Incorporating ecological networks and green infrastructure into spatial strategies – mapping optimal locations for habitat banks

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Abstract

Negative landscape change and impacts on biodiversity, as a consequence of development, must be mitigated for. In the UK mitigation is carried out through a variety of policy and planning instruments whose delivery measures are often piecemeal, thus restricting their ability to address the cumulative landscape impacts of multiple developments. One proposed response is the adoption of a type of mitigation banking – "habitat banking", where the creation, management or restoration of habitats is funded by the purchase of credits by the developer. This research proposes that spatial targeting of habitat bank locations is crucial to fully mitigate development impacts whilst maximising landscape function benefits.

A landscape scale modelling approach was developed for a case study area in the South Midlands to investigate spatial targeting of habitat banks, with ecological networks and ecoprofiles employed to guide their location. Ecological effects of bank composition, size and location were examined, combined with a network analysis to determine people's access to natural greenspace in the same area, and subsequently compared against the current selection mechanism - Accessible Natural Greenspace Standards (ANGSt). Changes resulting from human population growth and habitat bank introduction were examined in association with impacts of climate change on potential bank locations over a 50 year period, whilst a chain of climate envelope, dispersal and colonisation models determined the ability of ecoprofiles to keep pace with climate changes. The ability of habitat banks to contribute to landscape functionality was determined both spatially and temporally.

Habitat banks identified by the models increased the existing ecological network size by a factor of up to 2.72:1 and were able to deliver the majority of habitat creation targets set out in regional biodiversity action plans (BAPs). 100% of wetland, unimproved grassland and broadleaf and mixed woodland creation targets were met, whilst only 75% of the lowland heath target could be achieved. Multi-habitat banks of over 3700 ha were identified with such areas determined to be of importance in achieving landscape improvements for a wide range of species. Although ANGSt targets were not met, habitat bank locations did increase overall greenspace accessibility for over 3000 people. The ability of ecoprofiles to track climate change was directly related to both the area and connectivity of habitat patches, with the broadleaf woodland ecoprofile being the most capable of adapting to predicted climate change. Habitat banks contributed to increased landscape functionality in the short-term but predicted climate change impacts become insurmountable in the medium to long-term, drawing into question the long-term viability of the landscape in its current state to withstand potential climate changes.

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Abbreviations and Acronyms

ANGSt	Accessible Natural Greenspace Standards
ANN	Artificial Neural Network
BAP	Biodiversity Action Plan
BRANCH	Biodiversity Requires Adaptation in NW Europe under a Changing Climate
CEH	Centre for Ecology and Hydrology
CART	Classification and Regression Tree Analysis
CIL	Community Infrastructure Levy
CLERE	Community Landscape Ecology Recreation Economy Model
CQC	Countryside Quality Counts
DDD	density dependent dispersal
DID	density independent dispersal
DTLR	Department for Transport, Local Government and the Regions
DCLG	Department of Communities and Local Government
DETR	Department of the Environment, Transport and the Regions
EVC	Ecological Vegetation Class
EIA	Environmental Impact Assessment
EEA	European Environment Agency
GAM	Generalised Additive Model
GLM	Generalised Linear Model
GIS	Geographic Information System
GCM	Global Climate Model
HLF	Heritage Lottery Fund
IPCC	Intergovernmental Panel on Climate Change
JCA	Joint Character Area
LBAP	Local Biodiversity Action Plan
LDV	Local Delivery Vehicle
LNR	Local Nature Reserve
LPA	Local Planning Authority
MKSM	Milton Keynes and South Midlands Growth Area
MONARCH	Modelling Natural Resource Responses to Climate Change
NNR	National Nature Reserve
ODPM	Office of Deputy Prime Minister
OS	Ordnance Survey
OA	Output Area
PPG	Planning Policy Guidance
PPS	Planning Policy Statement
PAWS	Planted Ancient Woodland Sites
ROW	Public Right of Way
RU	reproductive unit
S106	Section 106 Agreement
SSSI	Site of Special Scientific Interest
SPECIES	Spatial Evaluator of Climate Impacts on the Envelope of Species
SAC	Special Area of Conservation

SPA	Special Protection Area
SRES	Special Report on Emissions Scenarios
SEA	Strategic Environmental Assessment
SPD	Supplementary Planning Document
UKCIP	UK Climate Impacts Programme
WCED	World Commission on Environment and Development

Contents

Figures

Tables

CHAPTER ONE | Introduction

1.0 Introduction

> Landscape scale planning practice is becoming increasingly systematic, and is no longer concerned with simply collecting a select number of protected sites. Far clearer strategies are emerging related to the condition, character, functionality and vibrancy of landscapes, and the appropriate balances between conservation, creation, strengthening or restoration which are necessary to address tendencies towards dysfunction and obsolescence. (Selman, 2006)

The narrative of landscape change reflects the strong anthropogenic influence of both past and current generations. The desire for rapid socio-economic development has led to a dramatic level of urbanisation, the spatial extent, nature and permanence of which has resulted in striking impacts on environmental and particularly ecological systems (Noon and Dale, 2002). However, conservation efforts have often focused on maintaining biodiversity by establishing protected areas and so minimising exposure to human activities (White *et al.*, 1997). Such an approach should only be seen as part of a solution in the mitigation of landscape change as it does not take account of the complex interaction of pattern, process, structure and function inherent within landscapes (Bürgi *et al.*, 2004). There is an explicit need to consider how landscapes function as a whole, the specific issues which exist currently (fragmented habitats, declines in species distribution and abundance), and which are likely to occur in the future (changes in climatic suitability, habitat connectivity and probable species range shifts). This dynamic nature of landscape allows it be considered as an interactive equation, ceaselessly making and remaking itself through processes of continuous and discontinuous change (Wood and Handley, 2001). Identifying drivers of change which encapsulate these influential processes allows the evolutionary trajectory of the landscape to be observed and influenced (Bürgi *et al.*, 2004). Understanding what is happening, why, and what might be done to resolve the situation is required for a structured landscape planning response to be developed (Wood and Handley, 2001).

The prediction of future impacts on landscape systems must also be explicitly considered. Climate changes will undoubtedly lead to an alteration of the way landscapes function, be that through changes in phenology, distribution of species, composition of habitats or experiences of recreational users of the landscape (Gill *et al.*, 2008; Hannah *et al.*, 2002; Parmesan and Yohe, 2003). Opdam and Wascher (2004) contend that, in the face of climate change, landscape strategies should re-examine their focus and direction as follows:

i. Replace a species orientated focus with one based on landscape conditions required for biodiversity.
ii. Focus on landscape networks including protected areas, connecting zones and intermediate landscapes instead of remaining focused only on protected areas.
iii. Move away from a defensive conservation strategy, accepting that biodiversity is only effective if we integrate it in the dynamic development of the landscape.

If it is accepted that landscapes are inextricably bound to anthropogenic influences it is clear that landscape change cannot be considered from a compartmentalised perspective. Drivers of change are multi-sectoral and therefore consideration of issues and responses must match this. The use of a multifunctional concept, *i.e. green infrastructure* which is examined in more detail in the proceeding chapter, provides a practical framework for the examination of interactions between landscape functions. It is clear, however, that multifunctionality is effectively an anthropogenic perspective with the term functions often used interchangeably with services. To consider that the development of a landscape strategy is anything other than a human-centric method of landscape management (Opdam and Wascher, 2004), even where environmental values are given high regard, is, however naïve. These originating principles of landscape strategies, however, do not preclude such plans from achieving considerable environmental benefits, *e.g.* Natura 2000 and UK Biodiversity Action Plan (BAP) have both contributed significantly to the retention and protection of biodiversity in the UK.

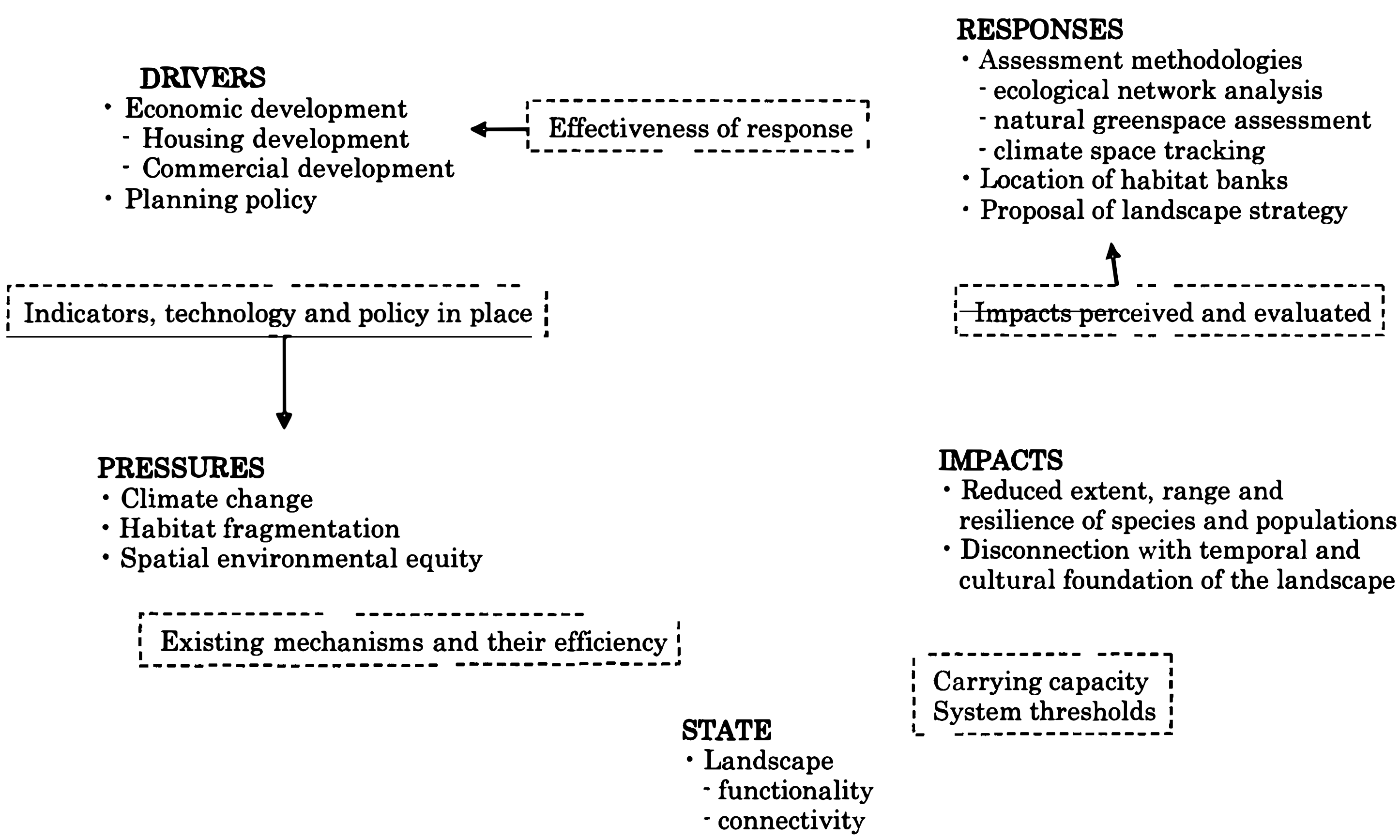
The legislative framework which underpins the protection of landscape functions is wide ranging in both scale and scope. That, explicitly linked to large scale built development and its ensuing detrimental impacts, includes at the highest level

