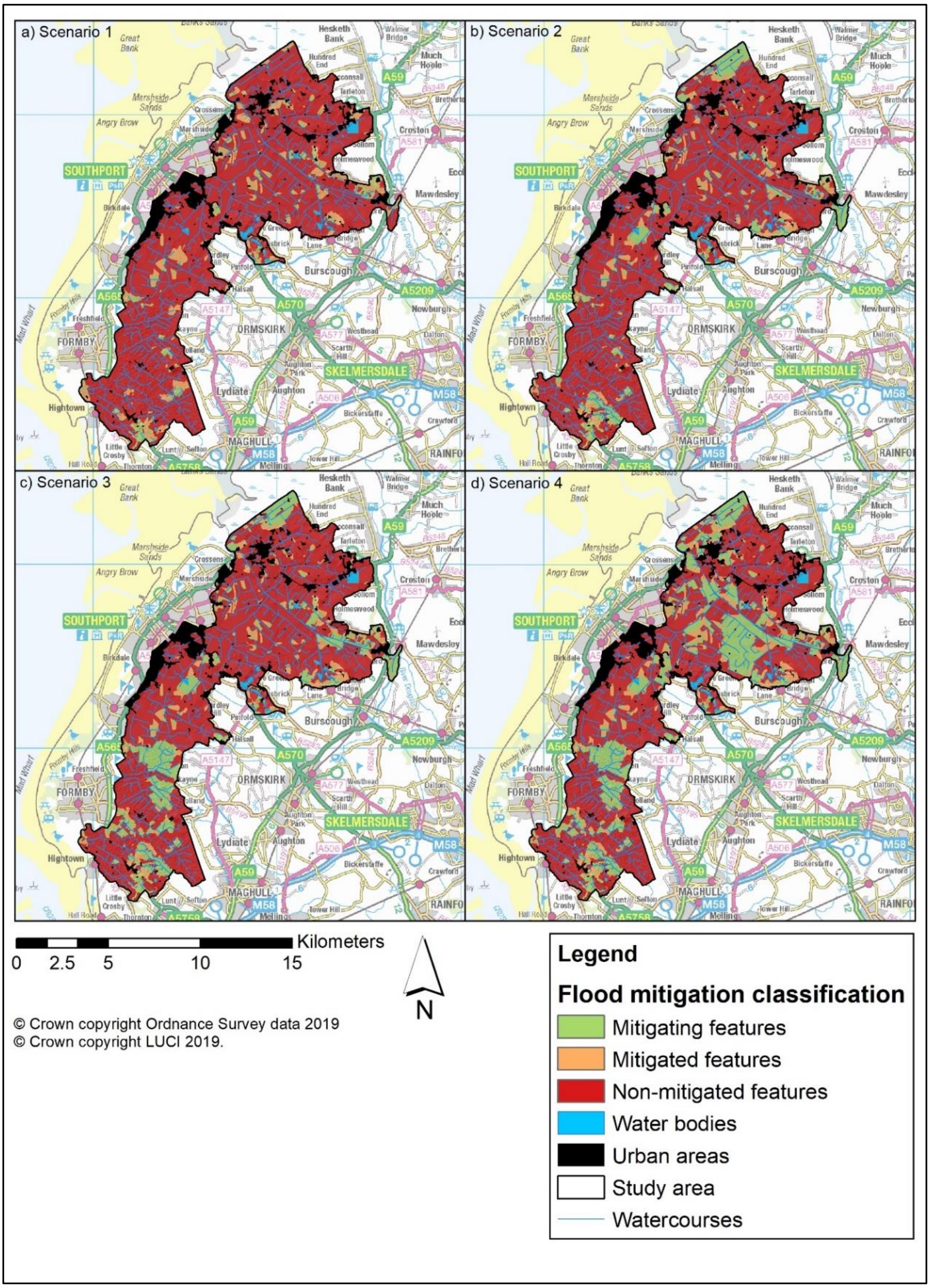


Figure 24: *Flood mitigation classification for the Alt Crossens for Scenarios 1-4.*

The influence of flood mitigating land was illustrated in *Figure 24* to signify the benefits provided by a movement towards a landscape suited to re-wetted conditions in the Alt Crossens. It is clear that the dominant feature from the baseline scenario was non-mitigating features offering no flood protection (86.7%), this was attributed to the largely agricultural basis of the Alt Crossens. Moreover, only 3.9% of the area was found to host natural flood mitigating features, associated with woodland areas and peatlands. Moving away from a drained landscape dominated with conventional agriculture, the benefit of alternative land management strategies for flood mitigation become apparent. This can be observed in *Figure 24* and *Table 8*, with a clear increase in flood mitigating features between Scenarios 1 and 4 (16.7%), a 1.7% increase in flood mitigated land, and an 18.9% decrease in non-mitigated land.

Table 8: *Area of each flood mitigation class under for each land management scenario.*

Status	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Mitigating	3.9	8.6	13.2	20.6
Mitigated	7.8	6.6	7.8	9.5
Non-mitigated	86.7	82.6	76.9	67.8

From *Figure 24*, it can be identified that there was a degree of under-representation surrounding the extent of flood-mitigating features in the study area, in comparison to what was expected within paludiculture-based land uses. Whilst large amounts of the candidate areas for intervention were classified as ‘neutral grassland’, an increased potential for flood mitigation would be expected as more specific land management practices such as sphagnum farming and reed cultivation would provide some form of flood mitigation. Therefore, despite the initial improvement in flood mitigation in the area, there is scope for larger improvements if future LUCI land covers were to include more specific examples of paludiculture.

4.1.6 Nitrogen loading

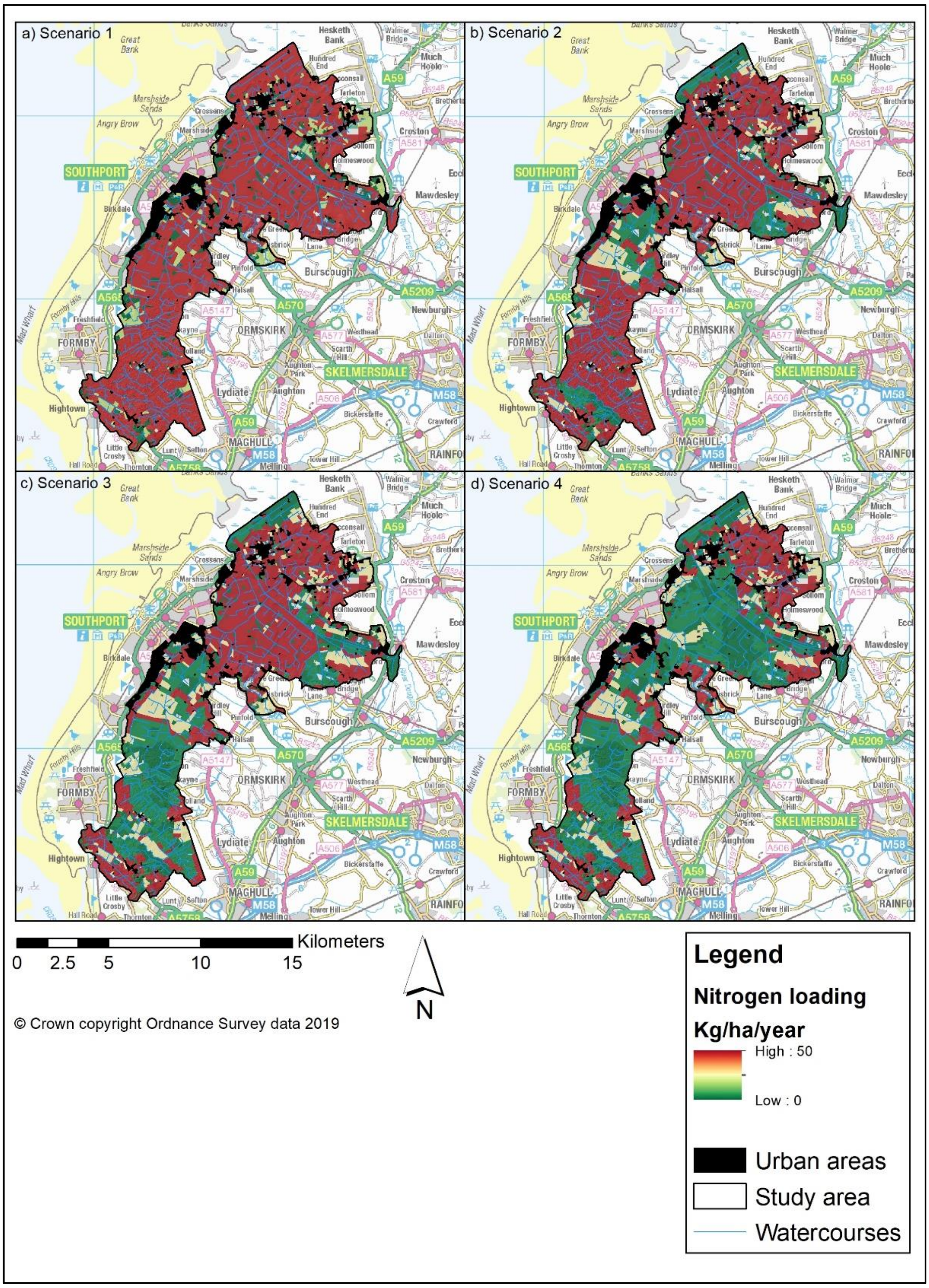


Figure 25: *Calculated annual nitrogen loading (kg/ha/year), reflective of the dominant land cover, based on four different land management scenarios.*

As with the previous ecosystem service analysis, there is a noticeable improvement in the contribution that the chosen land use has to the loading of nitrogen within the Alt Crossens. The reduction in the calculated amount of nitrogen loading between baseline and re-wetted scenarios was illustrated by *Figure 25*. Under present, baseline conditions mean loading of N for the study area was 43.0 kg ha^{-1} yr^{-1}, equating to a total loading of 559 tonnes of nitrogen per year. For Scenario 2, the calculated mean N loading decreased to 37 kg ha^{-1} yr^{-1}, equivalent to 484 tonnes of loading per year. This decrease was further experienced in Scenario 3 with the mean N loading falling to 30 kg ha^{-1} yr^{-1}, totalling 392 tonnes of nitrogen supplied to the study area each year. However, the most significant reduction was observed in Scenario 4, with the calculated mean loading of nitrogen being 22 kg ha^{-1} yr^{-1}, nearly halving the levels of N loading to 290 tonnes of nitrogen when compared to baseline findings (290 tN $year^{-1}$).

4.1.7 Phosphorus loading

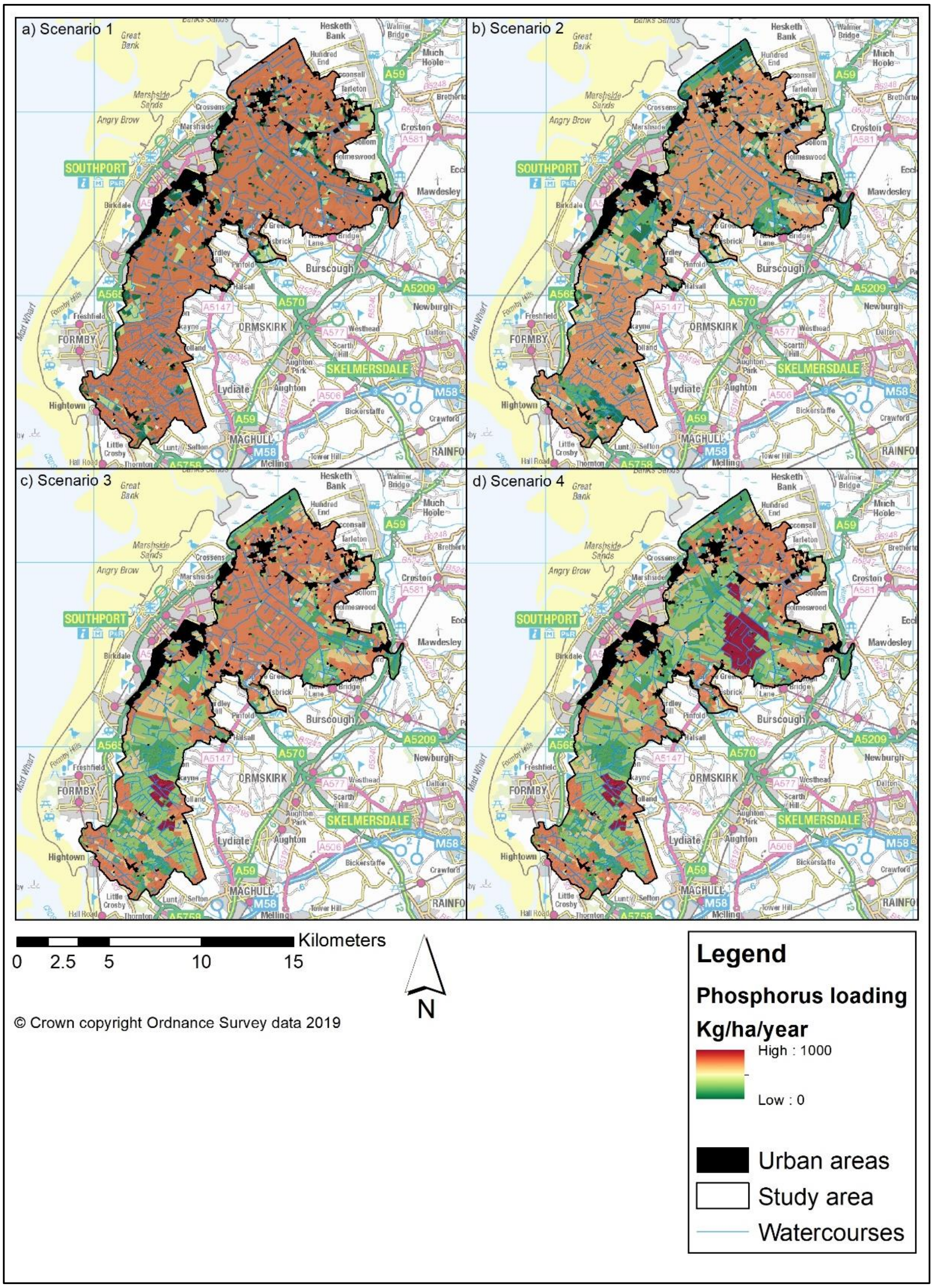


Figure 26: *Calculated annual phosphorus loading (kg/ha/year), reflective the dominant land cover, based on four different land management scenarios.*

In comparison, *Figure 26* reflects the calculated levels of phosphorus loading in the study area. For the generated baseline scenario, mean annual phosphorus loading was found to be 724 kg ha^{-1} yr^{-1}, equating to a total contribution of 9,424 tonnes of P per year. These values decreased to 657 kg ha^{-1} yr^{-1} for Scenario 2, contributing a total of 8,552 tP per year. Scenario 3 continued this trend, decreasing the mean phosphorus loading to 588 kg ha^{-1} yr^{-1}, equivalent to 7,654 of total P loading. However, Scenario 4 experienced the largest decrease in phosphorus loading, whereby mean values fell to 533 kg ha^{-1} yr^{-1}, and 6,938 tonnes of P per year.

One distinguishable difference between the nitrogen and phosphorus outputs was the contribution of phosphorus loading in peatland areas, acting as 'hotspots' with high rates of nutrient loading. The converse was observed for rates of nitrogen loading; peat bogs were found to decrease the loading of nitrogen in the landscape, indicating how further revision surrounding the chosen land use and how LUCI represents phosphorus could aid future applications.

4.1.8 Multiple ecosystem service assessment

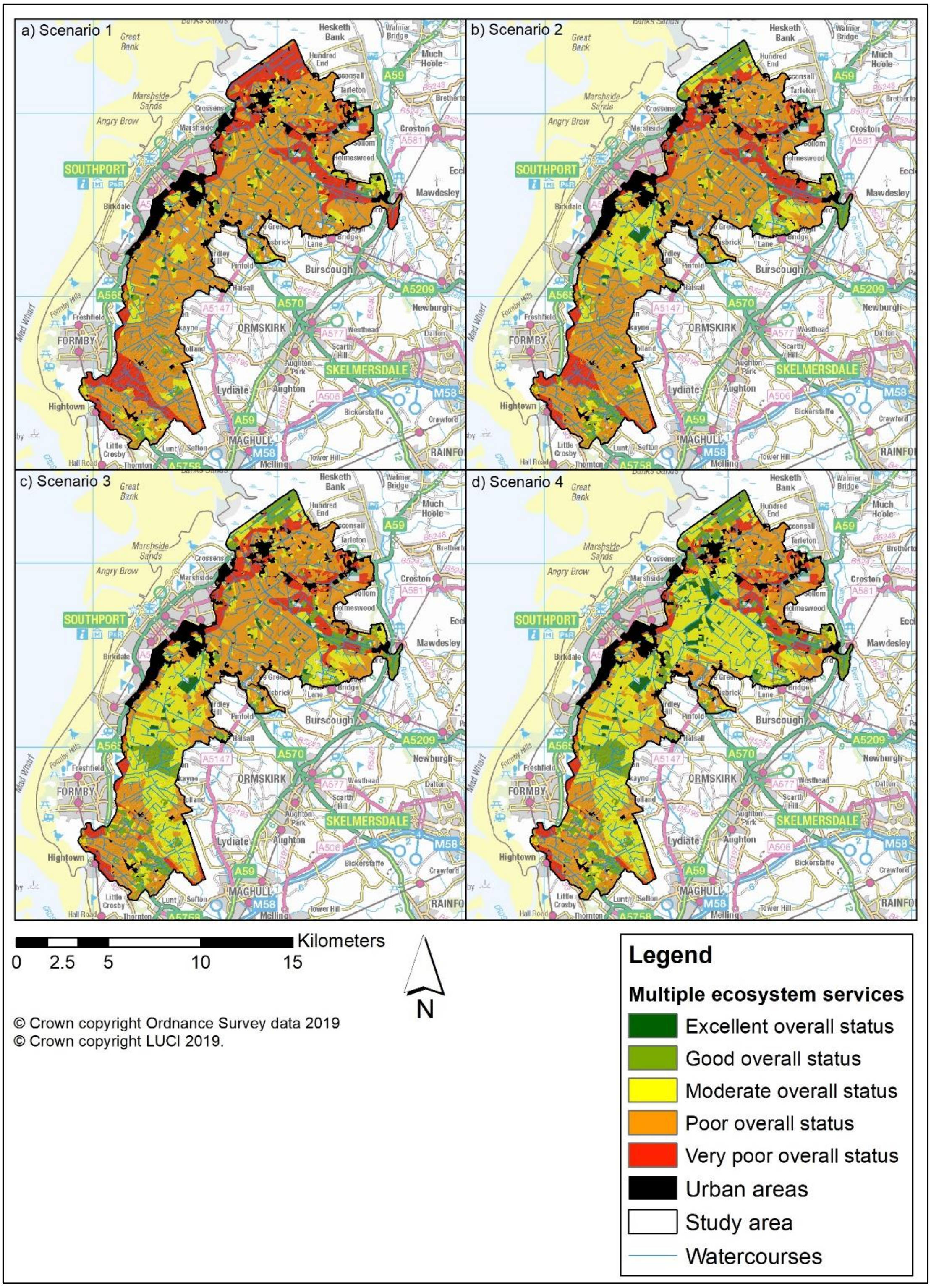


Figure 27: *Multiple ecosystem service assessment for each land management scenario*

Based on the findings from the prior analysis, each scenario was subject to a multiple ecosystem service assessment to analyse the overall status and provision of multiple ecosystem services simultaneously (*Figure 27*). From this, it was found that for the baseline scenario (Scenario 1), only 1% (103 ha) of the study area was identified to have an 'excellent' overall status and 3% (393 ha) having a 'good' status. As illustrated in *Figure 27* (above), 69% (8759 ha) of land was found to be of 'poor' status for multiple ecosystem services, and 18% (2,248 ha) being of a 'very poor' status.

However, through Scenarios 2-4, there was an observed increase in the ability for the Alt Crossens to provide multiple ecosystem services simultaneously (*Table 9*). This can easily be illustrated when comparing Scenarios 1 and 4, with Scenario 4 hosting an additional 223 ha of land with an 'excellent' status, a 1,192 ha increase in land with a 'good' provision of services, and a 3,909 ha increase in land with a 'moderate' status.

Table 9: *Multiple ecosystem service assessment classifications for Scenarios 1-4.*

Multiple Ecosystem Service Status	Scenario 1 ha	Scenario 2 ha	Scenario 3 ha	Scenario 4 ha
Excellent overall status	103	174	179	326
Good overall status	393	941	1,349	1,585
Moderate overall status	1,272	2,120	3,396	5,181
Poor overall status	8,759	7,827	6,480	4,533
Very poor overall status	2,248	1,652	1,311	1,089

4.1.9 Summary of findings

From this section of the results, it can be seen that a scenario-based approach offers additional insight when analysing the potential contribution of different ecosystem services in the Alt Crossens, both independently and holistically. Based on the relative elevation of the land, the location of drainage benefit areas and the relative status of different ecosystem services under the baseline scenario, a movement away from conventional agriculture was adopted, due to an increase in the amount of land unsuitable for arable and horticultural practices in a re-wetted condition. Therefore, the land management strategy was altered to better suit a wetter environmental condition that, following the principles of paludiculture, could enable an improved level of environmental sustainability whilst also enabling alternative methods of agricultural production to occur.

Following these changes to the landscape, each ecosystem service was re-processed using LUCI to assess the potential change in the mix of service provision, which are summarised in *Table 10* (below). Levels of agricultural productivity experienced a clear decline, with the amount of highly productive land falling by 49%. This coincided with a potential agricultural loss between Scenarios 1 and 4 of approximately £5 million per year. However, there were also significant net gains in the landscape; carbon storage in the landscape increased by 234,300 tC, and the spatial extent of land with a high and moderate potential to sequester CO_2 increased by 2.5% and 3% respectively. The presence of flood mitigating land rose from 3.9% to 20.6%. These net gains were also complemented by reductions in nitrogen and phosphorus loading, decreasing from 43.9 kg^{-1} ha^{-1} $year^{-1}$ to 22.3 kg^{-1} ha^{-1} $year^{-1}$ for nitrogen and from 724.4 kg^{-1} ha^{-1} $year^{-1}$ to 532.7 kg^{-1} ha^{-1} $year^{-1}$ for phosphorus. When comparing between the baseline and most re-wetted scenario (Scenario 4), despite the initial loss in agricultural productivity and economic value of the land, there was a distinct improvement in the provision and status of ecosystem services, increasing the extent of land with a 'good' and 'moderate' service provision by 1,192 and 3,909 hectares, respectively.

Table 10: *Overall status and response of different ecosystem services against generated scenarios.*

Ecosystem service tool	**Status of ecosystem services**			
	Baseline	**Scenario 2**	**Scenario 3**	**Scenario 4**
Agricultural productivity	Very high	↓	↓	↓
Economic value	Very high	↓	↓	↓
Carbon storage	High	↑	↑	↑
Carbon sequestration	Very low	–	–	–
Flood mitigation	Very low	↑	↑	↑
Nitrogen loading	Very high	↓	↓	↓
Phosphorus loading	Very high	↓	↓	↓
Multiple ES provision	Low	↑	↑	↑

4.2 Ecosystem services – Lyth Valley

Much like the Alt Crossens application, this analysis involves the generation of a baseline scenario for current, drained conditions, plus a single future re-wetted scenario. This future scenario principally involved the movement away from non-essential land drainage, and land uses better-suited to wetter conditions. As with the Alt Crossens application, an additional parameter was included in the baseline run to negate the influence of waterlogged conditions in low-lying areas, with the converse being applied for the future (re-wetted) scenario. Major land use changes between the two scenarios consist of the large decrease in improved grassland, and the transition towards a less heavily-managed landscape involving increased levels of restored peatlands, broadleaf woodlands, fens and neutral grasslands (*Table 11, Figure 28*). Re-processing this scenario through each ecosystem service tool subsequently enabled an analysis of the influence that land use and associated management could have upon these services, quantifying net gains and losses accordingly.

Table 11: *The change in land cover composition under the baseline (current) and an alternative future scenario.*

Land Cover Map Broad Habitat	Baseline cover (ha)	Future scenario cover (ha)	Change (ha)
Arable and horticulture	36	19	-17
Bog	348	503	155
Broadleaf woodland	211	422	211
Coniferous woodland	3	38	35
Fen, Marsh and Swamp	52	78	27
Freshwater	5	5	0
Improved grassland	2,093	719	-1,374
Neutral grassland	58	1,022	964
Saltmarsh	64	64	0
Suburban	54	54	0
Supralittoral sediment	19	19	0
Urban	6	6	0

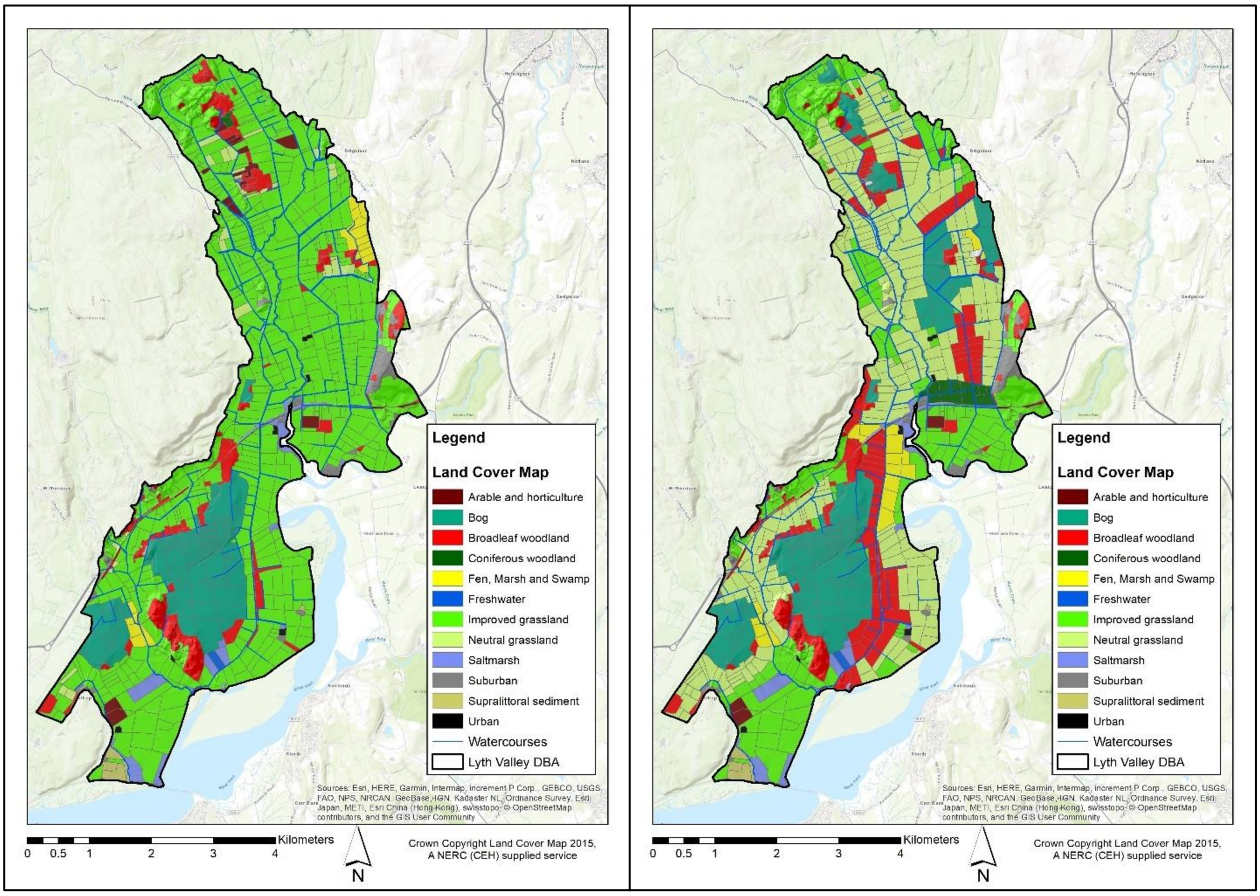


Figure 28: The change in land cover composition under the baseline (current) and an alternative future (re-wetted) scenario.

4.2.1 Agricultural productivity

Agricultural productivity for Lyth Valley was calculated using LUCI's agricultural utilisation tool, based on the current land cover, soil characteristics and drainage conditions of the landscape. Calculated levels of agricultural productivity were overall lower than that of the Alt Crossens, reflective of the differing forms of agricultural land use. As identified in *Figure 29*, a high level of productivity was found for 71.2% of the study area. This was followed by land which offers no agricultural production (16.7%), notably the south west of the valley at Foulshaw Moss and Meathop Moss. Areas with marginal production status were situated around the peripheries of these areas, accounting for 7.6% of the remaining land. Patches of highly productive land were also identified; however, these accounted for 1.6% of the area and were mostly situated on arable and horticultural land.