Environmental Impact Assessment (EIA) (Council Directive, 1997), Strategic Environmental Assessment (SEA) (Council Directive, 2001), Habitats (Council Directive, 1992) and Birds Directives (Council Directive, 1979) and the International Convention on Biological Diversity (Department of the Environment, 1994; UK Biodiversity Steering Group, 1995). However, the ability of such mechanisms to take into account and implement the mitigation action required when development takes place has been questioned in a submission to the Royal Commission on Environmental Pollution inquiry on Environmental Planning (Treweek, 2000; Wende *et al.*, 2005). Current approaches are particularly ill suited to address landscape attrition processes, *i.e.* where cumulative impacts of development lead to habitat fragmentation and small scale habitat loss (Treweek, 2000; Treweek and Thompson, 1997). The proposed response to these issues is the adoption of a type of mitigation banking, *i.e.* where actions to restore or recreate habitat occurs prior to development, therefore compensating for forthcoming displacement (Crooks and Ledoux, 2000). The approach would allow developers to purchase credits in habitat banks commensurate with the impacts of their developments, with the money being used to fund creation and management of the ecological resource. This ability to pool resources from a number of developments would allow large ecologically robust sites to be created whilst delivering added value through the provision of landscapes people could visit and enjoy (Gillespie and Hill, 2007). The development of such mechanisms in the USA, Australia, South Africa, Germany and the Netherlands provides considerable experience to draw upon.

The largest multifunctional gains possible through habitat banking require spatial targeting of banks. The existing use and state of the landscape from a social and environmental perspective alongside the predicted impacts of developments must be analysed and an appropriate strategy sought (Briggs *et al.*, 2009). A systematic examination of existing and aspirational landscape functions is required at a regional or sub-regional scale in order to propose a methodology which could be used to locate habitat banks. This study is concerned with the use of habitat banks as a multifunctional mechanism to counteract and compensate for the detrimental impacts of large scale and prolonged built development in the South Midlands, UK.

1.1 Structuring a landscape planning response

From the outset, development of a landscape planning strategy requires the specification of a number of aspects: the drivers of landscape change (political and physical), identification of external pressures, delineation of the landscape in question (spatially, structurally and functionally), and the impacts of changes (profound and incremental). Examined schematically (Figure 1-1), interactions between stages, alongside transitional considerations can be identified allowing an envisioning of the entire landscape strategy process.



Adapted from Smeets & Wetering (1999)

Figure 1-1 The key stages and processes in development of a structured landscape strategy

1.2 Aims and Objectives

The research is structured around three specific aims:

1. To examine habitat banking as a practical response to habitat fragmentation and the degradation of functional landscapes.

2. To determine the nature of green infrastructure expansions required in order to compensate for landscape changes and to address future requirements.
3. To develop a methodology which determines optimal solutions for habitat banking and wider green infrastructure location and development.

Five objectives have been identified in order to carry out the aims:

a. Determine quantitatively and qualitatively the current functional status of the study landscape.
b. Define how future scenarios, *e.g.* development and climatic changes will impact on landscape functionality and green infrastructure in the study landscape.
c. Develop a GIS approach to determine landscape connectivity and accessibility to green infrastructure.
d. Create a GIS based spatial targeting approach which can be used to develop and examine consequences of habitat bank scenarios.
e. Propose a suitable strategy which would enable retention and safeguarding of green infrastructure and landscape functionality in the study landscape.

CHAPTER TWO | Review

Research Approaches to Landscape Change and Policy

2.0 Introduction

Landscapes are dynamic rather than static systems, material and territorial entities and at the same time a way of seeing, using and perceiving the physical environment (Qviström and Saltzman, 2006). Whilst it is true that changes in the landscape are both natural and human induced it should be acknowledged that man's detrimental influence on, and exacerbation of, landscape change continues to increase. The future of most landscapes increasingly is being determined by human activities (Miller and Hobbs, 2002). Socio-economic processes are the primary drivers for land use and land cover change which in turn determine the structure, function and dynamics of most landscapes (Gutzwiller, 2002). These activities modify existing landscape patterns and processes, either deliberately or inadvertently; determining which is essential for rational land use planning and management (Hobbs, 1997). A key issue then, which must be addressed by an interdisciplinary approach, is the impact of man on the landscape, alongside an understanding that nature has a function for society in that it provides an intrinsic value for recreation and scenic beauty, a value people desire (Jongman, 2005). However, changes which occur in the landscape are significantly influenced by policy and strategies and therefore scenarios examined or proposals for change must be in conformity with agreed strategic priorities and timescales.

2.1 Land use planning

Land use planning when required in order to regulate and guide development or impending development needs to be approached from a perspective that ensures economic gain yet allows the retention of intact natural systems. The practice of land use planning is not a neatly definable task but involves mediating decisions through the imaginative deployment of a repertoire of complementary planning instruments in pursuit of multiple objectives across numerous terrains and networks (Selman, 2006). This perspective has become acknowledged and embedded in planning policy, identifiable through the occurrence over recent decades of such mechanisms as EIA, SEA, planning regulations, Section 106 Agreements, landscape character assessment and, more latterly,compensatory Supplementary Planning Documents (SPDs), roof tariffs and the Community Infrastructure Levy (CIL). Use of these mechanisms recognises that development is necessary and aims to ensure that matters affected by such development are addressed: green infrastructure, socio-cultural landscapes, spatial environmental equity and biodiversity. However, in order to contribute to landscape planning in a considered and holistic way there is a need to establish an explicit link between cause, effect and response.

2.1.1 Multifunctionality

Multifunctionality specifies that an activity may have multiple outputs and effects, potentially contributing to numerous societal objectives at once (OECD, 2001). Identifying non-commodity benefits inherent in the landscape, and which are affected by landscape change, *e.g.* land conservation, sustainable management of renewable natural resources, conservation of biodiversity and the socio-economic viability of rural areas, forms the basis of a multifunctional perspective. The use of multifunctionality as a way of quantifying landscape changes allows cause, effect and response to be correlated and holistic strategies to be developed. This approach to landscape planning can be identified in mechanisms such as EIA and SEA, although their consideration of impacts remains compartmentalised. A true integration of economic, social and environmental factors, however, is required in order that landscape plans can be achieved which correspond to the potentials of the natural physical structure whilst allowing development to occur (van Buuren and Kerkstra, 1993). Green infrastructure has been proposed as a systematic framework for examining landscape multifunctionality where planning for an area's green infrastructure allows conservation priorities to be evaluated (McDonald *et al.*, 2004). Such a systematic approach requires an ability to quantify impacts and responses and to be able to develop methods of compensation and mitigation. In fact, it is suggested that any green infrastructure project that seeks to improve ecological function firstly needs to define functions, evaluate current states and *only then* identify areas to maintain, enhance and restore (Catchpole, 2006).

2.1.2 Planning responses to land use change

Planning the mitigation of development impacts is therefore a complex process which must take into account tangible physical changes alongside more subtle functional interactions. Environmental mitigation encompasses a broad range of landscape conservation strategies from development avoidance to out-of-kind habitat replacement. Within Europe the EU Directives on EIA (85/337/EEC and 97/11/EC) (Council Directive, 1985; Council Directive, 1997), are the foundations of mitigation and compensation approaches (Glasson *et al.*, 1999). Mitigation is often used to mean minimization, such as limiting or reducing the degree, extent, magnitude or duration of adverse impacts. This can be achieved by scaling down, relocating or redesigning elements of the project (Rundcrantz and Skärbäck, 2003). The potential of mitigation has now expanded and there is a willingness to achieve no-net-loss for the impact of development rather than an acceptance of some level of residual negative impact. This entails ecological damage resulting from human intervention being compensated for financially with the funds being used for the benefits of nature and the environment (van Bohemen, 1998). Such approaches explicitly acknowledge the necessity of human alteration of the landscape but seek to ensure retention of quality and quantity of the natural environment. Compensatory mitigation can be thought of in two ways: restoration compensation – environmental compensation for lost environmental values in the right functional context (on-site, in-kind compensation) and replacement compensation – environmental compensation for lost environmental values implemented in another functional context (off-site and/or out-of-kind compensation) (Rundcrantz and Skärbäck, 2003).