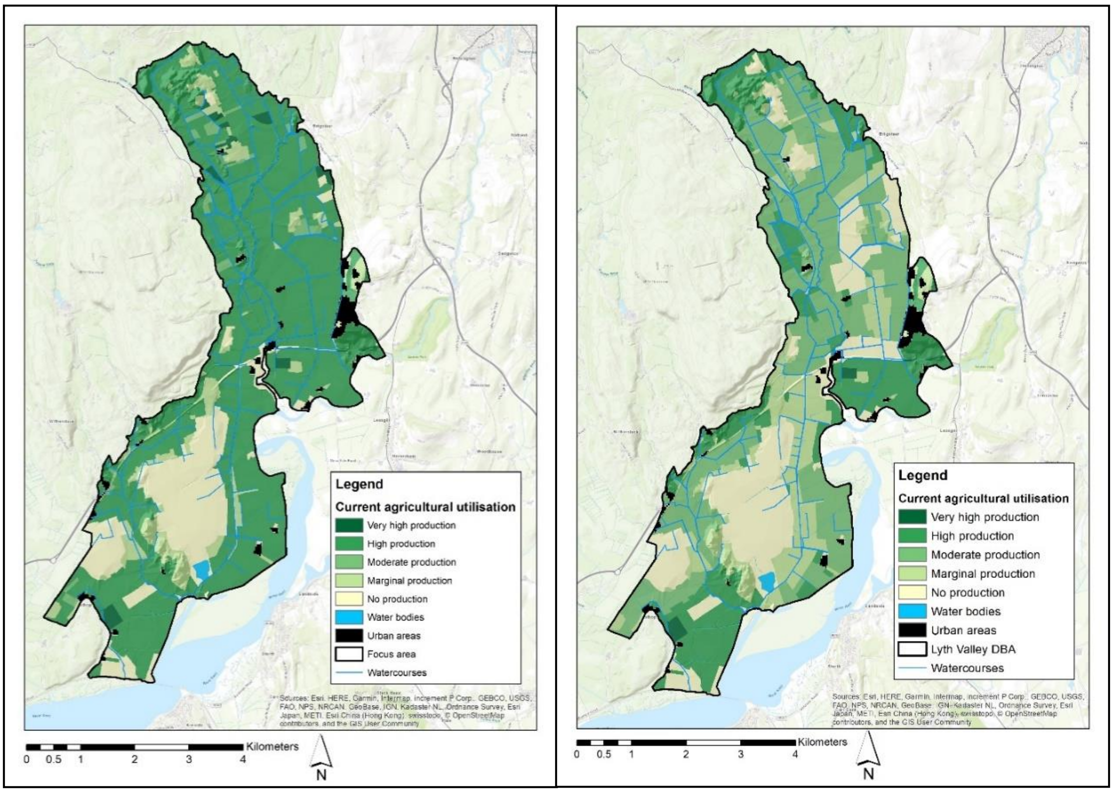


Figure 29: *Calculated agricultural utilisation status of Lyth Valley under drained (left) and re-wetted (right) conditions.*

Under the future scenario, more consistent with how the landscape is anticipated to experience re-wetting, there was a clear decrease in the production status of Lyth Valley. This was to be expected as re-wetting encourages a movement away from conventional agriculture, to a management strategy that is less-damaging to the environment, but at the cost of a reduced intensity of agriculture. This can be quantified by the changes in the spatial extent of each agricultural productivity status in *Table 12*, with a 1,390 ha decrease in land of a very high agricultural utilisation. This loss of highly productive land was spread out to the lower production classes, increasing the extent of land with a moderate production by 950 ha.

Table 12: *Calculated differences in agricultural utilisation status classifications for both scenarios.*

Agricultural utilisation status	Baseline scenario (ha)	Future scenario (ha)	Difference (ha)
Very high	46	18	-28
High	2,099	706	-1,393
Moderate	57	1,008	951
Marginal	223	498	275
No production	495	675	180

The optimal agricultural production tool was run for Lyth Valley under the present environmental conditions, and then again, under a future scenario with no influence of land drainage. No changes to the level of productivity were identified under these two scenarios (*Figure 30*). This indicates that if the drainage channels and pumping stations were to discontinue operation, then there would be no distinguishable loss in agricultural productivity. Equally, *Figure 30* illustrates how the optimal level of agricultural production in Lyth Valley is predominantly land with a 'very high production capacity', whereas the findings from the baseline scenario reflect a landscape with a predominantly 'high agricultural production capacity'. Therefore, LUCI may overestimate the condition of some areas in Lyth Valley, which have been subject to historic degradation, whereas site visits and historic knowledge of the land utilisation indicate that the condition of Lyth Valley's land lacks the ability for widespread highly-productive agriculture.

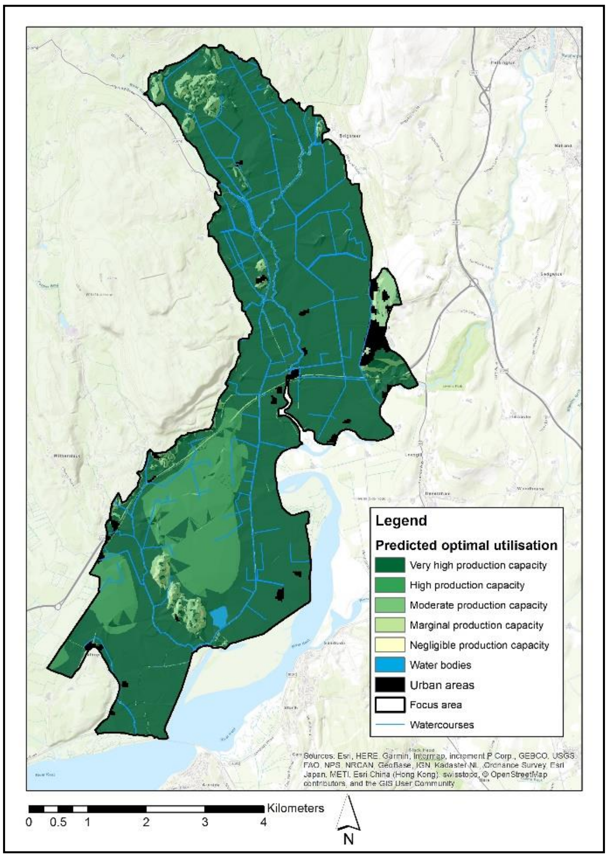


Figure 30: *Calculated optimum agricultural utilisation for Lyth Valley.*

4.2.2 Economic value of the land

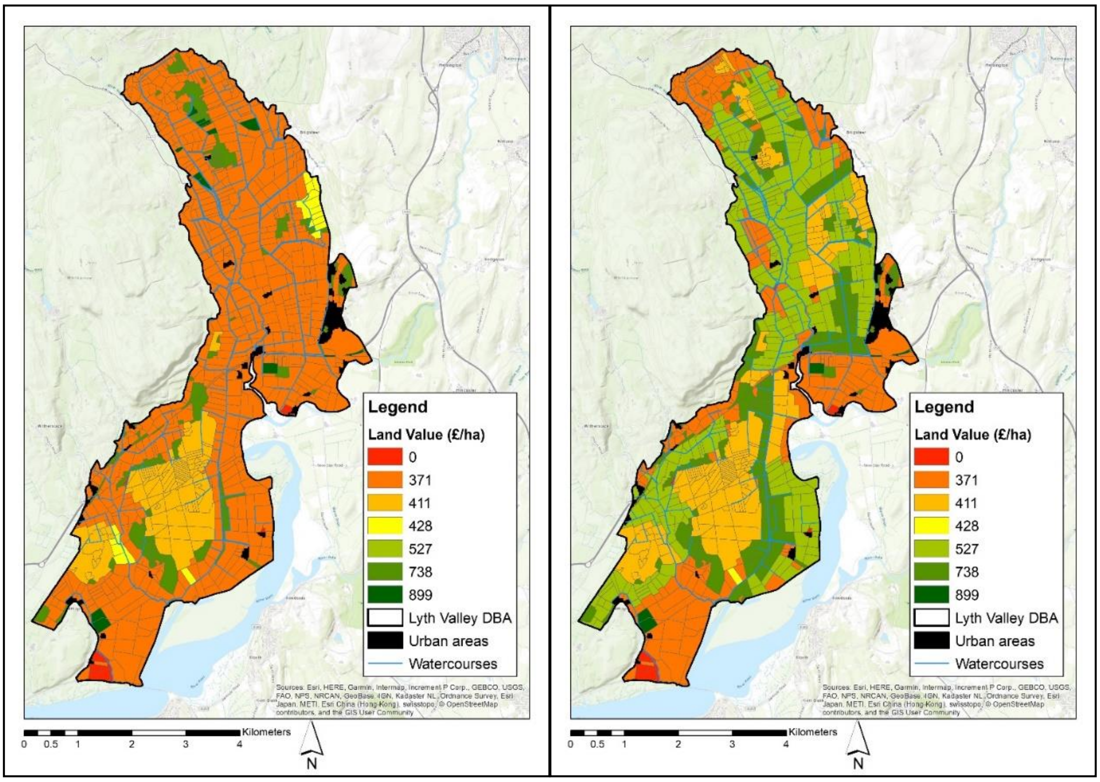


Figure 31: *The economic value of the land in Lyth Valley under current (left) and re-wetted (right) management.*

Figure 31 above represents the potential economic returns of Lyth Valley for each land management scenario. Values for *Figure 31* were homogenous throughout the landscape, representative of how the area is dominated by agriculture, principally improved grassland. The maximum value of land was £899 ha^{-1} per year, representative of small scale arable and horticultural practices in the area, whereas the minimum economic value was linked to land which had undergone significant amounts of modification through urbanisation. Despite this, it was found that the mean economic return for the study area under the baseline scenario was £402 ha^{-1} per year. In comparison, under a re-wetted future scenario, there was a calculated increase in these values, to £465 ha^{-1} $year^{-1}$.

Values representative of the economic returns of each land use were then combined with the area of each field, enabling the calculation of the total economic potential the Lyth Valley for each scenario. Using this, it was found that for the baseline scenario, the total economic potential of the land was calculated to be approximately £1.18 million annually. In comparison, the future re-wetted scenario presents a total economic potential of £1.43 million annually, a £240,000 increase. This signifies how alternative land management, more suited to a re-wetted landscape could enhance the economic return for land users and different stakeholders, when compared against current baseline conditions.

4.2.3 Carbon stock

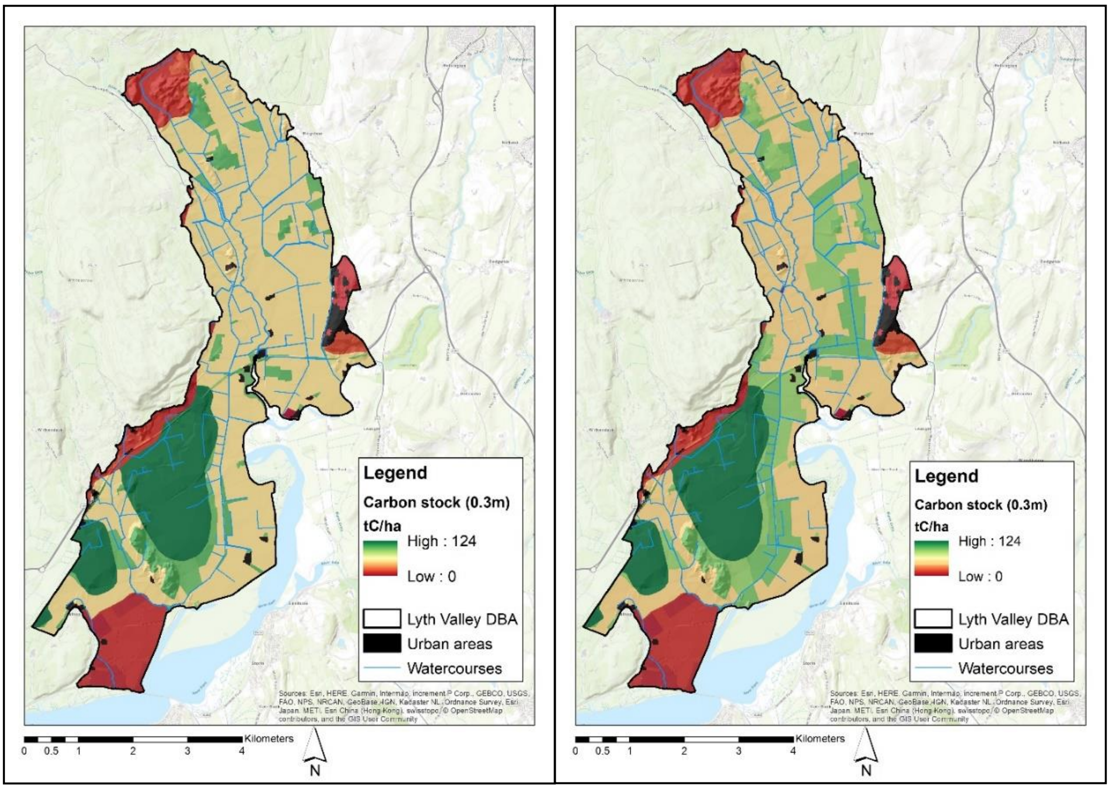


Figure 32: *Calculated carbon stored at a 0.3m soil depth for baseline (left) and re-wetted (right) conditions.*

The carbon stock of both the aboveground and belowground (0.3m soil depth) was calculated for each scenario. It is clear from *Figure 32* that the hotspots for carbon storage are found around Foulshaw and Meathop Moss, with patches of land with high storage potential tied to the presence of woodland and peatland areas. Zones providing no carbon storage were associated with urban areas, with the south-westerly zone lacking storage potential due to a change in the underlying geology. Maximum carbon storage was calculated at 124 tonnes of carbon per hectare for both sites. A small, but notable increase in the carbon storage potential was identified under the future scenario, equating to an increase in carbon storage on the order of 2,771 tonnes of carbon.

4.2.4 Classified carbon emissions

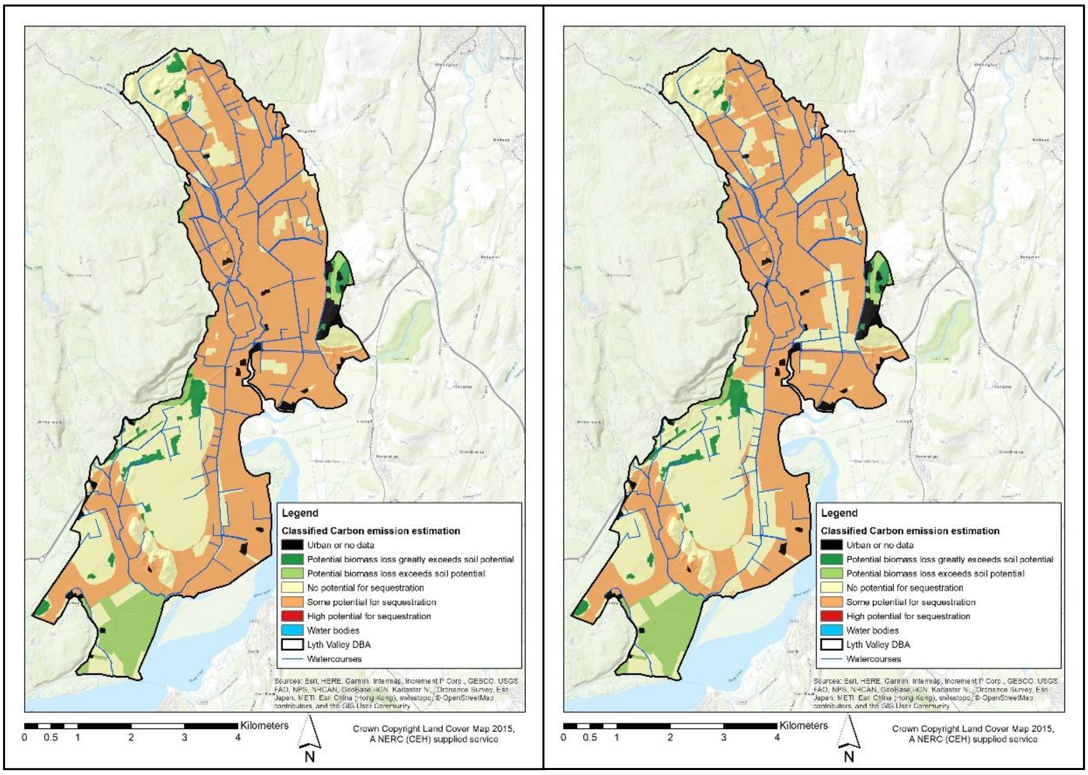


Figure 33: *Classified carbon emission estimation for a drained (left) and re-wetted (right) landscape.*

Figure 33 above illustrates the potential rates of emission and sequestration in the area, based on the land cover and soil type. It can be seen that there are minimal differences between the two scenarios, with the re-wetted scenario exhibiting a 223 ha (8%) decrease in land with some potential for additional sequestration. Subsequently, there has been a small decrease in the presence of land where alternative forms of LULC could impose a loss in biomass carbon which may offset any soil carbon gains. These reductions are linked to the transition away from improved grassland areas, towards the re-establishment of peatland and wetland areas. Given the lack of land with a high potential for additional CO_2 sequestration, small adjustments to the land management regime provided a restricted ability for significant changes to be observed in the future scenario. From looking at the changes in the spatial extent of different land uses between each scenario (*Table 11*), a considerable change was anticipated in the degree of carbon emission under a future scenario. However, the lack of this may lie with the values which underpin these classifications. For example, a change in the land use from improved grasslands to neutral grassland may reflect a clear change in the carbon fluxes with specific applications of paludiculture, however the ability to represent these smaller-scale differences may be restricted by the values which presently underpin LUCI.

4.2.5 Flood mitigation

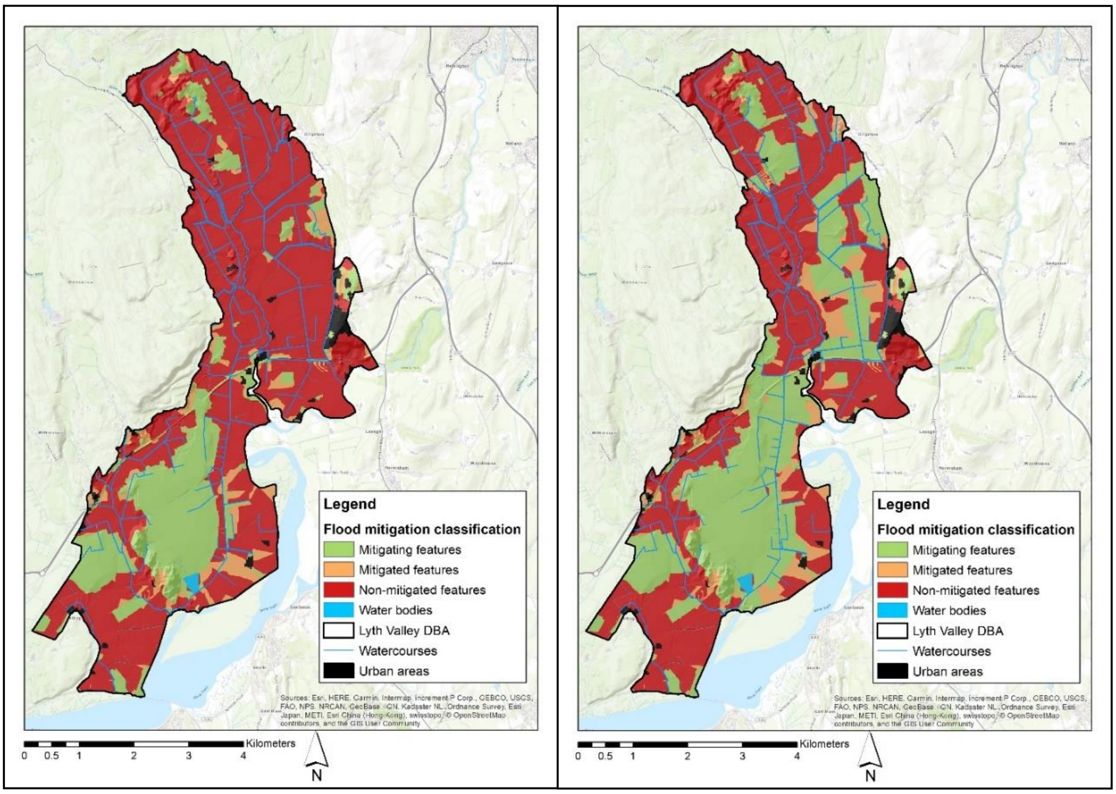


Figure 34: *The presence of flood mitigating features in Lyth Valley under current (left) and future (right) management strategies.*

It is clear that the presence of flood-mitigating features within Lyth Valley is influenced by the type of land use and land cover in each scenario *(Figure 34*). For the baseline scenario 2,077 ha of Lyth Valley (70%) was found to accommodate non-mitigating features, reflective of the how the landscape is dominated by agriculture. Conversely, 639 ha (21%) of the landscape provides flood mitigating features, corresponding to the location of restored peatland and woodland areas. When compared to the findings of the re-wetted scenario, the hydrological function of the landscape improved significantly, reflective of alternative management strategies. The presence of flood-mitigating features in Lyth Valley rose from 639 ha (21% of study area) to 1097 ha (37% of study area), an increase of 457 hectares (16%). Similarly, non-mitigated features in Lyth Valley decreased from 2077 ha to 1531 ha. These changes are reflective of the land management in the study area, building on the principles of paludiculture whereby areas of neutral grassland, peat bogs and woodland offer increased flood mitigation in comparison to conventional agriculture.

4.2.6 Nitrogen loading

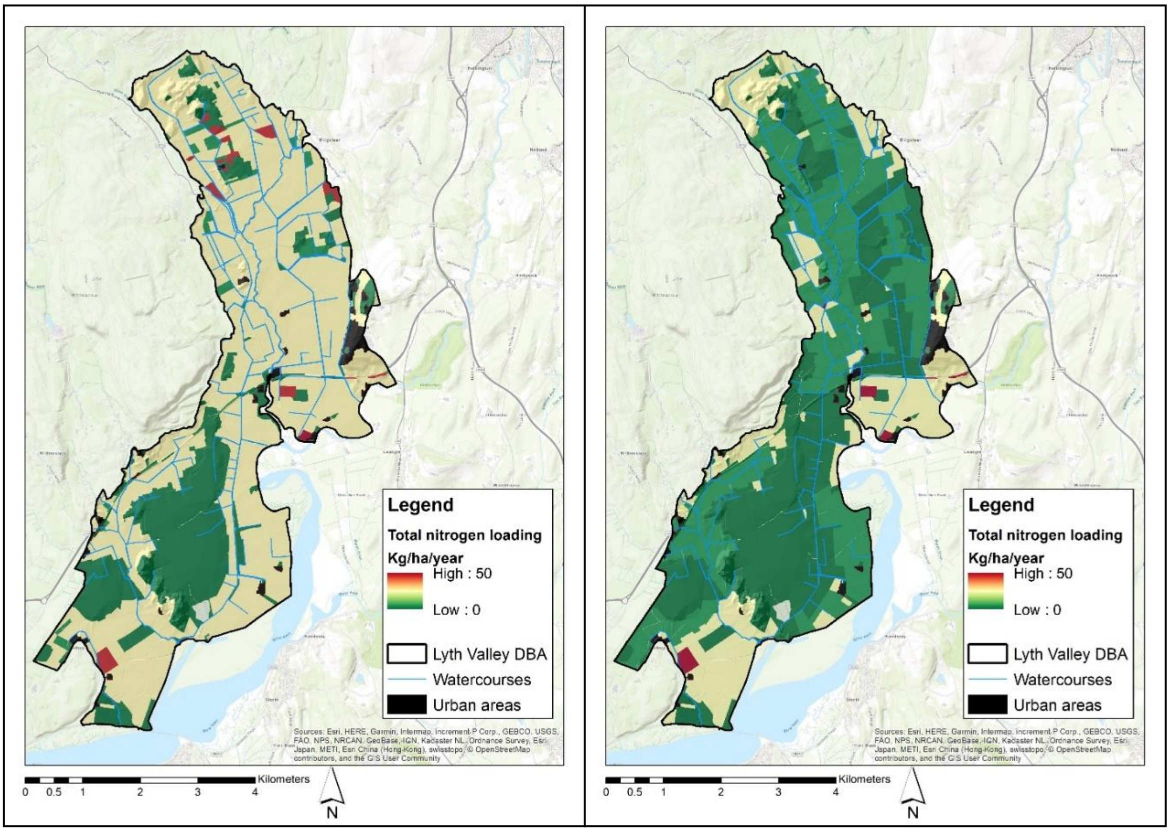


Figure 35: *Total calculated nitrogen loading for Lyth Valley under baseline (left) and re-wetted (right) conditions.*

A clear difference can be observed between baseline and future scenarios with regards to the presence of land with a lower/minimal nitrogen loading influence within Lyth Valley (*Figure 35*). This was identified by the areas which have been subject to changes in the land cover and land management, reflective of paludicultural practices that require minimal (if any) fertiliser application, hence making lower contributions of nitrogen to the landscape. This can be quantified by the decrease in mean loading values, falling from 21 kg^{-1} ha^{-1} year^{-1} for current conditions, to 9 kg^{-1} ha^{-1} year^{-1} in the re-wetted scenario. Surrounding areas where improved grassland and arable farming could persist maintained the same values of nitrogen loading for both scenarios.

4.2.7 Phosphorus loading

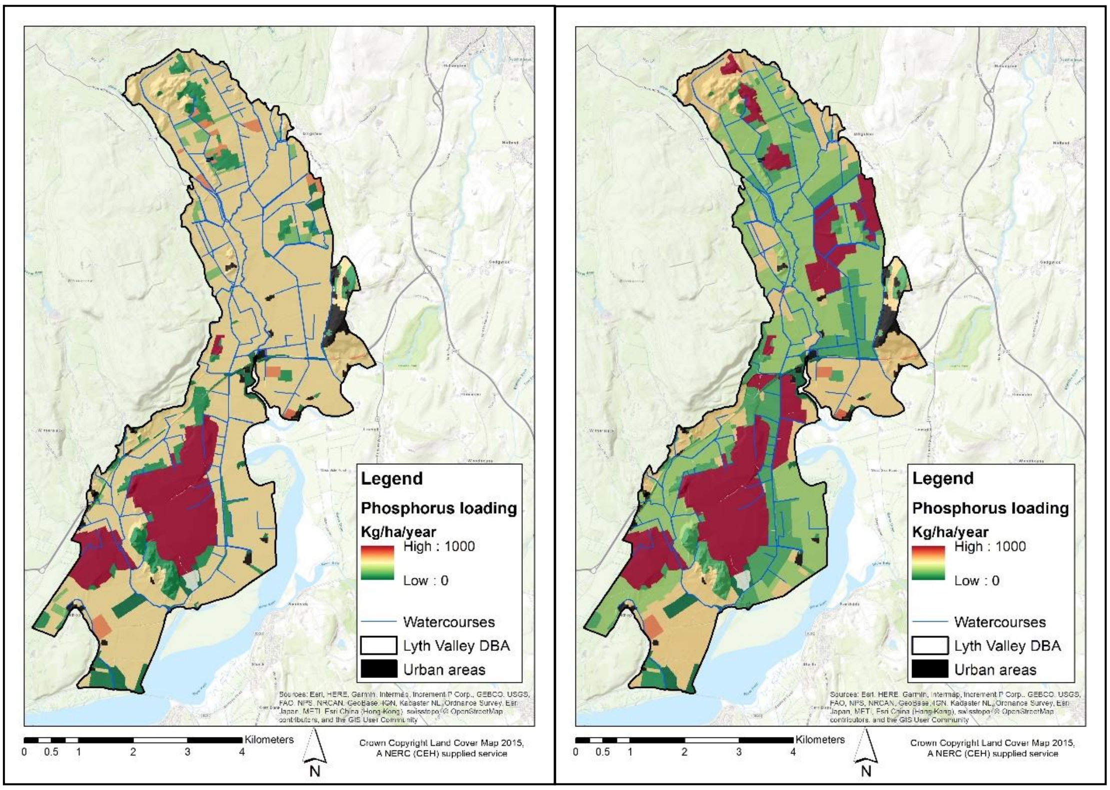


Figure 36: *Total calculated phosphorus loading for Lyth Valley under baseline (left) and re-wetted (right) conditions.*

The same trends from nitrogen loading were observed in *Figure 36*, with a clear reduction in the contribution that phosphorus loading has to the potential water quality in Lyth Valley. This is reinforced by the mean loading of the two scenarios, decreasing from 607 kg^{-1} ha^{-1} $year^{-1}$ to 493 kg^{-1} ha^{-1} $year^{-1}$. Again, the increased presence of neutral grasslands and broadleaf woodlands in the re-wetted scenario is associated with a lower phosphorus loading. This signifies where intervention in the landscape could lead to a further reduction in the contribution of phosphorus loading for Lyth Valley. However, as with the contribution of phosphorus loading in the Alt Crossens, peatlands are flagged as hotspot areas where phosphorus pollution is particularly high.

4.2.8 Multiple ecosystem service analysis

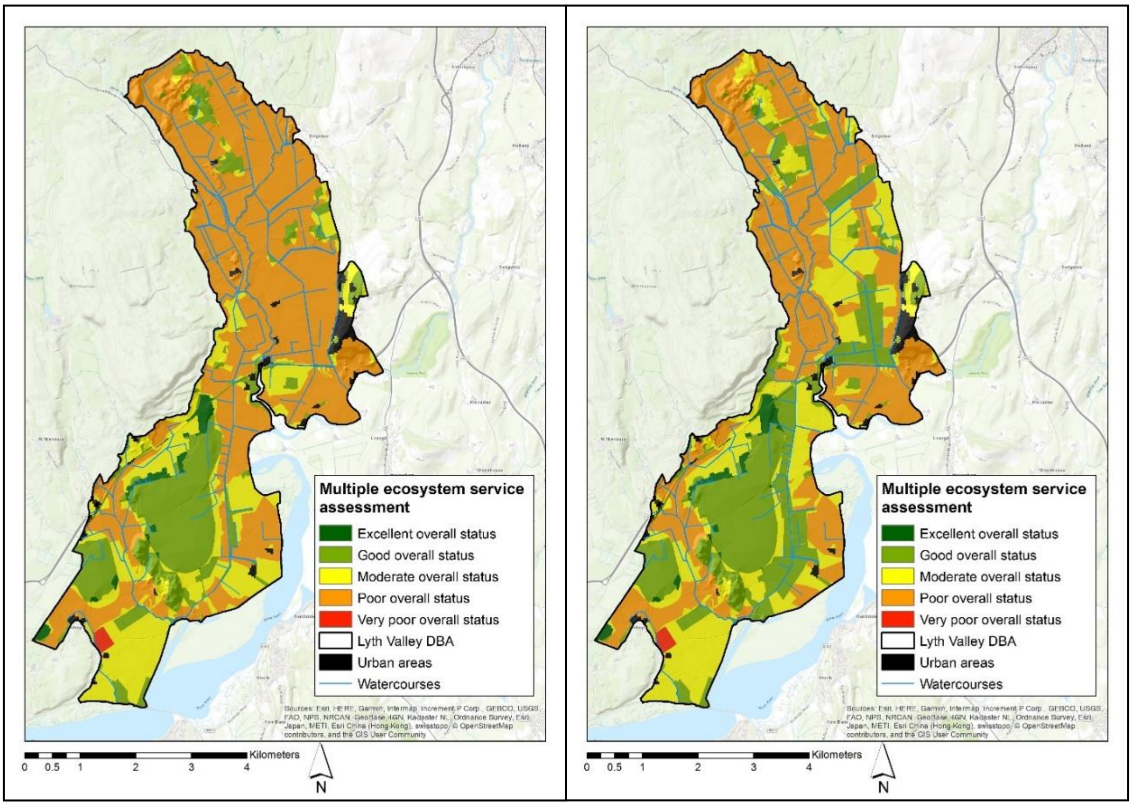


Figure 37: *Multiple ecosystem service assessment, based on the present (left) and future (right) levels of land management and drainage.*

Under current (baseline) conditions, it can be established that a significant proportion of Lyth Valley (58%) exhibits a 'poor overall status' in the provision of multiple ecosystem services, attributed to the improved grassland forms of land use (*Figure 37*). Very few areas were identified to have a 'very poor' status (0.4%), likely due to the lack of arable and horticultural practices in Lyth Valley. Only 19% of the study area (553 ha) in the baseline scenario were found to exhibit an overall 'good' provision of multiple services, limited to woodland and peatland areas.