2.1.3 Compensatory mitigation, biodiversity offsets and habitat banks

Approaches to compensatory mitigation have existed since EIA came into use, however, progress is required in order to address inadequacies of current approaches, *e.g.* enforcement of compensation impacts, and to take account of landscapes affected in a multifunctional way (Wende *et al.*, 2005). EIA is a systematic process that is used to predict and examine the environmental consequences of proposed development actions in advance: individual projects, groups of related projects or government policies (Treweek, 1996). The emphasis, compared with many other mechanisms for environmental protection, is on prevention (Glasson *et al.*, 1999). The stepped approach undertaken in an EIA ensures that alternatives and potential impacts are identified in the pre-development phase. To ensure entire 'families' of projects, programmes, plans and policies are assessed simultaneously, and therefore ensuring cumulative impacts are taken into account, SEA has been developed and put in place through a European Directive (2001/42/EC) (Council Directive, 2001). The process used incorporates an evidence based and systematic decision support process set within a structured framework (Fischer, 2007). SEA corresponds well with large scale landscape planning and anticipated landscape change, where the procedures can illuminate the connections between land use planning policies and environmental change, highlighting potential impacts of plan implementation (Jones *et al.*, 2005). However, numerous shortcomings have been highlighted in these current strategic level systems:

i) failure to analyse impacts beyond development site boundaries,
ii) failure to quantify ecological impacts (vague descriptive predictions are the norm),
iii) failure to identify or measure cumulative ecological effects,
iv) failure to mitigate important ecological impacts (proposed mitigation measures are inappropriate and implementation is not mandatory), and
v) lack of monitoring or follow up (actual outcomes are not known and no corrective action can be taken in the event of mitigation failure).

(Treweek, 2000)

At the local scale, planning applications, when submitted, should be in accordance with the policies in local development plans. In addition, conditions are often placed on planning applications in order for them to meet these policies more effectively. Planning conditions must be both necessary and reasonable as well as enforceable, precise and relevant to both planning and to the development to be permitted (Department of the Environment, 1995). A commonly utilised linked mechanism for gaining environmental compensation is the Section 106 Agreement

(Acts of Parliament, 1990). Here there is a requirement for developers to provide specified environmental improvements in order to mitigate the development or in compensation for resources lost as a result of the development. This is usually operated as a single agreement between a developer, other relevant organisations and the local authority. If the Section 106 Agreement is in order to replicate lost habitat then biodiversity targeting is certainly required in order that the habitat created is a suitable replacement or alternative. However, such agreements are usually site specific and therefore are only able to contribute in a piecemeal way to overall landscape functionality. A progression of the Section 106 Agreement has recently been developed in the Milton Keynes and South Midlands (MKSM) growth area – a roof tariff. This roof tariff required a 'tax' to be paid for each new dwelling, to create a fund for infrastructure provision including green infrastructure (Gillman, 2006; Milne, 2005). For the first time this approach allows a pooling of resources which can contribute to the impacts of development in a comprehensive rather than piecemeal way, ensuring impacts can be properly addressed (Association of London Government, 2004).

A national Community Infrastructure Levy (CIL) (Acts of Parliament, 2008) with a financial framework similar to the Milton Keynes roof tariff principle currently is being introduced (Community Infrastructure Levy Regulations, 2010; DCLG, 2009a). This will enable local authorities to require developers to contribute to the increased infrastructure required as a result of development and associated new communities (DCLG, 2008a). This simplified approach will allow charges based on simple formulae to be developed in order for local authorities to achieve infrastructure requirements within and appropriate to their areas and provide certainty over contributions for developers (Acts of Parliament, 2008; Community Infrastructure Levy Regulations, 2010). Infrastructure is defined in broad terms incorporating social and environmental infrastructure such as schools and parks, and therefore fits within obligations set out in Planning Policy Statement 12 (DCLG, 2008b).

It is widely acknowledged (Gillespie and Hill, 2007; Hill, 2006; Treweek, 2000; Treweek and Thompson, 1997), that current mechanisms and approaches to deal with fragmentation and habitat loss as a result of development are not adequate and an alternative approach is required. The proposal put forward is that of mitigation banking, an approach that operates by developers offsetting adverse effects and habitat loss from developments by the purchase of credits in a habitat 'bank' with a pre-determined nature conservation value. Through accumulation and strategic targeting of funds larger scale and higher quality habitat creation and enhancement can be achieved. This approach is consistent with both PPS12 and the CIL. Treweek identifies seven key benefits of the mitigation banking system:

i) nature conservation trade-offs are explicit;

ii) limited resources can be invested in places where benefits will be maximised;

iii) time and money can be saved by investing in fewer, larger mitigation projects;

iv) all other things being equal, larger nature reserves are more valuable for wildlife than smaller ones;

v) economies of scale in management, and often more options for supporting a range of habitats and species;

vi) funds are secured for long-term management;

vii) conservation and mitigation work is undertaken by suitably qualified professionals.

(Treweek, 2000)

There appears to be a growing interest in such market mechanisms for trading biodiversity credits and to achieve environmental goals (EEA, 2006; Madsen *et al.*, 2010; Treweek *et al.*, 2009). Experience overseas suggests that imaginative approaches to ecological mitigation can have considerable benefits for nature conservation and can streamline the mitigation process making it more straightforward for both developers and planners (Treweek, 2000). Indeed, there is significant evidence from the USA, South Africa, Australia and Northern Europe; Germany (Wende *et al.*, 2005; Wilding and Raemaekers, 2000), and the Netherlands (Cuperus *et al.*, 1996; Cuperus *et al.*, 1999) that mitigation banking has the potential to address many of the issues raised with the current system.

2.1.4 Mitigation banks

Mitigation banking is essentially a focused form of ecological restoration whereby the impacts can be readily quantified through the EIA, SEA or general planning systems. In many ways this allows a more thorough approach to ecological restoration to be taken than has been possible in the past, and can be considered to have direct correspondence to the 'polluter pays' principle as set out in the European Directive on environmental liability (2004/35/EC) (Council Directive, 2004). This approach also allows the issue, that impacts may still persist after mitigation, to be addressed (Cuperus *et al.*, 1999), so providing a counterbalance for adverse impacts of development on nature such as habitat loss, degradation and isolation. These areas can be single habitat areas or mosaics of multiple habitats. In land use planning terms habitat banking offers a seemingly advantageous system with Brown and Lant (1999) identifying that in most situations, developers planning to convert wetlands to other uses lack the expertise to mitigate wetlands through restoration or other means, and in fact view the issue as an expensive and time-consuming requirement. The purchase of credits from a habitat bank administrator therefore allows economies of scale to be achieved, suitably qualified people to carry out the mitigation and ultimately a larger strategy to be developed and carried out.

The standard of no-net-loss is central to environmental mitigation and compensation. This relies on the principle that for a given development impact credits must be obtained in parity with the resources and functional value lost, and commensurate with scale and magnitude of impact (Latimer and Hill, 2007; Marsh *et al.*, 1996). This may be no-net-loss of habitat conditions, habitat types, populations of single species or species groups, or ecological functions (Cuperus *et al.*, 2001; Cuperus *et al.*, 1999). The principle holds for both in and out-of-kind compensation. In-kind compensation is the replacement of habitats or ecological functions with the same. This is likely to be a compensation method used to counteract habitat loss or habitat degradation (expressed in terms of species densities prior to development) and is therefore appropriate for both replacement and restoration compensation (Cuperus *et al.*, 1999), whereas out-of-kind is where the habitat affected by development is substituted for an alternative in the compensation scheme. On-site and off-site compensation are intuitive concepts, however, the zone of effect of a development needs to be specified in order to determine which type of compensation is required and implemented. The types and locations of mitigation and compensation sites are strongly influenced by i) the possibility of achieving the same ecological functions, and ii) the location of the compensation site relative to the development site (Brinson and Reinhardt, 1996).

Choosing sites that are already in protected areas or have high biodiversity value means that there is an overall net loss in biodiversity (Gibbons and Lindenmayer, 2007), whereas sites which have the potential to increase in ecological function and biodiversity value remain within the principle of no-net-loss.

The development of a systematic approach to compensation ratios would ensure impacts are adequately mitigated (Moilanen *et al.*, 2009). Two types of ratio must be applied, that relating to the ecological value of the bank's habitat and that assigned to the mitigation seeker. The accurate assigning of ratios ensures an alignment between mitigation and ecological impacts (Fox and Nino-Murcia, 2005). However, stipulated ratios agreed at the outset are often not achieved at the implementation stage. Quigley and Harper (2006), report a ratio of 6.8:1 (area gained : area lost) being stipulated, however, a ratio of only 1.5:1 was implemented. Where there is difficulty in quantifying impacts compensation ratios greater than 1 are more likely to be required. The case for larger compensation ratios can also be made where relatively slow ecosystem development times exist or where compensation is likely to have a high failure rate (Cuperus *et al.*, 2001). Whilst it may be possible to mitigate negative impacts on some habitats by compensation measures, other habitat types which take a long period to develop or are remnant ancient habitat cannot be considered to be truly replaceable, leading to restoration compensation being the only option suitable in these cases (Briggs *et al.*, 2009; Latimer and Hill, 2007).

2.1.5 Existing approaches: USA, South Africa, Australia and Northern Europe

Mitigation banks originate in the Clean Water Act, USA (Clean Water Act, 1977). Under this legislation mitigation was required to compensate for development taking place specifically on wetlands, with later amendments leading to the principle of 'no-net-loss' (US Environmental Protection Agency and Department of the Army, 1993). This approach has since been extended beyond wetlands and is now used as a mechanism to gain adequate compensation for unavoidable damage to a broader range of habitats and threatened or endangered species (State of California Department of Fish and Game, 2008). Therefore, two systems operate in the USA within the compensatory banking framework: mitigation banking – applicable to wetlands and administered through Section 404 of the Clean Water

Act (Clean Water Act, 1977), and compensation banking – applicable to habitats associated with threatened or endangered species and administered through Section 10 of the Endangered Species Act (Endangered Species Act, 1973). The mitigation and liability aspect of the two approaches applies to the restoration, creation, enhancement and, in exceptional circumstances, preservation of habitats expressly for the purpose of providing compensatory mitigation for authorised impacts to similar resources (Marsh *et al.*, 1996). The federal guidelines emphasize compensation as near to the permitted impact site as possible and for the same habitat type and function. The determination of compensation ratios is carried out in both quantitative and qualitative ways: best professional judgement, biotic indices, assessment based on species composition or habitat suitability for specific indicator species, surveys of habitat characteristics and landscape level assessments using GIS (Rundcrantz and Skärbäck, 2003; Stein *et al.*, 2000). In addition, multiple-habitat banks have been created containing native grassland, chaparral, coastal scrub and oak woodland, and to meet habitat requirements of endangered species such as the red-cockaded woodpecker *Picoides borealis* and Florida scrub-jay *Aphelocoma coerulescens* (Fox and Nino-Murcia, 2005). As an approach, habitat banking fits within wider planning legislation and therefore restrictions on location and types of development are determined through the existing system.