For the re-wetted scenario, a small improvement in the provision of multiple ecosystem services was identified with a 'good' overall status increasing from 553 ha to 783 ha (*Figure 37*). Moreover, a 186 ha increase in land with a 'moderate' ecosystem service provision was observed, both of which were attributed to a 434 ha decrease in land with the 'poor' provision of multiple ecosystem services. This analysis signifies that despite the initial high level of agricultural production in the area under the baseline scenario, the provision of additional ecosystem services was lacking. Therefore, the re-wetted scenario with alternative forms of land use has allowed agriculture to persist whilst also improving the relative status of the remaining ecosystem services in this study.

4.2.9 Summary of findings

Table 13 (below) summarises the observed changes in the condition of different ecosystem services in a re-wetted future scenario, when compared against that of the calculated baseline. For agricultural productivity, the re-wetted scenario experienced an overall decrease when compared against the current baseline, reflective of the transition away from conventional agriculture (improved grassland) towards more sustainable examples of land use. Using the optimal agricultural production tool, LUCI predicted the ideal levels of agricultural utilisation for both areas, however no difference in the levels of production were observed, indicating that if land drainage ceased then no effects would be felt upon the ability to continue agricultural production. Using an economic assessment, it was found that the re-wetted scenario provided an increased level of economic gain for the land owners and users, increasing by £240,000 from the baseline.

Results from the classified carbon emission tool provided a small (223 ha) decrease in land with 'some CO_2 emissions', however this application was limited by LUCI's inability to represent actual carbon emissions, instead examining the potential for additional carbon sequestration. Conversely, the carbon stock analysis provided some larger increases in carbon storage for the area, amounting to the increased storage of 2,771 tonnes of carbon in the landscape. There was also an observed increase in the presence of flood mitigating features in the re-wetted scenario, rising from 639 ha to 1097 ha. These benefits were also observed for both aspects of nutrient loading in the landscape, with mean rates of both nitrogen and phosphorus falling by 11.5 kg^{-1} ha^{-1} $year^{-1}$ and 114.0 kg^{-1} ha^{-1} $year^{-1}$. When each aspect of the analysis was combined to calculate the overall provision of multiple ecosystem services, there was an observed 434 ha decrease in the spatial extent of land with a poor service provision. These results signify the potential net gains for different ecosystem services under a re-wetted scenario with relatively low levels of land modification, moving away from conventional land drainage and management practices.

__Table 13:__ Overall status and response of different ecosystem services against generated scenarios for Lyth Valley.

Ecosystem service tool	**Status of ecosystem services**	
	Baseline	**Future (undrained)**
Agricultural productivity	High	↓
Economic value	Poor	↓
Carbon storage	Moderate	↑
Carbon sequestration	Low	–
Flood mitigation	Very low	↑
Nitrogen loading	Moderate	↓
Phosphorus loading	Moderate	↓
Multiple ES provision	Low	↑

5 Discussion

5.1 Overview

This chapter will discuss the key themes highlighted throughout this project; interlinking the findings within the context of the academic literature to explore how the research aims and objectives have been fulfilled. The application of LUCI will firstly be critically analysed for each ecosystem service, followed by an assessment into the ability to implement future changes in either study area. Barriers will be discussed in detail, taking into account the potential social, economic, environmental and political implications faced by a change in contemporary land use. Finally, limitations surrounding the application of the research strategy will be examined to address the potential for ability for this decision-support framework to be improved in future applications.

5.1.1 Key research findings

- Based on the hydrological function of LUCI and the presence of low-lying land, the Alt Crossens was found to be at a significant risk of waterlogging, should conventional methods of land drainage be abandoned. In comparison, the re-wetting of Lyth Valley was found to exhibit minimal potential agricultural losses.
- Ecosystem service models offer technocentric solutions to challenges associated with unsustainable forms of land use; a decision-support framework was created, centred around the Land Utilisation and Capability Indicator (LUCI), to provide informative outputs that could be used by a variety of stakeholders surrounding the effects of land use decisions.
- A scenario-based approach was adopted to represent a phased movement away from land drainage. Novel and innovative forms of land management were introduced to both sites, consistent with the principles of paludiculture, to create an inclusive solution.
- A 'win-win' situation was identified in Lyth Valley, with paludiculture-based practices being shown to improve both stocks of natural capital and the economic potential of the land.
- A large-scale movement away from non-essential drainage in the Alt Crossens could decrease economic returns by £5 million. However, opportunities to improve the status of ecosystem services whilst minimising economic losses through a phased approach could be adopted.

5.2 Findings and analysis of results

5.2.1 Agricultural productivity

The key finding from the generated agricultural productivity outputs was how the implementation of 'wetter farming' was associated with a reduced level of agricultural productivity, based on using LUCI's rank-based system. Lower levels of production were observed for both sites, particularly where a transition away from arable and horticultural practices occurred. However, the ability for these conclusions to be fully-backed was limited by LUCI's capacity to represent detailed LULC classifications beyond what is currently offered within the 2015 Land Cover Map Broad Habitat classes. This signifies how levels of agricultural productivity were somewhat undervalued, as paludiculture-based land uses were largely classified as 'neutral grassland' and 'bog' – both of which typically exhibit a minimal level of agricultural production.

Using the optimal agricultural productivity tool, mixed results were produced for both sites as to the influence that re-wetting would have upon levels of productivity. A large proportion (48%) of the Alt Crossens focus area was found to host conditions capable of allowing agriculture to persist. In contrast, under the same re-wetted conditions for Lyth Valley, LUCI calculated that no areas which had a negligible capacity for agricultural production. Based on these findings, the two study areas had a varying level of dependency on artificial drainage to enable agriculture to persist. The Alt Crossens focus area was predominantly low-lying (<5m AOD) on peaty soils, whereas Lyth Valley had a lower presence of low-lying land with freely-draining soils that could enable agriculture to continue in an undrained condition. This signifies how Lyth Valley has a higher level of resistance to re-wetting and that current management practices would be expected to experience a smaller agricultural loss in comparison the Alt Crossens. However, this also highlights the potential limitations in the hydro-topological model, whereby reliability could be enhanced by datasets with a higher spatial resolution and the long-term water table levels in both sites. Whilst these points were beyond the scope for the timeframe in this application, it is important to acknowledge how future iterations could build upon these findings to create more conclusive results.

Furthermore, it was apparent that the outputs from the Agricultural Productivity toolkit lacked a level of detail in providing values representative of crop yields and the value of land; therefore, an economic valuation was conducted to augment the agricultural productivity aspect of this analysis. This significantly improved the assessment of productivity and economic returns from different land parcels. This was achieved by linking crop types to anticipated yields, based on the known Agricultural Land Classification, and the market value of said produce (Craig, 2018). A much clearer picture of the differences between baseline and future conditions was established, building on published economic and harvestable yield values for paludiculture. This was shown to reduce magnitude of initial agricultural losses from previous agricultural productivity outputs, more accurately representing the feasibility of paludiculture against current land management practices. For Lyth Valley, improving the economic returns of areas with alternative land management illustrated how land owners and tenants would benefit in the long-term from a transition away from conventional agriculture, enabling a cost-benefit analysis to be conducted to determine the most feasible form of future land management.

5.2.2 Carbon

It was found that under future land management scenarios, both study sites had an improved capacity to store carbon within the landscape, indicative that the chosen LULC combinations were beneficial to the provision of this ecosystem service. This was consistent with the practices of paludiculture, with the slow replenishment of peat-forming vegetation under anaerobic conditions (Joosten et al., 2012; Wichtmann et al., 2016). When illustrating the changes in these classes for different scenarios, it is clear that on a smaller scale (Lyth Valley), there were minimal levels of improvement in comparison to the findings from the Alt Crossens. Reasoning for this stems from the idea that the levels of carbon storage the dominant form of land use in Lyth Valley (improved grassland farming) are similar to that of 'wetter farming' and paludiculture (neutral grasslands), hence a lower change was observed. Whereas, the Alt Crossens experienced a much larger change between baseline and future scenarios, highlighting how improvements in carbon storage were subject to the transition from arable farming to low-intensity farming.

The 'classified carbon sequestration potential' maps identified where existing levels of carbon storage were already high, and how a change in the land management conditions could improve future levels of the area. There was a reduced ability to extract information surrounding the sequestration potential for either site; however, when comparing baseline results to future iterations there was an observed reduction in land with a potential for additional carbon sequestration in the Alt Crossens. The extent of success in improving carbon sequestration in Lyth Valley was somewhat limited, with the smaller improvements in the potential for additional sequestration being identified, linked to the optimisation of LULC in re-wetted scenarios.

5.2.3 Flood mitigation

Whilst flood mitigation is an ecosystem service typically undervalued in decision frameworks, given the low-lying status of both study areas and the history of flooding (most notably in December 2015), the relative importance in the ability to naturally mitigate against flooding is crucial (EA, 2017).

The results from the flood mitigation tool for both study sites provide qualitative outputs that demonstrate the ability for alternative land management strategies to provide additional mitigation against the threats posed by flooding, through an increased presence of flood-mitigating features and slower hydrological routing of near surface flow (Jackson et al., 2013). Results from Lyth Valley indicate a clear opportunity to improve the influence of flood mitigation in the areas through paludiculture (Paludiculture UK, 2017). This was also shown to be the case for the Alt Crossens, with a movement away from flood-generating land uses, such as arable farming, towards conditions capable of creating buffer zones and barriers to the hydrological routing of the landscape. However, the ability to improve flood interception is largely dependent on the chosen drainage scenario and land management practices implemented.

5.2.4 Nitrogen and Phosphorus loading

The findings from the nutrient loading outputs signify how, under present baseline conditions, both the Alt Crossens and Lyth Valley have a high degree of nitrogen (N) and phosphorus (P) loading within the landscape. Point sources of N and P, such as sewage treatment sites and septic tanks, were not covered in this application, meaning that predominant sources of N and P were attributed to the influence of agricultural systems, notably the application of pesticides and fertilisers.

Trodahl et al (2017) identified how the two methods should be adopted to reduce the influence of agriculture on water quality; either by targeting and decreasing the direct sources of N and P or by focusing on understanding the pathways to the groundwater reserves. A change in the overriding LULC was chosen as the most suitable means of reducing the loading of these chemicals, whilst simultaneously creating wetland buffer zones capable of reducing the direct passage of N and P into watercourses. Trodahl et al (2017) also signifies the possibility in achieving significant reductions in the leaching of N and P to groundwater stores, whilst maintaining a high level of agricultural production. However, in the context of a re-wetted future scenario LUCI is yet to be parameterised to include the influence of management practices, such as targeted applications and shallow injection (Rütting et al., 2018).

A large reduction in the loading of N was observed in future scenarios for both sites, through the creation of neutral grasslands and peatland areas, moving away from arable farming and improved grasslands. The decrease in N loading was particularly apparent in Lyth Valley, lowering to more than half of the present values. A similar loss was also observed in the Alt Crossens, most noticeable within Scenario 4, however the calculated mean of N loading remained high in comparison to that of Lyth Valley. This may be explained by the persistence of arable and horticultural practices in low-risk areas where waterlogging was not anticipated.

Distinct decreases in the levels of phosphorus loading were also observed where alternative management strategies had been implemented. Although initial P loading values were much larger in comparison to the contribution of N in the landscape, a much larger decrease in calculated values was observed within the proposed land use changes. The spatial extent of these changes was found to reflect that of the prior analysis, with one distinct difference. Areas designated as peatlands, alongside areas highlighted for peatland restoration, were found to accommodate the highest levels of P loading at both study areas. Whilst it is commonly found that peatland and wetland ecosystems are long-term nutrient stores, there are examples of N and P exports in artificially-drained peatlands (Tuukkanen et al., 2017). This statement would be applicable to both sites should the presence of drained peatlands exist in future scenarios; however, the converse was applied, re-establishing wetlands with a higher water table. These findings indicate that the representation of P loading for restored peatlands within LUCI may be inconsistent academic literature.

The major findings from this indicate the overall positive influence that future management strategies and re-wetting has upon both study sites, reducing the overall loading of both N and P to the landscape in the areas. However, it is worth noting that should re-wetting occur and select areas continue with conventional agricultural practices, allowing fertiliser and pesticide applications to persist, there will be a reduced pathway for N and P to leach into groundwater stores, due to the presence of a higher water table (Tuukkanen et al., 2017; Rütting et al., 2018).

5.2.5 Multiple Ecosystem Services

It is clear from a 'combined' perspective that both study areas experienced improvements in the provision of a variety of ecosystem services simultaneously, aiding the ability for land managers to realise an improved potential from the landscape. This demonstrates how the deliverance of environmental benefits can be successfully integrated within agricultural systems to ensure an enhanced level of environmental stability, mitigating against the potential losses experienced by a poor crop harvest one year. This also signifies the ability to identify and establish high nature value (HNV) farmland within the UK, an area which is regarded as crucial for the EU in meeting its 2020 biodiversity targets (Maxwell et al., 2017). However, the application of biodiversity modelling is an area somewhat lacking within this application and is has yet to be extensively parameterised within LUCI. Whilst Biodiversity Net Gain, the increase in flora and fauna communities and species, has been shown to strongly influence the provision of ecosystem services, the converse may not always be applicable (Dıaz et al., 2005). This calls for a more informed use of landscape planning in regard to identifying and implementing alternative forms of land management that have been shown to support an improvement in both ES provision and biodiversity. Through paludiculture, with the informed re-establishment of semi-managed wetland habitats (sphagnum moss lawns, alder plantations and reed beds), there would be a distinguishable increase in both the Environmental and Biodiversity Net Gains of both study areas, compared against the current forms of heavily-managed land uses (Maxwell et al., 2017). Whilst the complete return to a natural landscape through conservation could offer higher levels of particular ecosystem services, notably carbon sequestration and storage, there would be minimal agricultural production in the area. This highlights the importance of creating trade-offs and co-benefits within the landscape to satisfy the requirements of different stakeholders to the highest achievable level.

5.3 Barriers to implementing change

5.3.1 Authoritative issues

Through the formation of Internal Drainage Boards (IDBs), there is a possibility that current levels of hydrological management could follow the *status quo* in future for both the Alt Crossens and Lyth Valley. This would counteract any arguments promoting the ability to implement re-wetted LULC practices to provide wider benefits to the environment. However, there are a number of economic, societal and political barriers that pose risks to the establishment of such IDBs. Most notably, the fact that currently IDBs utilise the 1991 Land Drainage Act (LDA) as a basis to value domestic land, an area which is now 30 years out of date, meaning that the economic feasibility of a new IDB is incredibly difficult to pass (ADA, 2018). In previous consultancy assessments, the low value of land in Lyth Valley was found to be insufficient to maintaining drainage practices (JBA, 2015). In contrast, whilst the Alt Crossens an improved economic feasibility through higher value land, there is still a major requirement to update LDA policy to ensure that values are consistent with environmental legislation. Should an IDB be successfully created for either study site, this will significantly improve the possibility of conventional agriculture to persist for many years to come. Under present forms of agriculture on areas of existing peatland, there begs the question as to how long conventional agriculture can persist before the environmental implications become too significant to avoid. Land managers should consider the present status of agricultural land and look forwards to determine how these values will influence agricultural yields under future drainage conditions and societal requirements. Equally, decision-makers should acknowledge how the success of wetland restoration and the provision of multiple ecosystem services is influenced by the condition of the landscape (Bain et al., 2011). This signifies the importance in understanding the relative fertility of agricultural land within the context of each catchment.

5.3.2 Stakeholder views

Another clear barrier to implementing alternative land management practices is the reluctance that the majority of land users have towards deviating from the *status quo*. To overcome this, examples of social learning should be considered to provide working examples that alternative management strategies, like paludiculture, has a place in today's society and can provide benefits for all stakeholders. The provision of ground-proofed working examples works as an extension of this, whereby information is crucial in overcoming conflicting views and discovering a shared social purpose (Mostert et al., 2007; Newson, 2009). Keen at el (2012) further reinforces this concept by stating how social learning is vital in ensuring social change, enabling society to address environmental challenges on both a local and global context.

Palomo et al (2018) signify the presence of a 'bottleneck' when communicating detailed information and results between land users (decision makers) and land users (mainly farmers in this instance). Often the differing requirements for detailed analysis can limit the ability to discuss different options for alternative management, dependent on the size, scope and number of participants in of a study. To overcome this, workshops and meetings should look to clearly communicate the influence that specific decisions possess in fully-realising the potential of ecosystem service provision through trade-offs and co-benefits. An example of this can be identified by Paludiculture UK (2017), a workshop between numerous different stakeholders to promote the discussion surrounding the possibilities of implementing paludiculture within Cumbria. Whilst it is true that paludiculture is only part of the answer to tackling the challenges faced by re-wetting, of working examples can start to build support behind a movement away from the *status quo*.

5.3.3 Food security

There is also the question surrounding food security in the UK with a shift towards a much wetter landscape in the North West of England. As there is an ever-increasing demand in achieving the maximum possible crop yields, the process of re-wetting agricultural land is counter-productive and adds additional stress to agricultural landscapes in other areas of the UK. Implementing paludiculture would exacerbate this problem through the large-scale removal of land used for food production, favouring land uses which provide alternative ecosystem service benefits, such as the production of biomass for combustion. To minimise these losses, one option is to simply continue in arable farming in the areas which host a low risk to waterlogging conditions. By focusing on the cultivation high-yield crops (potatoes and cereals), instead of pastureland and the production of fodder for livestock, there could be a more efficient means of managing the land available for farming. The use of informed farming decisions may prolong the ability for agriculture to exist in the Alt Crossens, minimising soil subsidence and peat wastage through the utilisation of flood-tolerant crops with shallower rooting depths (Mustroph, 2018; Shiono et al., 2019). However, more needs to be done in assessing the expected increases in the water table on a more localised scale.

5.3.4 Paludiculture-derived issues

There is also the challenge of implementing chosen methods of alternative management strategies in both study areas. Whilst a range of commercial opportunities are offered through paludiculture, there needs to be enough demand to justify the transition away from a food-based agricultural system. The cultivation of reed beds have been shown to provide a sustained high level of biomass (Wichtmann, 1999), however the low-energy yield from the combustion of pellets and briquettes requires low transport distances between source and processing plants to ensure economic and environmental efficiency (Thomas et al., 2013). Therefore, some consideration should be taken into account surrounding the supply chain of raw materials and the scale demand for these products in the UK.

Large set up costs are also attributed towards the implementation of paludiculture-based land uses due to the value of specially-modified machinery and the establishment of certain plants, such as peat-forming sphagnum diaspores (Wichtmann et al., 2017). Once proved to work within the UK, paludiculture should develop from pilot studies towards the large-scale adoption practices. However, this is only feasible if practices are economically viable from a business perspective (Wichtmann et al., 2016). Therefore, it is crucial for the UK is to secure funding for small-scale pilot sites in order to illustrate how a movement away from conventional agriculture is still economically viable. Whilst paludiculture may hold the key to sustainable land management, the ability to implement these changes on both a large and small scale is limited by the access to funding, meaning a lag in the ability to cover investment costs until a market for the products of paludiculture is fully-established.

5.4 The unknown future of agri-environment policy

It is important to take into consideration the UK's current plan to leave the EU Common Agricultural Policy (CAP) through Brexit, posing a number of significant challenges and opportunities for the UK and EU environment policies (Hepburn and Teytelboym, 2017).

However, Brexit also offers an opportunity to restructure policies that better-suit the UK's agenda under present circumstances when meeting existing targets for climate change. Agriculture in the UK has largely been influenced by EU legislation and the support of the CAP for the last 40 years (Wallace and Scott, 2017). Whilst it is important to maintain a high level of agricultural production, the efficiency of agricultural use comes into question, with direct payments through the CAP being available regardless of the level of production (Swinbank, 2017). As it stands, without the replacement of the existing climate-base policies, the UK may fall short in its ability to satisfy current domestic carbon targets. Therefore, it is likely that post-Brexit there will be a reduced level of taxpayer support for agriculture, instead focusing on meeting environmental objectives (Matthews, 2016). The formation of these updated policies and subsidies could lead to the disappearance of many small family-owned farms which rely on the CAP funding as part of their livelihoods. However, in replacement there may be the formation of subsidies which build upon the 2nd Pillar of the CAP, whereby additional payments to farmers could be made based on their ability to implement agri-environment schemes (Swinbank, 2017). Based on this notion, whilst there may be an initial shortfall in the number of small farms on 'marginal land' there is a unique opportunity to access additional funding through the provision of sustainable land management practices. Given this, quantifying the environmental potential of alternative land management is crucial in pushing forward the notion of environmental subsidies based on the provision of ecosystem services. This could spur an emergence in new forms of agriculture in the UK, allowing innovative solutions in tackling environmental policy to be implemented on a large-scale. Therefore, whilst the large-scale adoption of paludiculture may currently be out of reach, the additional funding from a post-Brexit UK could make re-wetting economically feasible for the land users who are set to currently lose out through changes in agricultural subsidies and the economic shortfalls of re-wetting the landscape.

5.5 Potential futures of study areas

It is important to identify how, despite the similarities in the locality and history of agricultural land use between the Alt Crossen and Lyth Valley, there are distinct differences in their overall condition and land use capability. The Alt Crossens is currently used for arable farming on 13,000 ha of high-grade agricultural land, dependent on drainage practices which cost £870k to maintain annually. In contrast, Lyth Valley predominantly consists of improved grassland on predominantly 3,000 ha of low-grade land, which cost £150k to maintain annually (JBA, 2015).

In an ideal world there would be a movement away from environmentally-damaging land management towards a sustainable use that remains highly productive and economically viable. This research identified how large areas of the Alt Crossens offer the poor provision of multiple ecosystem services, maximising agricultural production. It is clear that something needs to be done to improve the relative sustainability of the landscape, which coincides with the desired movement away from costly, non-essential land drainage. One benefit from the Alt Crossens study was the generation of a phased-approach in alleviating the economic strain of non-essential drainage and enabling arable farming to persist in low-risk areas whilst moving towards the desired levels of natural re-wetting. It may not be possible to encourage all stakeholders to adopt a paludiculture-based approach to coping with the desired movement away from land drainage, however without the formation of an IDB there is a serious risk of large-scale economic losses to the area. It is therefore crucial to encourage stakeholder discussion to create feasible solutions surrounding the future of the Alt Crossens with regards to meeting environmental commitments, reducing non-essential land drainage, and safeguarding livelihoods of land users in a post-Brexit climate.

With the case of Lyth Valley, given the low economic returns from the present land utilisation and the relatively poor status of agricultural land in the area, there is a much smaller case in safeguarding land drainage for conventional forms of agriculture. Not only this, but given the present conditions in Lyth Valley, the leap from a grassland-dominated landscape towards a future consisting of wetlands and wet grassland involves a much smaller transition in comparison to the Alt Crossens. This study signifies how a re-wetted landscape can improve the economic returns of the landscape, whilst simultaneously improving the provision of additional ecosystem services. Unless the establishment of a new IDB passes in the near-future, paludiculture offers a clear 'win-win' scenario for multiple stakeholders. As identified above, one of the barriers in implementing paludiculture on a large-scale is the lack of time-tested examples, without which the ability to ascertain funding and establish fully-functioning markets will be restricted. Lyth Valley therefore offers a unique opportunity introduce these practices within the UK as a viable solution to tackling the challenges faced by non-essential land drainage and environmental sustainability.

5.6 Limitations of LUCI and data

LUCI was designed to generate and represent ecosystem services to function as part of a decision-based framework for different stakeholders (Jackson et al., 2013). One of the benefits of this was the ability to represent values for different services in a fashion that is clear when providing objective evidence to stakeholders. As with every ecosystem service model, it is difficult to represent the fluxes and stores of natural capital in the form of algorithms. It should therefore be acknowledged that the reliability of generated results was limited by both the ability for each service tool to be accurately represented and the quality of the input data.

5.6.1 LUCI

Despite the Agricultural Productivity toolkit utilising biophysical thresholds for potential agricultural production, current agricultural production classifications are based solely on land cover, ranked based on their expected productivity to reflect the present levels of utilisation. Whilst this is useful to an extent, it was impossible to extract detailed information from calculated outputs without the introduction of additional information. Therefore, an economic appraisal was calculated to determine the agricultural returns of the landscape, improving the strength and detail of the analysis. Whilst finding a compatible agricultural productivity tool in ecosystem service models can prove difficult, this method was used to create a universal means of comparison in £ per hectare, as an example.

LUCI is a continually developing model that produces outputs for a number of ecosystem services. Whilst LUCI has the ability to cover a range of ecosystems on a field-level to a high level of detail, this analysis would benefit from the inclusion of cultural and recreational benefits. Both the Alt Crossens and Lyth Valley possess a number of sites which possess such benefits, such as Martin Mere and Foulshaw Moss. However, this research design hosted no capability to credit these interest areas and future work would benefit from the incorporation of these services from additional ecosystem service models.

Calculated values for carbon emission and sequestration were shown to be somewhat misleading in this analysis; indicative of the potential for additional carbon sequestration, not actual rates of carbon sequestration in tCO_2 ha^{-1} $year^{-1}$, which would prove more informative. LUCI possesses the ability to calculate rates of carbon sequestration through a supplied change in land cover. However, this aspect of the tool was not operational during this application, indicating how further work needs to be done to incorporate a means of calculating rates of sequestration for baseline and future scenarios. It is important to also acknowledge that there are other carbon fluxes associated with peatlands, such as waterborne dissolved organic carbon (DOC), particulate organic carbon (POC) through drainage channels, alongside the emission factors used to represent the release of additional greenhouse gases, such as methane (Evans et al., 2017). These fluxes contribute to a significant proportion of the carbon cycle within wetland areas, representing how LUCI adopts a singular approach when representing the multi-faceted nature of the carbon store. More comprehensive models are available to represent changes in the carbon stores and fluxes, indicating how other carbon-based tools to represent the stores and fluxes of carbon could supplement this analysis.

Emissions from low-lying peatlands in England provide a large, yet fairly uncertain contribution to the total UK carbon emissions. Given this, there is an increasing importance surrounding identifying conducting field measurements that satisfy the gap in knowledge for direct and indirect carbon emissions. Evans et al (2017) signifies how future LULUCF inventories (Tier 2) for the UK will include updated emission factors as well as DOC leaching, with a potential Tier 3 approach taking into account variations in the water table and fertiliser usage when calculating CO_2 and nitrous oxide (N_2O) emissions. Whilst these issues are beyond the scope of this thesis, it highlights how additional considerations should be made into the strength and reliability of the LUCI carbon model when establishing a comprehensive carbon inventory for real world landscape planning.

Whilst LUCI outputs provide a high level of spatial detail, there was a limited ability to account for the temporal changes in the fluxes of ecosystem services. Bain et al (2011) identify how the re-establishment of wetland conditions can take upwards of 50 years to introduce fully-functioning ecosystem services. In this time, the interactions between each ecosystem service, dependent on the forms of management adopted, are dynamic and difficult to represent. Instead, a ‘space for time’ substitution is applied, utilising steady-state values to represent the average annual change in fluxes over a 150-year period (Jackson et al., 2017). In the long-term this approach can reliably represent the changes in stores and fluxes, however it can be argued that LUCI over-represents these values in the short-term. Therefore, the ability to draw conclusive evidence from LULC changes over the short-term should be considered and compared against reported values before utilising before management decisions are implemented.

It is also worth mentioning that the ability to conduct such ecosystem service modelling was based on the accessibility of LUCI. The LUCI model was sub-licenced by research fellows from Victoria University, Wellington, to use a server-based version of the toolkit (still in its development stage) alongside a stand-alone version. This research was the first external application of the server-based platform in the UK, and significant delays were experienced during initial setup and trial runs for each ecosystem service tool. The more commonly used stand-alone version of LUCI was utilised to ensure there were no errors in the generated outputs.

5.6.2 Data

One of the apparent limitations in conducting the analysis for areas that are most at risk of waterlogging was the ability to utilise high-resolution Digital Elevation Model (DEM) data within ArcMap 10.4. LiDAR (Light Detection and Ranging) datasets are openly available to download and utilise for this basis of research, with elevation data available at resolutions of up to 0.5m^2. However, this limitation stems from the large differences in study area size, ranging from 3,000 ha in Lyth Valley to 13,000 ha in the Alt Crossens. For larger areas incomplete datasets alongside the large processing power require to operate at such resolutions have meant that LiDAR data was not viable for this application. Based on previous reports using LUCI, a 5m^2 resolution DEM was utilised to provide a relatively detailed, easy to use dataset (Jackson et al., 2016). However, the use of a DEM with a finer resolution would definitely improve the analysis on a smaller scale when pinpointing the presence of low-lying areas.