Unlike the USA, mitigation banking in South Africa has been adopted at the province level, with only the Western Cape having guidelines detailing how biodiversity offsets should be implemented. This reflects the globally unique biodiversity found there and the additional functions associated with biodiversity in the province, *e.g.* ecotourism and clean water supplies (Department of Environmental Affairs and Development Planning, 2007). However, biodiversity offsetting currently relies on thorough implementation of EIA (National Environment Management Act, 1998). As such, it is being used as a way of mitigating residual impacts from development after measures have been taken to avoid, reduce and repair impacts. Spatial plans and mechanisms are in place to deliver biodiversity offsets with receiving areas designated which reflect conservation priorities within the province. Clear area based compensation ratios have been developed to take account of ecosystem vulnerability, condition of affected ecosystem, significance of impact on threatened species, special habitats, ecological corridors and wider ecosystem services (Department of Environmental Affairs and Development Planning, 2007).

In Australia, impact mitigation is carried out and legislated for at the state level, *e.g.* Threatened Species Conservation Act in New South Wales (Threatened Species Conservation Act, 1997; Threatened Species Conservation Amendment Bill, 2006) and Planning and Environment Act in Victoria (Planning and Environment Act, 1987). Each state has adopted different approaches most suitable to the type and scale of development and ecological impacts encountered there. Mitigation, usually referred to as biodiversity offsets, focuses on restricting damage to and the management of native vegetation (Department of Planning and Community Development, 2008). The 'BushBroker' scheme operating in Victoria is the most advanced of the Australian approaches, allowing credit trading between land managers and developers with the aim of achieving a net gain in native vegetation. Ratios of impacts to biodiversity offsets are determined using Ecological Vegetation Classes (EVC) which allow quality benchmarks to be determined for each class. EVCs are aggregations of botanical communities defined by a combination of species composition, life form, position in the landscape and an inferred fidelity to particular environments (Parkes *et al.*, 2003; Treweek *et al.*, 2009). Their use allows a 'habitat hectare' score to be determined in a transparent and systematic way, taking account of habitat and landscape components.

In a European context the use of environmental compensation measures has a legal basis in EIA, Birds (79/409/EEC) (Council Directive, 1979), and Habitats Directives (92/43/EEC) (Council Directive, 1992). Whilst compensation of some type can be found in a number of European countries including Germany, the Netherlands and the UK (Rundcrantz and Skärbäck, 2003), it is most established in Germany.

The compensation impact regulation has been in place since 1976 (Federal Nature Conservation Act, 1976) in Germany. It is used to conserve and develop the capacity of nature and the landscape to perform their essential functions, and to define proper mitigation and compensation measures to reach a balance, *i.e.* essential functions in nature and the landscape are the same after a project is realised as they were before (Peters, 1993; Wende *et al.*, 2005). The 2002 Federal Nature Conservation Act amendment retained the instrument making it an expression of the German environmental policy principle 'the polluter pays'. Two principles make this approach viable for all scales and types of project: eco-account and compensation pools. The eco-account allows local authorities or developers to implement compensation measures in advance. It also addresses the issue of undertaking many assessments of multiple small projects. For a single developer, appropriate and available sites for replacement compensation measures may be difficult to find. For this reason replacement compensation pools have emerged, three type exist:

1. pools of sites, sufficient sites for the location of compensation measures related to a specific project will be bought or rented;
2. pools of measures for the compensation of the impacts related to specific planned projects;
3. pools of measures implemented in advance for future projects not yet planned (eco-account).

(Rundcrantz and Skärbäck, 2003)

The German system focuses essentially on replacement measures from a broader holistic perspective on a larger scale and with respect to overall spatial and functional connection. Developers of future projects can later make allowance for these measures by paying an equivalent amount that their measures should have cost.

The compensation system in the Netherlands, developed in the early 1990s, with its policy background in the National Structure Plan for Rural Areas (MANF and MHPE, 1993) and is linked to the Spatial Planning Act (Spatial Planning Act, 2008). However, its implementation has no national legal basis and it requires provincial authorities to incorporate the principle into their regional plans and implement it through agreements between affected parties (Rundcrantz and Skärbäck, 2003). The approach has two objectives: i) to enhance the input of nature conservation interests in decision-making on large-scale infrastructure projects and similar developments, ii) to bring about a no-net-loss situation for nature when a given development project is implemented, through the compensation principle (Cuperus *et al.*, 2001). The principle is applicable to National Ecological Network areas, nature areas outside the network and national species and protection plans. Targets exist for prevention and reduction of fragmentation and more generally acknowledge the many activities which contribute to a fragmented landscape (van

Bohemen, 1998). Restoration compensation such as adaptive design and subsequent management of farmland is preferred to replacement, however, ongoing management of compensation sites is required. Larger compensation areas are preferred to smaller in order to reduce fragmentation, which many result in replacement compensation playing a larger role in overall compensation schemes. A number of approaches to restoration and replacement compensation have been developed: habitat creation through land acquisition, adaptive design and subsequent management of farmland. The funding mechanism which allows the physical compensation to be undertaken is divided into two sections i) acquisition (where necessary), adaptive design and management, and ii) supplementary costs to include a quality allowance the level of which is determined by the development time (<25 years or 25-100 years), of the ecological quality on site (Cuperus *et al.*, 2001).

Whilst these approaches to development mitigation are advances in linking human and natural systems and ensuring retention of viability in both, there are still drawbacks which should be examined. Legislatively, the Netherlands, UK and South Africa are in a weak position with no mitigation banking principle in primary legislation other than the ratification of the SEA Directive in Europe. Cuperus *et al* (2001), suggest development of legislation would allow better accountability, encouraging initiators to reach consensus with involved parties, with citizens having a right of appeal when such consensus is not reached. Operationally, there has been a struggle to achieve the goal of no-net-loss. In Germany the legislation passed in 2002 was an attempt to ensure this principle, resulting in development of compensation pools. A review three years later revealed there still to be some, although possibly surmountable barriers to no-net-loss, namely, neglect of impact avoidance or reduction, loss of functional connection between impact and compensation, and competition for land (Wende *et al.*, 2005). In the USA, wetland mitigation has undergone several evaluations (Race and Fonseca, 1996; Roberts, 1993; Spieles *et al.*, 2006), however, there is still reported to be a continuing difficulty in translating mitigation concepts into legal principles, regulatory standards, and permit conditions that are scientifically defensible and sound (Race and Fonseca, 1996).

2.2 Landscape and ecological systems

Without a comprehensive understanding of the current landscape and species, ecological planning to counteract detrimental effects of development will not be scientifically or operationally robust (Moilanen *et al.*, 2009). It has long been recognised that a detailed analysis of spatial relations is indispensible for comprehensive planning on an ecological basis (Vos and Opdam, 1993). Much of the science of landscape ecology has been engaged in quantifying and analysing the current state of the landscape and provides a framework allowing landscape planning to be systematically approached through structure, function and change (Forman, 1995).

2.2.1 Landscape change and fragmentation

From the perspective of landscape ecological planning, compensatory approaches allow issues of fragmentation and habitat quality to be examined and addressed. Habitat fragmentation is a landscape-scale process involving both habitat reduction and habitat division. Four effects of the process can be defined: i) reduction in habitat amount, ii) increase in number of habitat patches, iii) decrease in sizes of habitat patches, and iv) increase in isolation of patches (Fahrig, 2003). It is widely recognised that habitat fragmentation as a result of social and economically driven processes is a major threat to the retention of species, viable populations and their associated habitat (Andrén, 1994; Markovchick-Nicholls *et al.*, 2008; Saunders *et al.*, 1991). The response to the fragmentation process of a population will vary widely among species (Opdam *et al.*, 1993). In the UK, decreases in bird species richness have often been associated with fragmentation (Bailey, 2007), as have the distribution of vascular plants associated with woodlands (Usher *et al.*, 1992). The results of persistent habitat fragmentation have long been recognised and form a major constituent of landscape ecology research. Much of this research has focused on identifying and describing the links between landscape pattern, function and process (Gustafson, 1998). It is now recognised that the maintenance of biodiversity (the abundance, variety and genetic constitution of native animals and plants) requires a landscape perspective that complements population, community, and ecosystem considerations (Franklin, 1993; Turner *et al.*, 2001). Investigations and suggestions as to how habitat fragmentation can best be countered draw together four strategies: i) increasing habitat amount, ii) decreasing number of habitat patches, iii) increasing sizes of habitat patches, and iv) decreasing isolation of patches.

Whilst describing and categorising the effects of habitat fragmentation is important, developing appropriate approaches to limit and reduce the causes of these phenomena needs also to be considered. Hobbs identifies this issue as one of the top ten research topics in landscape ecology, suggesting application development is of primary importance in tackling causes, processes and consequences of land use and land cover change (Hobbs, 2002).

> Long-term landscape changes imposed by economies and climate change, as well as land use legacies, need to be considered in the study of land use and land cover change. In addition, highly dynamic or chaotic landscapes, *e.g.* urbanizing landscapes, may provide unique opportunities for studying land use and land cover change. (Wu and Hobbs, 2002)

A number of methods are attempting to address the issues of extent, quality and connectivity of habitat, including the UK BAP, monitoring of Sites of Special Scientific Interest (SSSI) condition, and defining the extent of ecological networks (Catchpole, 2006; Watts *et al.*, 2005a). In a recent BAP review, priority actions were identified for the revised list of 1150 species and 65 habitats including

i. research,
ii. additional surveillance and monitoring,
iii. improvements to priority habitat extent or condition,
iv. site specific actions ,
v. conservation management to benefit a single species, and
vi. legal protection.