Another limitation was the level of information surrounding soil type and depth. The Soilscapes NATMAP dataset used in this study offers a complete breakdown of different soil types on a national scale. However, it was clear that there were some inaccuracies in the classifications for Lyth Valley. This fundamentally falls back to the lack of detailed soil maps available and highlights the benefit of local expert knowledge to improve the reliability in analyses. This also ties in with the lack of knowledge surrounding the depth of soil in the landscape, notably on peatlands. LUCI calculates the level of carbon storage for biomass and soil at a depth of 0.3m and 1.0m, however there is no clear understanding into the actual depth of these soils. For the Alt Crossens, an extensive peat depth survey was conducted in 1950 and 1980, however no further research has been conducted on as large a scale since, highlighting the need for updated and reliable soil depth data to understand rates of soil subsidence and peat wastage in the areas. Doing so would better-represent the fluxes and stores of ecosystem services in these landscapes.

LUCI is capable of modelling a variety of different land covers from several established land cover datasets. The 2015 Land Cover Map (LCM) provides high-resolution data for up to 21 Broad Habitat classes at a national scale; however, there is a limited ability to include additional classes compatible with the ecosystem service tools offered by LUCI. It should be acknowledged that paludiculture-based land covers are yet to be parameterised within LUCI, owing to an element of uncertainty in accepting the produced outputs. Given this, candidate areas for intervention through paludiculture, were instead assigned broader land covers accepted within previous applications of LUCI. Whilst much more detail could be added to improve the accuracy of the results, this was ultimately limited by both the LCM and capacity for LUCI to include additional classification in its calculations.

5.7 Overarching research goals

The aim of this thesis was to assess the current state of ecosystem services for two low-lying coastal catchments in the North West of England; each having previously been wetland areas and have since been altered for agricultural purposes. The following research objectives (below) will be taken into consideration to determine the success that each point has been satisfied within the study.

To identify the extent and influence of re-wetting, based on the desired movement away from non-essential land drainage and change in landscape conditions.

Utilising the same methodology proposed by JBA, Jacobs and ARUP in prior analysis of Water Management Units (WMUs) for the Alt Crossens (Environment Agency, 2010), it was possible to distinguish the Drainage Benefit Areas (DBAs) within both catchment systems. Doing so enabled the identification of areas surrounding water pumping stations and drainage channels which rely on artificial drainage to mitigate against the influence of surface flooding and high water tables. By doing so, it was possible to establish a variety of plausible re-wetting scenarios, enabling a ‘phased approach’ to be taken in the Alt Crossens, allowing decision-makers to analyse the influence of the planned withdrawal of conventional drainage practices. As Lyth Valley was a much smaller study size and relied on fewer pumping stations, only two scenarios were created to represent the baseline (drained) and future (fully-rewetted) conditions, involving the closure of all non-mandatory artificial drainage practices.

Adopt an ecosystem service-based modelling framework capable of assessing the provision of different services under a variety of drainage and LULC conditions.

A number of different ecosystem models were critiqued for their strengths and weaknesses prior to analysis; the Land Utilisation and Capability Indicator model (LUCI) was chosen as the most appropriate toolkit for creating an ecosystem service framework. Within this, it was deemed that carbon storage, carbon sequestration potential, flood mitigation, nitrogen loading, phosphorus loading and agricultural production were the most desirable variables that should be assessed. Utilising the different DBA scenarios, alongside LUCI’s built ability to generate outputs with/without natural drainage, the provision of each ecosystem service was assessed under baseline (drained) and re-wetted conditions. An economic appraisal was also produced, augmenting the agricultural production outputs to provide additional information for the value of agricultural land in both areas, alongside the relative economic value held in non-market ecosystem services. The ecosystem service framework was designed to work in a cyclic manner, allowing the user to complete multiple iterations to allow for further landscape optimisation as well as being capable of being used throughout a variety of catchments with different characteristics.

Assess the provision of multiple ecosystem services for each scenario, signifying where trade-offs and co-benefits between modelled services could be implemented.

By adopting the classified outputs for each ecosystem service tool, based on pre-determined biophysical thresholds, LUCI produces a number of additional map outputs which reflect the overall status of each service, ranging from ‘very good’ to ‘very poor’. Doing so enables the user to identify the possibility for co-benefits and trade-offs to exist, assigning weighted values to the status of each classification. The sum of multiple output layers, through weighted analysis, allowed an assessment into the potential for synergies to occur under future scenarios, or where maintenance of the *status quo* is better-suited. These outputs were parameterised against site visits and local expert knowledge to ensure that generated results were accurate and reliable against what is observable at present.

Introduce alternative forms of land use to candidate areas where present forms of land use would be unfeasible under re-wetted conditions, whilst simultaneously improving the provision of ecosystem services.

Utilising the map outputs for each ecosystem service, alongside the 'multiple ecosystem service assessment', candidate areas for intervention were highlighted to represent where the land use and land cover should be revised to better-suit future environmental conditions. Informed decisions were made surrounding the potential for alternative land management practices to be established within both sites to improve the overall status of multiple ecosystem services whilst better-suiting a re-wetted landscape. Given this, paludiculture-based land uses were implemented within the high-risk areas for Lyth Valley and the Alt Crossens and have shown that the overall provision of multiple ecosystem services had improved under re-wetted conditions (Tables 9 and 13). Whilst LUCI has not been directly parameterised for land covers involving paludiculture, a combination of 'peat bogs', 'broadleaf woodland' and 'neutral grassland' were deemed the most appropriate in reflecting the conditions of different forms of paludiculture.

Quantify the net gains and losses in the provision of different ecosystem services, based on a change in the land management strategies between baseline and future scenarios.

The generation of a variety of different scenarios for both sites enabled the quantification of the net gains and losses associated with land management decisions. Utilising the biophysical outputs in combination with the known area of DBAs at each study site enabled the quantification of different ecosystem services, whilst the area was used in the case of qualitative services (flood mitigation). The implementation of an economic appraisal further enhanced this aspect by relating a monetary value to land parcels and different land covers, quantifying the net economic gains and losses associated with alternative land management. This study shows how the provision of almost every single ecosystem service modelled in this application of LUCI was shown to improve in status under a re-wetted scenario when compared against the baseline (currently drained) scenario.

5.8 Conclusions

This research provides an ecosystem service-based approach in determining the re-wetted land use capability of future scenarios for the Alt Crossens and Lyth Valley, following a movement away from publicly-funded non-essential drainage. The growing agenda of meeting climate change targets encourages a focus towards more sustainable forms of land use, maximising Environmental Net Gain of natural capital stocks to satisfy these requirements. Ecosystem service modelling tools can provide powerful and informative decision-support outputs. Whilst there are a number of different models, each with their own strengths, this application of LUCI has shown a number of clear benefits in how the land use of both study catchments could be altered to realise the full potential of multiple ES provision, moving away from a 'single-use' landscape.

Under re-wetted conditions it was found that conventional agriculture experiences significant losses, particularly in the Alt Crossens. If an Internal Drainage Board is not formed in the near-future, to prevent a large-scale economic shortfall, alternative management options are required to safeguard agriculture in a re-wetted landscape. A range of management options were explored, each subject to a number of barriers in implementing chosen practices a large-scale. Paludiculture, the process of 'wetter farming', was the favoured management strategy, acting as an inclusive solution for all stakeholders. Within this, LUCI has shown how despite a decrease in agricultural productivity and economic returns in the Alt Crossens, there was a calculated increase in the provision of additional ecosystem services. Whereas, for Lyth Valley a 'win-win' situation was observed by the increased provision of ecosystem services alongside enhanced economic returns of the area.

A number of challenges face the large-scale adoption of paludiculture in the UK, namely the lack of funding to aid initial setup costs, due to the novel practice which is yet to make its mark in the UK. With the planned departure from the EU, there is a unique opportunity to restructure agricultural policy, directing subsidies towards agri-environment practices that will help the UK meet existing climate change and environmental targets, whilst encouraging a more efficient use of land for agricultural production. This presents the potential for Lyth Valley to accommodate trial sites that could show different stakeholders that paludiculture has a place within the UK economy. However, one of the clear barriers in implementing any form of environmental improvement is the desire for change, which should be influenced by social learning and provision of case studies to represent the potential benefits beyond the realm of agricultural production. Whilst it was not possible to simulate in this study, there is clear evidence from present examples that cultural and recreational benefits are offered by re-wetting, an area which is particularly prominent in Lyth Valley.

There is no straightforward solution to the challenges of re-wetted land management in both areas, due to the complex social, environmental, economic and political factors which influence the decisions of different stakeholders. This research has provided a framework for an ecosystem service-based approach in assessing how UK landscapes could be managed in a more sustainable manner. As with all ecosystem service models, there is a level uncertainty in the generated results, however this research has highlighted each limitation in detail with comments on the future application of this framework for future study.

5.8 Future research

One of the areas that could be built upon in future research is the hydrological function of the land in ascertaining the extent that drainage features influence the landscape. A combination of approaches was utilised in identifying these areas most at risk of flooding; principally, the presence of Drainage Benefit Areas and/or Water Management Units alongside the location of pumping stations, extensive drainage networks and the elevation of the land. A hydrological survey was conducted prior to this study for the Alt Crossens, which would prove valuable in determining where areas are likely to experience surface flooding following a large precipitation event. In this sense, future research could involve flagging areas which are most prone to waterlogging using real-time data and surveys of the seasonal fluctuation in the water table.

It would also prove useful to run the data inputs through different ecosystem service models to gauge the effectiveness and capabilities in contrast to LUCI. This research utilised LUCI based on a review of a number of ecosystem service models; however, other toolkits such as InVEST and ARIES also possess significant capabilities that should be explored to assess how LUCI performs in comparison.

There is a clear need to attach economic values to different land covers and the ecosystem services provided. Whilst this research has linked the gross market values of different crops and ecosystem services to the chosen methods of land use, this is only part of the picture. A significant number of UK farmers rely on the Common Agricultural Policy alongside other subsidies, thus meaning that agricultural land may be undervalued in this research. By including information surrounding the number of subsidies available through future environmental policy, the economic value of the land could be improved. This concept works together with the ability to value other ecosystem services; using projected values for traded carbon values (2019 = £26.30 per tCO_2e), enabling users to apply a monetary value to carbon savings associated with alternative forms of land use (BIES, 2019). Whilst it was found that sphagnum farming provided an economic return of ~£312 per ha annually (Wichtmann et al., 2016), the cultivation of sphagnum was found to sequester an additional 15 tCO_2e ha^{-1} $year^{-1}$ compared to improved grassland practices. This could account for an additional £394 ha^{-1} $year^{-1}$, alongside other ecosystem service benefits, which could be used to subsidise the land user for better practices of environmental sustainability in agriculture.

To truly assess the reliability of the outputs it would be beneficial to run a number of workshops with different stakeholders to discuss and critique the findings of each scenario, acting as a method of ground-truthing the application of LUCI in the focus areas (Jackson et al., 2013). This could coincide with speaking to members of the National Farmers Union (NFU) alongside members of Internal Drainage Board action groups, to discuss the viability of closing different pumping stations and the options that land users and farmers may have in coping with re-wetted conditions. Whilst this was beyond the scope and length of this study, it would prove extremely valuable to both decision-makers and stakeholders in assessing the viability of implementing change within the context of low-lying rural areas.

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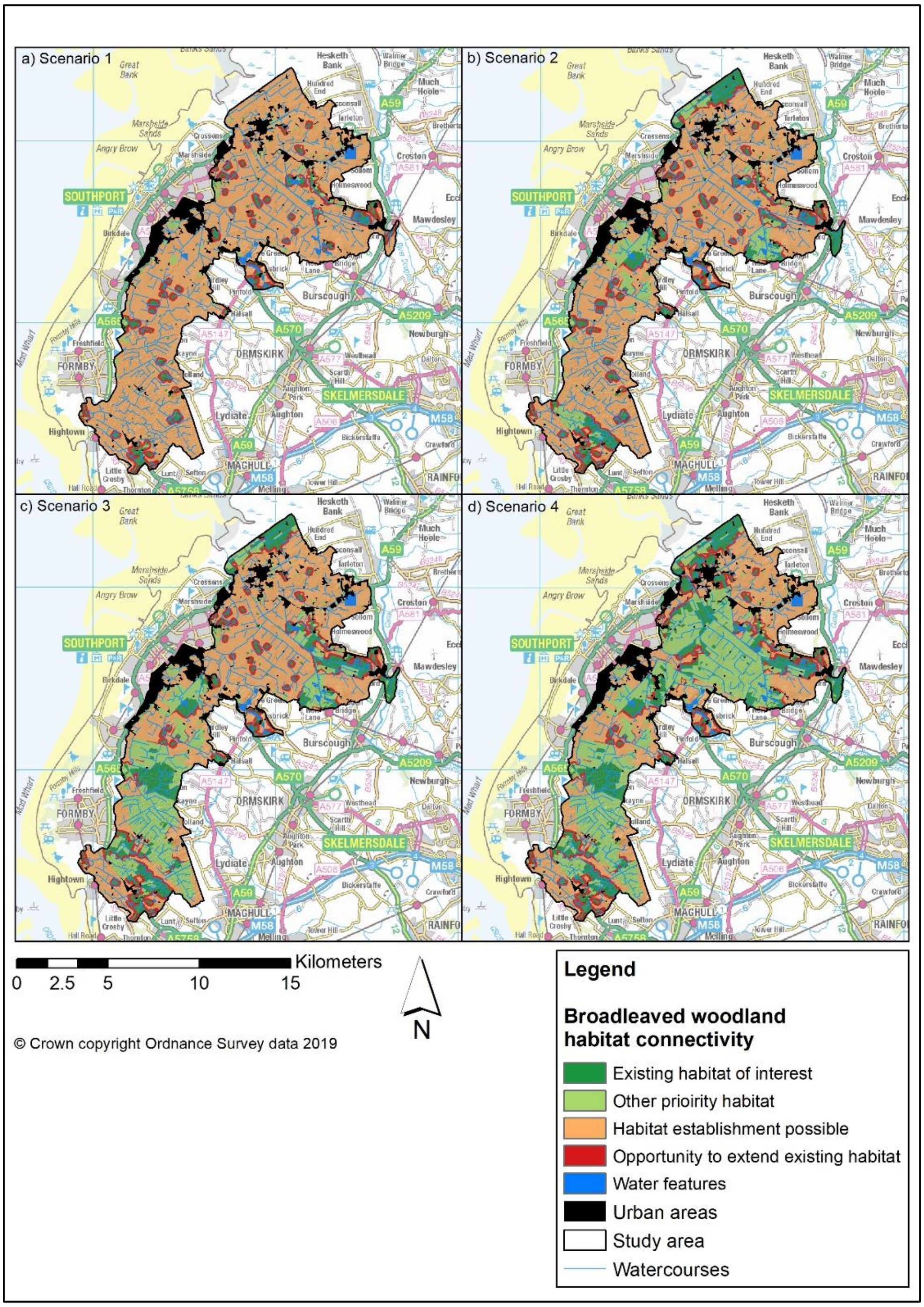
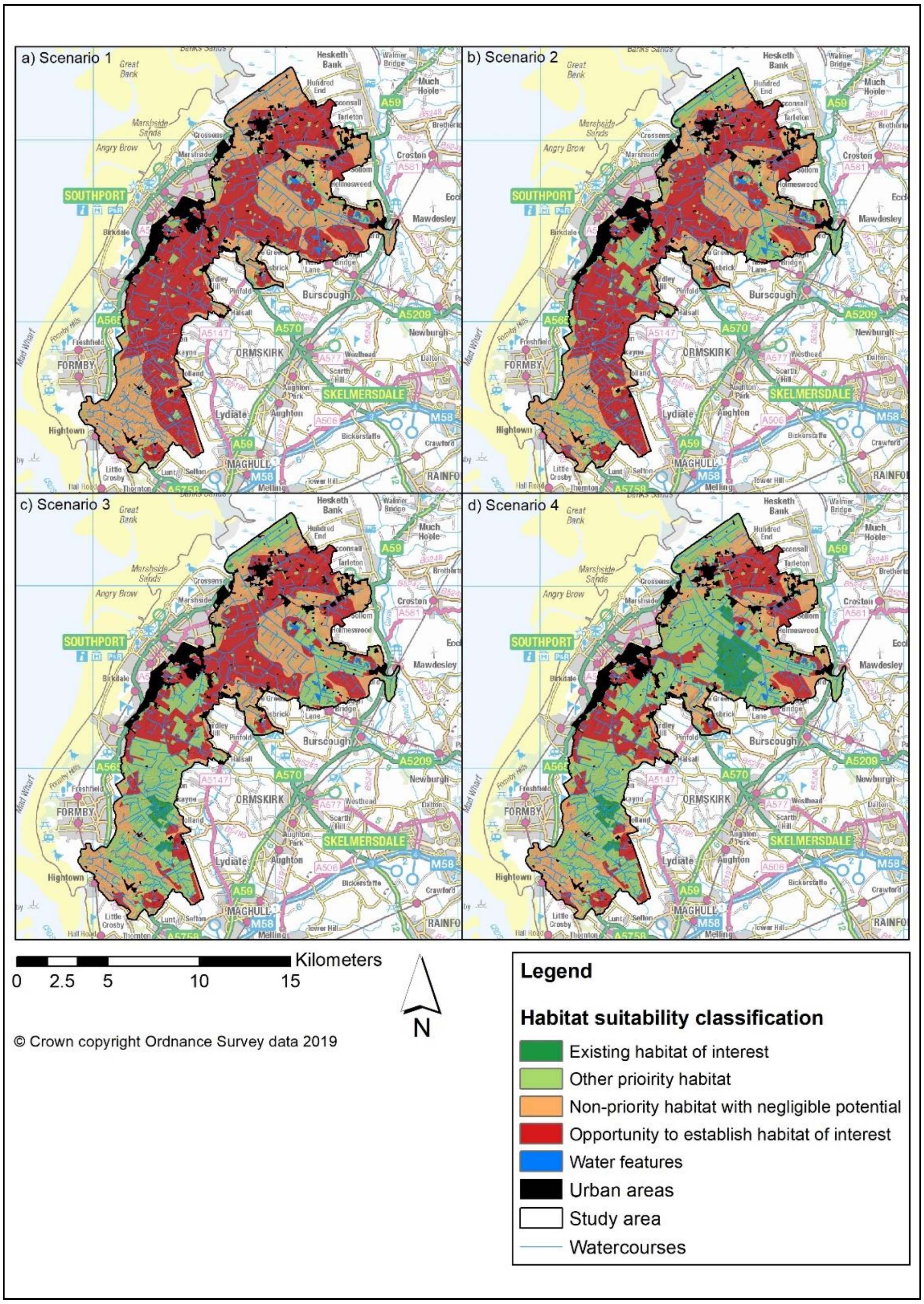
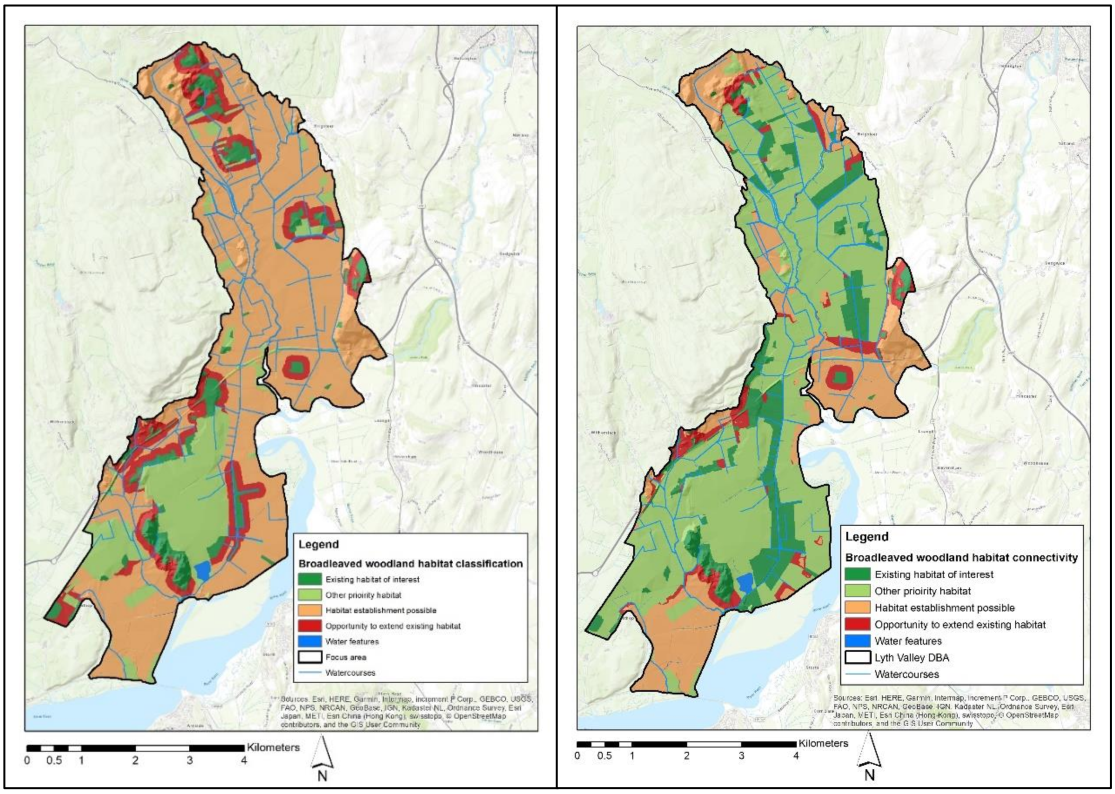
Appendix A: Biodiversity figures



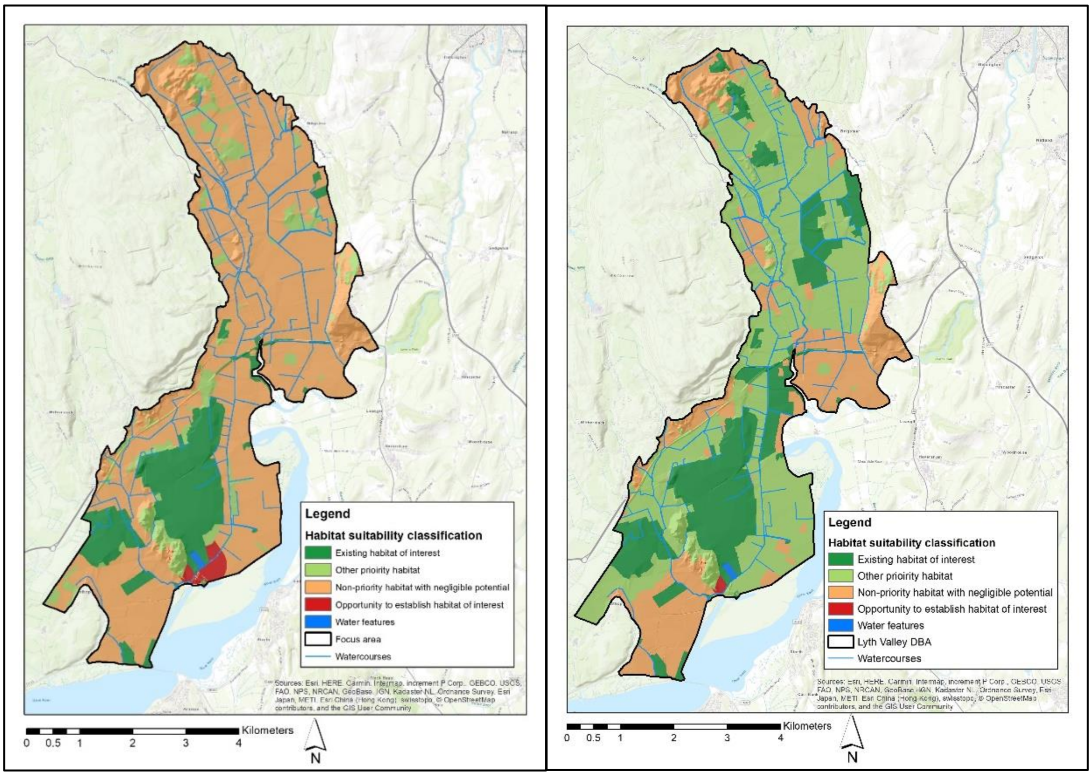
Appendix A. i: Broadleaf woodland habitat connectivity for Scenarios 1-4 in the Alt Crossens.



Appendix A. ii: Wetland and wet grassland habitat suitability for Scenarios 1-4 in the Alt Crossens.



Appendix A. iii: Broadleaf woodland habitat connectivity for Baseline and Re-wetted Scenarios in Lyth Valley.



Appendix A. iii: Wetland and wet grassland habitat suitability for Baseline and Re-wetted Scenarios in Lyth Valley.

Appendix B: Alt Crossens summary tables

Appendix B. i: Calculated levels of agricultural productivity for Scenarios 1-4 in the Alt Crossens.

Agricultural productivity	Scenario 1 % cover	Scenario 2 % cover	Scenario 3 % cover	Scenario 4 % cover
Very high	76.5	61.6	45.3	27.4
High	6.9	10.3	12.0	13.3
Moderate	2.8	9.1	19.2	28.3
Marginal	3.7	8.5	11.7	14.6
No production	8.3	8.2	9.6	14.1

Appendix B. ii: Classified carbon emissions for Scenarios 1-4 in the Alt Crossens.

Classified carbon emissions	**Scenario 1 % cover**	**Scenario 2 % cover**	**Scenario 3 % cover**	**Scenario 4 % cover**
High sequestration	9.7	10.9	10.6	12.2
Moderate sequestration	2.2	4.3	4.4	5.2
Minimal sequestration	17.1	15.2	22.9	13.6
Moderate emissions	27.8	27.4	28.7	25.7
High emissions	33.2	33.9	33.4	35.0

Appendix B. iii: Presence of flood mitigating features for Scenarios 1-4 in the Alt Crossens.

Flood mitigation	**Scenario 1 % cover**	**Scenario 2 % cover**	**Scenario 3 % cover**	**Scenario 4 % cover**
Mitigating features	3.9	8.6	13.2	20.6
Mitigated features	7.8	6.6	7.8	9.5
Non-mitigating features	86.7	82.6	76.9	67.8

Appendix B. iv: Broadleaf woodland habitat connectivity for Scenarios 1-4 in the Alt Crossens.

Habitat connectivity	**Scenario 1 % cover**	**Scenario 2 % cover**	**Scenario 3 % cover**	**Scenario 4 % cover**
Existing	3.7	8.4	11.6	14.6
Other priority	2.9	9.2	20.7	34.3
Establishment possible	83.6	70.0	55.4	38.6
Opportunity to extend	8.0	10.1	10.0	10.3

Appendix B. v: Wetland and wet grassland habitat suitability for Scenarios 1-4 in the Alt Crossens.

Habitat suitability	Scenario 1 % cover	Scenario 2 % cover	Scenario 3 % cover	Scenario 4 % cover
Existing	0.1	0.1	1.5	6.1
Other priority	6.6	17.9	31.4	43.6
Non-priority	41.7	35.3	31.9	25.6
Opportunity to extend	51.2	46.3	34.7	24.3

Appendix B. vi: Provision of multiple ecosystem services for Scenarios 1-4 in the Alt Crossens.

Multiple ecosystem services	Scenario 1 % cover	Scenario 2 % cover	Scenario 3 % cover	Scenario 4 % cover
Excellent status	9.7	10.9	10.6	12.2
Good status	2.2	4.3	4.4	5.2
Moderate status	17.1	15.2	22.9	13.6
Poor status	27.8	27.4	28.7	25.7
Very poor	33.2	33.9	33.4	35.0

Appendix C: Lyth Valley Summary tables

Appendix C. i: Calculated levels of agricultural productivity for Baseline and Re-wetted Scenarios Lyth Valley.

Agricultural productivity	**Scenario 1 % cover**	**Scenario 2 % cover**
Very high	1.6	0.6
High	71.2	23.9
Moderate	1.9	34.2
Marginal	7.6	16.9
No production	16.8	22.9

Appendix C. ii: Classified carbon emissions for Baseline and Re-wetted Scenarios Lyth Valley.

Classified carbon emissions	**Scenario 1 % cover**	**Scenario 2 % cover**
High sequestration	2.7	2.1
Moderate sequestration	5.6	5.5
Minimal sequestration	26.8	34.5
Moderate emissions	62.3	55.3
High emissions	0.0	0.0

Appendix C. iii: Presence of flood mitigating features for Baseline and Re-wetted Scenarios Lyth Valley.

Flood mitigation	**Scenario 1 % cover**	**Scenario 2 % cover**
Mitigating features	21.7	37.2
Mitigated features	6.5	9.6
Non-mitigating features	70.4	51.9

Appendix C. iv: Broadleaf woodland habitat connectivity for Baseline and Re-wetted Scenarios Lyth Valley.

Habitat connectivity	**Scenario 1 % cover**	**Scenario 2 % cover**
Existing	7.0	14.8
Other priority	16.5	9.0
Establishment possible	62.7	75.0
Opportunity to extend	14.5	1.1

Appendix C. v: Wetland and wet grassland habitat suitability for Baseline and Re-wetted Scenarios Lyth Valley.

Habitat suitability	**Scenario 1 % cover**	**Scenario 2 % cover**
Existing	14.1	22.1
Other priority	56.0	48.9
Non-priority	22.2	28.6
Opportunity to extend	6.3	0.2

Appendix C. vi: Provision of multiple ecosystem services for Baseline and Re-wetted Scenarios Lyth Valley.

Multiple ecosystem services	**Scenario 1 % cover**	**Scenario 2 % cover**
Excellent status	1.5	1.6
Good status	18.9	26.8
Moderate status	20.4	26.8
Poor status	58.8	44.0
Very poor	0.4	0.4