(Biodiversity Reporting and Information Group, 2007)

The use of habitat banks would need to take account of these existing approaches in terms of location and component habitats. Whilst the initial habitat banks in the USA were exclusively wetlands, the provision of a bank with a mosaic of semi-natural habitats is likely to be more robust and benefit a wider range of species. In areas where few natural ecosystems remain, linking sites into a network spreads the risk of local extinction across the landscape and allows linkages to be formed between conservation of protected sites and changes in land use and landscapes (Opdam *et al.*, 2006).

2.2.2 Ecological networks and metapopulations

Ecological networks, which are spatially defined systems of interdependent ecosystems that interact with the region in which they are embedded (Alterra, 2003), have gained in profile in recent years with approaches from the Netherlands leading the way (Jongman and Pungetti, 2004; Opdam *et al.*, 2006; Verboom and Pouwels, 2004). The scientific principles on which ecological networks are based: metapopulations and landscape ecology, explain why the concept is considered suitable and particularly well adapted to multidisciplinary research. Additionally, it is recognised that ecological networks might facilitate communication and decision making by actors in the planning process responsible for goal setting and design (Opdam *et al.*, 2006). James argues that the concept of an ecological network is implicit in Articles 3 and 10 of the Habitats Directive (James, 1999). It is here that EU Member States are encouraged to strengthen the functioning of Natura 2000 by protecting landscape elements for species dispersal and exchange (Jongman, 1995), so promoting ecological network development.

An ecological network is a multi-species concept determining both where individuals and populations exist currently and how they are linked into a spatially coherent system through flows of organisms and interactions with the wider landscape (Opdam *et al.*, 2006). The purpose of defining ecological networks theoretically and spatially in this research is to determine to what extent the landscape has undergone fragmentation and is therefore functioning in a suboptimal way. However, it is clear that the spatial definition of ecological networks must be based upon a direct evaluation of current connectivity of sites (Catchpole, 2006). Ecological effects of fragmentation are species-specific and depend on functional area, dispersal ability and isolation caused by barriers (Jongman, 2004). It is clear that the less fragmented a habitat resource is, *i.e.* the more it resembles its original extent, then the better it is for ensuring viability of populations. Ecological networks are defined using the language of metapopulations theory. Metapopulations are constructed of distinct populations which interact with each other, some groups undergo net loss (sink) whilst others undergo net gain (source), although patches are generally large enough to support local breeding populations (Hanski, 1999). In a fragmented landscape the colonisation rate of the metapopulation is reduced as there are fewer local populations and empty patches and a reduced level of patch connectivity, ultimately leading to a reduction in the number of patches that are occupied. This suggests that before all suitable habitat is destroyed the population level drops below the critical threshold and population extinction occurs (Hanski, 1999).

Networks at the landscape and ecosystem level are viewed as a mechanism necessary for the retention of important habitats which support a diverse range of species. This concept moves beyond the individual species and reserve focus that has so far dominated biodiversity conservation responses. Habitat networks can be regarded as an ecological, species based sub-division of ecological networks and greenways, based around the specific landscape requirements of a number of ecologically representative focal species (Watts *et al.*, 2005a). However, it is clear that no single optimum design can be developed for an ecological network which suits 'biodiversity' generally as each species has distinctive spatial requirements (Hawkins and Selman, 2002). Lambeck proposes three distinct sets of species that are sensitive to landscape change: area or habitat limited species, movement limited species and management limited species (Lambeck, 1997). Species level information therefore needs to be taken into account in the design of ecological networks. To this end a number of methodologies have been developed which attempt to bring a species level perspective to the design of ecological and habitat networks.

2.3 Ecological modelling and scenario testing

The use of models to examine ecological systems allows large scale and long timescale events to be observed within an appropriate time frame. However, the core of predictive geographical modelling in ecology is the quantification of species-environment relationships (Guisan and Zimmermann, 2000). The increased availability of digital data alongside GIS tools and applications means that the construction of models to represent habitat linkages and to assist reasoned environmental decision making is eminently possible (Clevenger *et al.*, 2002). Such approaches allow numerous alternative scenarios to be considered and both methodologies and results to be compared, so assisting in the proposal of optimum approaches and solutions to landscape planning questions. A growing area of ecological modelling is that incorporating climate change predictions and the resulting potential ecological impacts. Developing such models has allowed much interdisciplinary research to be initiated which supports a multifunctional landscape perspective. This is particularly important as knowledge of land use – landscape function relations is a prerequisite for the optimisation of both and the promotion of a multifunctional landscape (Wiggering *et al.*, 2006).

2.3.1 Ecological models and ecoprofiles

The most simplistic ecological models take single species and examine them in a modelled landscape (real or otherwise). However, without considering interactions between species, habitats and the wider landscape mosaic, little can be inferred as to the ecological processes occurring. Single species based approaches have been criticised as they do not provide whole-landscape solutions to conservation problems, they cannot be developed at a rate sufficient to deal with the urgency of the threats, and they consume a disproportionate amount of conservation funding (Franklin, 1993; Lambeck, 1997). However, the impossibility of monitoring the status and assessing the viability of all species is also clear (Noon and Dale, 2002). In response, the concepts of umbrella or flagship species has developed. These are species whose requirements are believed to encapsulate the needs of other species and ecological processes (Lambeck, 1997). However, it is unlikely that any single species could act as an umbrella for all species found within a particular ecosystem. It is suggested that for an umbrella species approach to be useful a multi-species approach that identifies species whose spatial, compositional and functional requirements encompass those of other species in the region is required – this subset can be termed 'focal species'.

Focal species are intended to act as representatives of wider biodiversity and key ecological processes. They are designed to represent various habitat types and particular ecological processes and vary in their sensitivity to habitat modification and fragmentation (Bolck *et al.*, 2004). Opdam *et al* (2003), stress that focal species or their ecological profiles should be regarded as part of the evaluation toolkit and not direct targets themselves. However, the focal species approach is not universally accepted with Lindenmayer *et al* (2002), arguing that the theoretical and practical underpinnings of the approach are not well established, with surrogate taxa unable to account for the requirements of all other species. Such issues of spatial and temporal scale, and population viability, data availability and inconsistency, whilst valid, are not unique to this approach (Coppolillo *et al.*, 2004;

Lambeck, 2002). The focal species approach should therefore be used as an aid to integrated landscape planning by assessing the relative merits of a landscape for particular representative focal species alongside other approaches (Lindenmayer *et al.*, 2002).

In many cases, however, it is desirable and necessary to create a number of generic ecological profiles (ecoprofiles) in order to reinforce the focus on landscape processes and represent the bulk of species for which insufficient autecological knowledge exists (Watts *et al.*, 2005a). Ecoprofiles are groupings of species which broadly share similar habitat requirements and exhibit similar behaviour in the landscape in response to landscape pattern and process, *e.g.* ancient woodland specialists with high area requirements and limited dispersal. Ecoprofiles are defined to be representative of a number of species groups, priority habitats and key ecological processes. Using this approach allows functionality to be incorporated into any network definition rather than merely identifying location extents.

Using ecoprofiles as the agents of an ecological network approach allows a wider range of species and habitats to be represented. This does mean that species specific details are not incorporated and individual species records are not used. However, there is much evidence that an ecoprofile or generic focal species approach has advantages (Lambeck, 1997; Vos *et al.*, 2007; Vos *et al.*, 2001; Watts *et al.*, 2005b), allowing entire landscapes to be incorporated into ecological networks. This type of multi-species approach fits well with the way landscapes are planned, that is, for biodiversity rather than for single species (Opdam *et al.*, 2003). It is clear then that any tools used must integrate conditions for a variety of species and, where appropriate data are not available, as is often the case, tools that are independent of actual species distribution data should be used (*ibid.*).

2.4 Green infrastructure planning

> Multifunctional landscapes are complex nature-culture interaction systems: landscapes, together with living systems and ecosystems are belonging to a special class of 'ecological interaction systems' whose elements are coupled with each other by mutual, mostly non-linear and cybernetic relations. For this organised complexity neither mechanical nor statistical approaches are satisfactory and innovative approaches and methods are required. (Naveh, 2001)

Naveh's fourth premise of a multifunctional landscape is clearly a founding principle of green infrastructure which considers the green and natural structures within the landscape as being part of an interacting system similar to the road or grey infrastructure system. Konijnendijk *et al* (2006). suggest green infrastructure refers to the functioning of the green structure which provides various services in line with other 'hard' types of urban infrastructure such as protecting biodiversity, and providing a social infrastructure. The green infrastructure approach to landscape planning is a growing area of research and practical interest in the UK where many see ecological networks and biodiversity conservation as an integral framework of green infrastructure into which other functions link (English Nature, 2003a; NW Green Infrastructure Think Tank, 2008; Stubbs, 2008; Tzoulas *et al.*, 2007). However, because of its interdisciplinary nature the green infrastructure debate it is not necessarily being progressed by landscape ecologists but is often seen as a strategic, policy driven concept (Milton Keynes and South Midlands Environment & Quality of Life Sub Group, 2005; Town and Country Planning Association, 2004). In the USA, where this approach is more fully developed, greenway planning is seen as a viable starting point which can, through logical progression, become a green infrastructure (Ahern, 2007; Benedict and McMahon, 2006). It is clear, however, that which ever sector takes the lead in this area green infrastructure has a major role to play in delivery of sustainable landscape planning.

2.4.1 People in the landscape

In landscapes undergoing significant levels of development there is a requirement to ensure spatial environmental equity for existing and future communities alongside retention of ecological functionality. Spatial equity is the degree to which services or amenities are distributed in an equal way (Neighbourhood Renewal Unit, 2005; Omer, 2006). Green infrastructure therefore provides a useful framework within which to explore this more holistic approach to landscape change. The use of landscapes by people allows strong connections between ecological functions and recreation and enjoyment to be made (Alessa *et al.*, 2008; Bishop, 1992; Neuvonen *et al.*, 2007). However, as economic development takes place the ability of communities to access natural greenspaces can become compromised and these connections weakened (Skärbäck, 2007). The measurement of community access to natural greenspace has therefore become necessary, however, the technical approach to this has been in development for a number of years (Handley *et al.*, 2003a; Harrison *et al.*, 1995). Whilst it has been identified that frequent contact with nature is a central component of well-being and a good quality of life in addition to safeguarding wildlife and geological features, facilitating learning and ensuring ecological functionality (particularly) of urban areas (Handley *et al.*, 2003a), the methodology for measurement and implementation remains in debate. Proposals have been made for a systematic approach to determine levels of greenspace accessibility as part of the development of English Nature's Accessible Natural Greenspace Standards (ANGSt) (Handley *et al.*, 2003b). It is suggested that candidate sites should be identified, their details (location, boundaries, access points and areas) incorporated into a Geographic Information System (GIS), and accessibility analysis undertaken (four size and distance categories are proposed) (Handley *et al.*, 2003b; Pauleit *et al.*, 2003). Once accessibility levels have been established a policy and management response can be proposed. The recommended approach, however has rarely been completed in full. Although researchers carrying out natural greenspace assessment commonly use the ANGSt size and distance criteria (Barbosa *et al.*, 2007; Comber *et al.*, 2008; McKernan and Grose, 2007), accessibility analysis methods used are varied in both type and precision. However, without a thorough assessment of the accessibility of natural greenspace, green infrastructure cannot be considered to be a multifunctional system for landscape planning (Kambites and Owen, 2006).

2.5 Climate change

Whilst economically and socially driven development has major impacts on the way the landscape functions, the impacts of climate change are likely to be significantly more wide ranging and indeed are already being felt (Berry *et al.*, 2003). Incorporation therefore of predicted impacts of climate change into landscape planning strategies is vital (Hansen *et al.*, 2010) particularly in considering how climate change scenarios may induce landscape and ecological system changes. The Intergovernmental Panel on Climate Change (IPCC) concluded that, by increasing the concentration of greenhouse gases man has a discernible influence on climate which is expected to be a long-term phenomenon affecting the environment in the forthcoming decades or even centuries (IPCC, 2007). Climate change is therefore likely to exert considerable effects on current biodiversity conservation goals (Opdam and Wascher, 2004). In response to the predicted impacts, Hansen *et al.* (2010) identify four tenets of 'climate-smart' conservation:

i. protect adequate and appropriate space;
ii. reduce non-climatic stresses;
iii. apply adaptive management to implement and test adaptation strategies;
iv. reduce rate and extent of climate change to reduce the overall risk.

It is suggested that genetic constraints on adaptation, together with land cover changes that impede gene flow, are likely to reduce the rate of species adaptation well below the unusually rapid pace of expected future climate change (Davis and Shaw, 2001). The suggestion that species response to climate change through genetic action will not be adequate to keep pace with climatic change in the landscape will be further exacerbated if it can be assumed that the spatial configuration of the landscape will also be a major influence on species (Opdam and Wascher, 2004). Travis (2003) examines this fragmented landscape scenario, an issue which has been ignored in many studies which assume availability of a universally colonisable surface (Opdam and Wascher, 2004). It is concluded that the interaction between climate change and habitat loss might be disastrous, with suitable habitat availability thresholds being reached sooner during climate change and therefore species suffering more from climate change in a fragmented habitat (Travis, 2003; Vos *et al.*, 2008). Clear, negatively synergistic effects of climate change and habitat fragmentation are therefore evident and, whilst methods have been outlined which can determine the drivers of landscape changes and fragmentation and potential responses through habitat expansion and functional linkage, the full cycle of cause, effect and response has yet to be completed.

In order to examine the many environmental parameters affected by climate change, systematic modelling and scenario testing needs to be employed. Various forms of ecological system modelling are proposed to inform land use planning and management and ultimately to promote the persistence of biodiversity (de Groot *et al.*, 2002; Drielsma *et al.*, 2007; Watts *et al.*, 2005a). It is suggested that for routine applications in conservation assessment and land use planning requiring consideration of large numbers of species, there is a need for modelling techniques that are ecologically rigorous, yet simple and tractable (Drielsma *et al.*, 2007).

2.5.1 Predictive models and bioclimatic envelopes

In order to make predictions about the impact of climate change on ecological systems, predictive simulation models are often employed. Such models may be either static or dynamic, with static models providing time-independent predictions while dynamic models predict time-dependent dynamic responses to a changing environment (Robertson *et al.*, 2003; Beerling *et al.*, 1995). Since the dynamic response of species to environmental change has infrequently been studied in the necessary detail, static distribution modelling is often the only approach possible in considering the consequences of a changing environment on species distribution (Guisan and Zimmermann, 2000; Woodward and Cramer, 1996). Model types can be divided further however, into mechanistic and correlative, highlighting the data types and techniques used in analysis of species – environment interactions. Mechanistic models use known interactions between organisms and their environment, e.g. climate, vegetation or terrain, to develop species range limits independent of species distribution data (Kearney *et al.*, 2010). However, the predictive accuracy of such models is acutely dependent on the identification of key limiting ecological processes. Correlative models exploit the statistical association between spatial environmental data and occurrence records to capture implicitly the processes limiting the distribution of species. Although this results in a flexible and simplistic model problems may arise when predicting species distributions in new environmental space, as it must be assumed that the many ecological processes evident in the original space are preserved in the new space (*ibid.*).

An acknowledgement of the shortcomings of mechanistic and correlative modelling approaches has led to the use of ensemble modelling (Araújo and New, 2007). Taking qualitative information alongside functional species traits has been identified as one way of incorporating ecological processes within a correlative model environment (Kearney *et al.*, 2010; Araújo and New, 2007). The ecological components of such models are often represented by bioclimatic envelopes which are used as surrogates for the range or potential range of the species of interest (Pearson *et al.*, 2002). The validity of bioclimatic envelopes, the distributional extent of a species which incorporates its climatic requirements (Berry *et al.*, 2002), has been frequently questioned, particularly as predictive climate models become increasingly sophisticated (Araújo *et al.*, 2005; Davis *et al.*, 1998). The central concern is that the future distribution of species, as may be encountered under a potential climate scenario, cannot be inferred merely by the spatial identification of climatically similar areas in the future landscape (Davis *et al.*, 1998). In response to this, multiple approaches have been proposed which additionally incorporate habitat and vegetation requirements of species (Baselga and Araújo, 2009; Beaumont *et al.*, 2007; Berry *et al.*, 2002; Brooker *et al.*, 2007; Harrison *et al.*, 2003). The requirements for such habitats to develop, *e.g.* soil water availability, growing days and temperature ranges, are then input into models along with various future climate scenarios, the result being a determination of where species may be found, with the caveat of the species requiring the ability to disperse to such a location (Pearson *et al.*, 2002). The validity of this next generation of climate models appears more readily accepted with large scale (national and international), projects using their results in order to develop adaptation strategies (Berry *et al.*, 2007a; Berry *et al.*, 2007b; Walmsley *et al.*, 2007).

2.5.2 Adaptation and landscape investment

Adaptation strategies resulting from climate change predictions take many forms including policy, strategy development and practical implementation. Whilst most modelling that has taken place to date considers ecological systems, particularly species responses, a multifunctional approach to landscape planning requires a broader reflection of potential impacts. The impact on people's health and broad-scale ecosystem services have been examined with the aim of understanding potential further issues which may arise from climate change and to subsequently propose response strategies (Ebi and Semenza, 2008; Gill *et al.*, 2008). The consideration of land use cover and linkage of this to the beneficial effects of urban greenspace (improved air quality, reduction of energy consumption in adjacent buildings and improvements in human health and well-being), corresponds particularly well with the perceived requirement for accessible natural greenspace as a fundamental part of the green infrastructure system (Gill *et al.*, 2008). Despite approaches considering climate change impacts on people and communities, a multifunctional approach which brings together response strategies has yet to be developed.

2.6 Conclusion

Whilst developing concepts and applications with a strong basis in landscape ecology and ecological functionality may initially appear to be biased towards environmental issues it is clear that the real world context of this research leads to a more holistic consideration of factors. It is evident that humans themselves and their activities must constitute an integral part of the ecology of landscapes (Wu and Hobbs, 2002). In order to develop an approach within the domain of green infrastructure, particularly in order to inform the suggested approach of mitigation banking, a more explicit interaction between natural and social sciences is required. This reveals a number of issues particularly relating to quantifying landscape and ecological functions. However, linking modelling scenarios to existing and widely accepted policy and standards will ensure assumptions are defensible.

CHAPTER THREE | Background

Materials and Methods

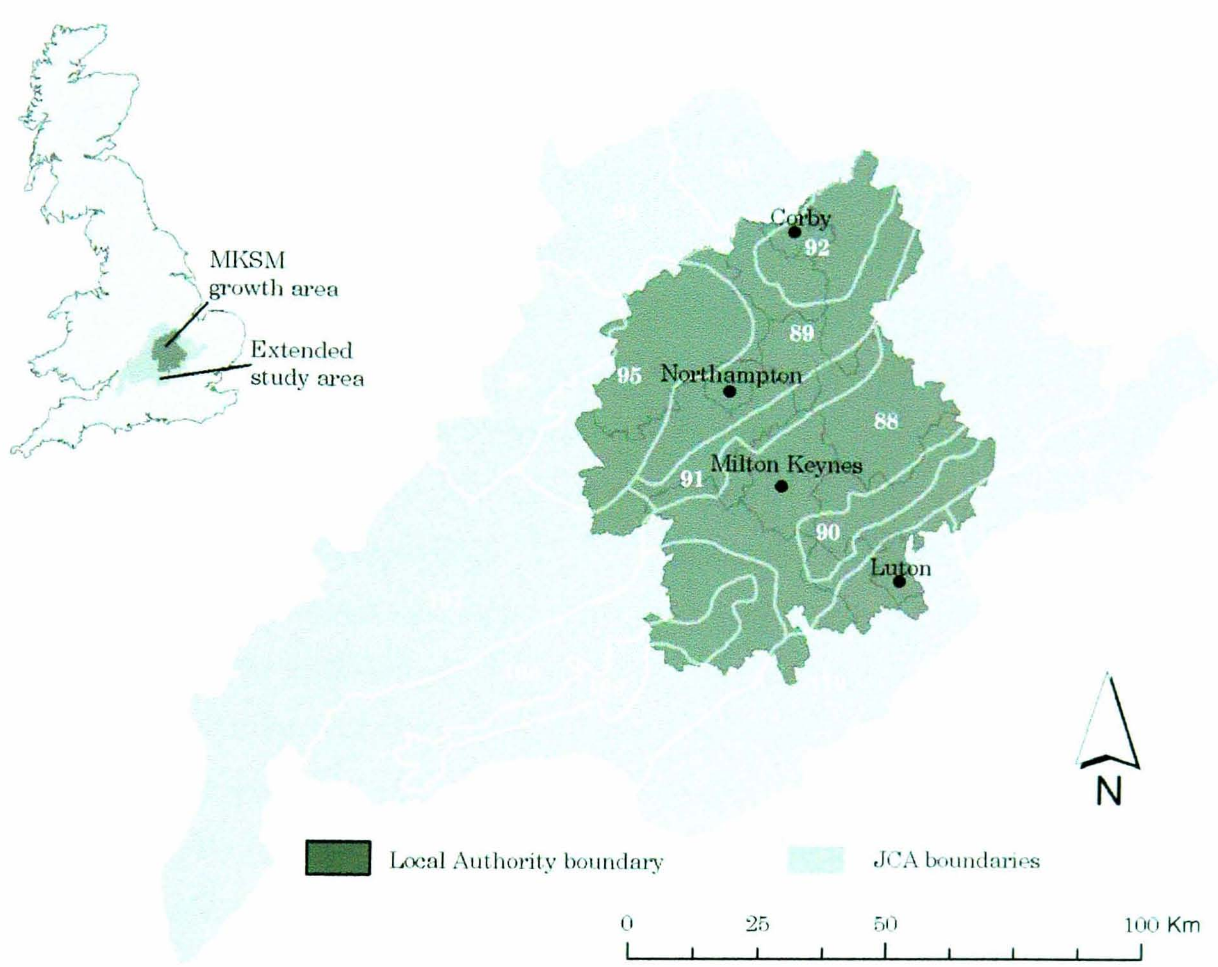
3.0 Introduction

This research required the use of three distinct methodologies: ecological network analysis, natural greenspace assessment and climate space tracking. These were considered in a nominated study area and were brought together to examine how the multifunctional nature of green infrastructure could be utilised in developing a landscape strategy. The flow of information from the ecological network analysis methodology to the natural greenspace assessment and climate space tracking methodologies placed ecology at the centre of the research approach. This provided a common thread through the research and acknowledged it as an essential component of a functional green infrastructure.

The interdisciplinary nature of this research is borne from a requirement to ensure a sustainable approach to landscape planning first identified by the United Nations (WCED, 1987). Significant and large scale changes are proposed in the study area, therefore, in order to develop an appropriate landscape strategy, a systematic approach incorporating economic, social and environmental landscape effects is required. The research approach taken examines these three elements of sustainability in the context of landscape planning, with an emphasis on spatial analysis. Examining the study landscape, its characteristics and attributes allows its current state to be established and an understanding of the drivers and pressures behind likely changes to be identified. Subsequently, methodologies used in the development of a response to the predicted impacts and landscape pressures were established. Detailed methodological approaches are included within chapters four, five and six.

3.1 Study area

The study area focuses on part of the South and East Midlands (Figure 3-1), covering approximately 4800 km^2. At the centre of this area is the Milton Keynes and South Midlands Growth Area (MKSM), a defined socio-economic activity area and a crucial part of the Government's ambition to increase levels of housing supply (DCLG, 2009b).



JCA Codes

87	East Anglian Chalk
88	Bedfordshire and Cambridgeshire Claylands
89	Northamptonshire Vales
90	Bedfordshire Greensand Ridge
91	Yardley – Whittlewood Ridge
92	Rockingham Forest
93	High Leicestershire
94	Leicestershire Vales
95	Northamptonshire Uplands
96	Dunsmore and Feldon
107	Cotswolds
108	Upper Thames Clay Vales
109	Midvale Ridge
110	Chilterns

Figure 3-1 Research study area

Joint Character Areas (JCAs) located within MKSM were examined in order to consider the socio-cultural elements of the landscape. JCAs are a spatial land typology determined through the consideration of landscape, wildlife, natural and cultural features and define areas of distinct character at a national scale (The Countryside Agency and Scottish Natural Heritage, 2002). The study area boundary was therefore created to prevent splitting of JCAs and thus encompasses 14 JCAs (Figure 3-1), the key characteristics of which are provided in Appendix 1.

3.1.1 Social and economic status

MKSM is one of the Government's four growth areas. Within this area are proposals for approximately 169,000 new dwellings to be built by 2021 (Government Offices for the South East *et al.*, 2005). Whilst it is planned to locate developments near to existing settlements, a significant proportion of the development will occur on greenfield sites (Figure 3-2). In concert with the planning of housing developments a novel method of compensatory mitigation has been developed for the Milton Keynes section of the growth area – a roof tariff (Milton Keynes Partnership, 2006). This requires a sum to be paid for each new dwelling to create a fund for infrastructure provision, including green infrastructure (Gillman, 2006). For the first time this approach allows a pooling of resources which can contribute to the alleviation of development impacts. Habitat banking, a proposal put forward to strategically target funds towards larger scale and higher quality habitat creation and enhancement (Gillespie and Hill, 2007; Hill, 2006; Treweek, 2000; Treweek and Thompson, 1997), appears to fit into the Milton Keynes model. This research proposes to link the roof tariff to the provision of habitat banking using green infrastructure to spatially determine appropriate bank locations. Habitat banks need to be both locally and nationally relevant incorporating national policy, whilst allowing delivery of local targets and being tailored to local landscapes.

There is currently a population of approximately 1.5 million within MKSM. With the proposed developments this is likely to increase to around 2 million by 2021 which will result in the need for significant new infrastructure resources. It is important to ensure green infrastructure is not overlooked within this infrastructure planning and development, as many benefits are associated with

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green infrastructure. The CLERE Model identifies five distinct aspects of greenspace: community, landscape, ecology, recreation, economy (Barber, 2005), whereas other approaches determine cross cutting yet specific benefits: urban renaissance, rural renewal, social inclusion and community cohesion, health and well-being and sustainable development (NW Green Infrastructure Think Tank, 2008). In line with sustainable landscape planning these potential benefits should be ensured for both current and future residential communities within MKSM.

3.1.2 Landscape and ecological status

The study area is a typical English lowland landscape dominated by large scale agriculture and interspersed with conurbations. Considering land cover by area reveals five dominant types: arable / horticulture 49.30%, improved grassland 22.22%, woodland 9.96%, urban / suburban 8.93% and natural grassland 8.50% (Figure 3-3).

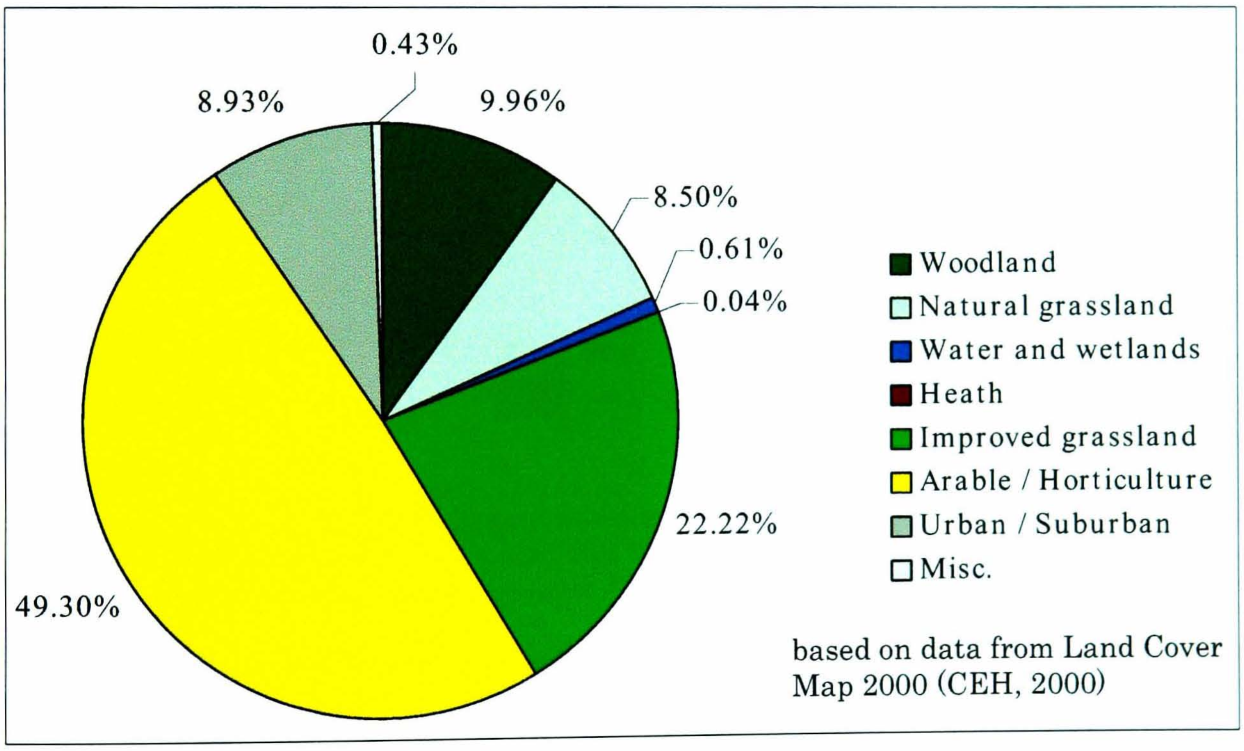
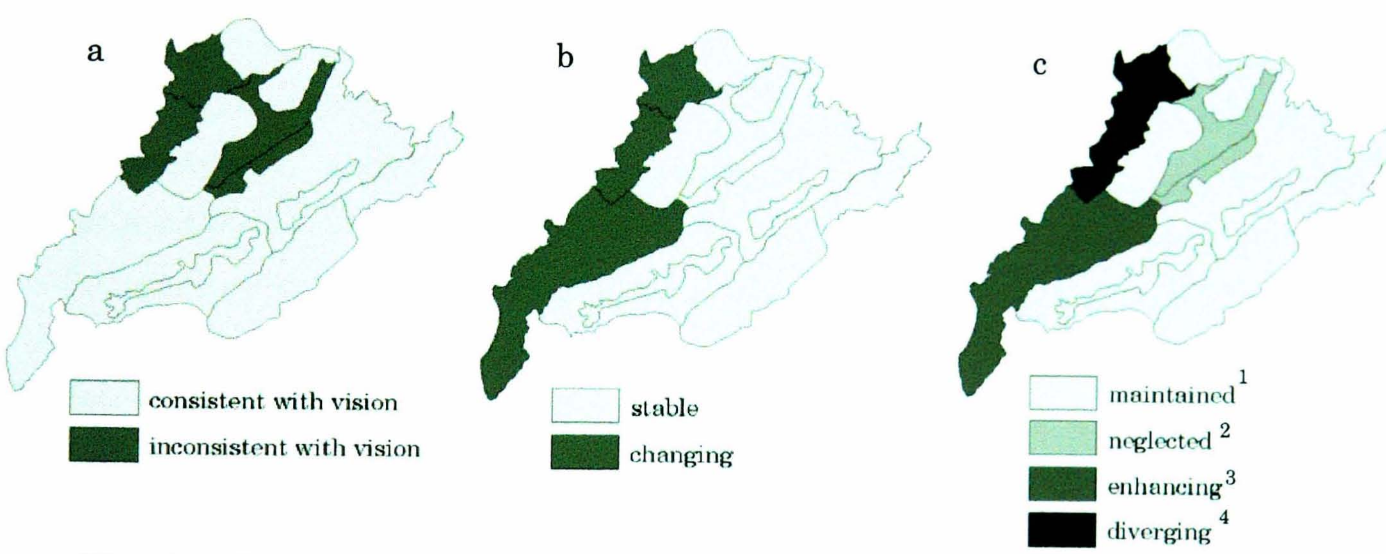


Figure 3-3 Land cover types found within the study area (% area)

This implies a landscape that has been heavily modified by man over a long period. An examination of the temporal changes in landscape character and the residual ecological value of the landscape reveals the extent to which this anthropogenic influence has resulted in fragmentation accompanied by a decrease and degradation of semi-natural habitats. The Rural White Paper 2000 (DETR, 2000) set out to consider ways in which the English landscape was changing. The Countryside Quality Counts (CQC) project developed from this notion of a need to monitor the landscape in order to understand temporal change and considered the period 1990-2003 (Natural England *et al.*, 2009a). The indicators developed, which assessed change in countryside quality, identified over 70% of the JCAs found in the study area to be stable and consistent with the character area visions created when the JCA typology was initially defined (The Countryside Agency, 1999a; The Countryside Agency, 1999b; The Countryside Agency, 1999c). Only 21% of JCAs were determined to be changing and 14% to be neglected (Natural England *et al.*, 2009b). Whilst it is accepted that landscapes evolve in response to their use and the values placed on them by their communities of place and interest, if the ostensibly positive results of the CQC project are considered spatially it is possible to identify definite patterns of change (Figure 3-4).



a. Direction of change from JCA vision statement
b. Magnitude of change from JCA vision statement
c. Status compared to JCA vision statement[1-4]

based on data from CQC
(Natural England *et al.*, 2009b)

Figure 3-4 Indicators of changes in landscape character 1990 - 2003 as determined by the CQC project.

[1] *Maintained*: if the character of an area is already strong and largely intact, and the changes observed for the key themes served to sustain it, or simply because the lack of change meant that the important qualities are likely to be retained in the long term. [2] *Neglected*: if the character of an area has been weakened or eroded by past change, and the changes observed in the key themes were not sufficient to restore the qualities that made the area distinct. [3] *Enhancing*: if the changes in the key themes tended to restore the overall character of an area or to strengthen it. [4] *Diverging*: if the change in the key themes appeared to be transforming the character of the area so that either its distinctive qualities are being lost, or significant new patterns are emerging (Haines-Young, 2007).

All JCAs identified as being, inconsistent with the JCA vision, in the process of changing, or being neglected, enhancing or diverging are located on the west side of the study area and represent only five of the 14 JCAs. This may suggest these were areas of vulnerable character, *i.e.* they contained particularly sensitive features or they have been subject to a higher degree of landscape pressure than areas in the east of the study area. It is interesting to note that the majority of development pressure exerted from MKSM is focused on the central and East side of the study area suggesting that the stronger and more intact JCAs are likely to be affected. JCA 89 – Northamptonshire Vales, however, is identified as being directly affected by development (East Midlands Regional Local Government Association, 2003), and also having suffered from a negative change in character (Natural England *et al.*, 2009b). Whilst stable, its direction of change is identified as being inconsistent with the JCA vision statement and of a neglected nature, suggesting an area of high sensitivity and vulnerability.

In addition to the character of the study area landscape its intrinsic ecological value, both quantitative and qualitative, requires examination to ensure a comprehensive context is available for this research. One approach to quantitatively determine ecological value is to identify the percentage of the landscape with a nature conservation designation: statutory, *e.g.* SSSI, or non-statutory, *e.g.* UK Biodiversity Action Plan (BAP) priority habitats, and consider how this compares to national and regional levels of designation (Table 3-1). Considering only statutory designations shows the study area to be a poor representation of the national and regional picture with only LNRs showing a similar level of occurrence. This result is a reflection of a number of factors and without thorough examination of historic data it is not possible to determine whether this landscape has ever been highly valued from a nature conservation perspective (according to the criteria of the individual designations) or whether the situation has arisen through the attrition of valuable ecological features. Conversely, UK BAP priority habitats and Ancient Woodland show the study landscape to have more in common with the landscapes of its constituent regions, with nine of the 14 habitats considered showing values within the range found in the regions and nationally. This reveals an interesting result – whilst over 7.5% (105,651 ha) of the study landscape is considered to be of high biodiversity value only 2.6% (44,154 ha) is legally designated. Whilst the importance of UK BAP

Table 3-1 Ecological value determined by considering statutory and non-statutory nature conservation designation types

Nature conservation designation type	statutory / non-statutory	England (%)	Region[1] (%)			Study area (%)
			South East	East Midlands	East of England	
RAMSAR	statutory	2.99	3.44	6.43	6.11	0.34
NNR	statutory	0.72	0.46	0.52	1.46	0.12
SAC	statutory	7.62	4.53	11.58	9.21	0.23
SPA	statutory	5.65	4.15	8.92	8.27	0.33
SSSI	statutory	8.10	7.12	6.66	7.70	1.49
LNR	statutory	0.27	0.52	0.15	0.24	0.14
Natura 2000 (approx.)[2]	statutory	8.79	6.15	11.70	12.02	0.54
Coastal and Floodplain Grazing Marsh	non-statutory	1.80	1.91	0.85	2.14	1.22
Lowland Beech and Yew Woodland	non-statutory	0.24	0.51	0.03	0.07	0.50
Lowland Calcareous Grassland	non-statutory	0.35	0.25	0.23	0.13	0.30
Lowland Dry Acid Grassland	non-statutory	0.39	0.18	0.62	0.49	0.01
Lowland Fen	non-statutory	0.89	1.78	0.38	0.22	0.10
Lowland Heathland	non-statutory	0.71	2.05	0.28	0.43	0.10
Lowland Meadow	non-statutory	0.24	0.25	0.16	0.45	0.16
Lowland Mixed Deciduous Woodland	non-statutory	1.01	2.72	0.55	0.49	0.94
Purple Moor Grass and Rush Pasture	non-statutory	0.18	0.17	0.20	0.31	0.02
Reedbed	non-statutory	0.51	1.03	0.18	1.46	0.23
Upland Hay Meadow	non-statutory	0.01	0.00	0.00	0.00	0.00
Upland Oakwood	non-statutory	0.19	0.13	0.02	0.04	0.12
Wet Woodland	non-statutory	1.19	3.44	0.90	0.82	1.15
Ancient Woodland[3]	non-statutory	2.65	6.73	1.53	1.41	2.70

[1] The majority of the study area (79%) is located within the three regions shown, the remaining study area is found within the West Midlands (6%) and South West (15%) regions.
[2] These values were calculated by merging the SAC and SPA layers and removing overlaps then calculating areas of newly formed polygons. The parent data is ver.2.6 SPA and SAC which is England-wide data only (Natural England, 2006a; Natural England, 2006b).
[3] Ancient Woodland whilst not a UK BAP Priority habitat can also be considered as an indicator of ecological value given that it details land which has been subject to continuous woodland cover since at least 1600 AD (Natural England, 2008a; Spencer and Kirby, 1992).

priority habitats are acknowledged through guidance, *e.g.* UK Biodiversity Action Plan (Department of the Environment, 1994), the planning system, *e.g.* Planning Policy Statements and Guidance (ODPM, 2002; ODPM, 2005) and through the provision of financial incentives, *e.g.* Environmental Stewardship (Natural England, 2008b; Natural England, 2008c; Natural England, 2008d; Natural England, 2008e), these systems afford relatively little legal protection for the habitats in question. This is of significance within the study area which has few protected areas and major and long term development pressures. Determining the quality of the ecological resource in the study area, whilst difficult, is a necessary prerequisite for developing a comprehensive landscape strategy. It is important to understand whether the existing habitat resource is robust and can be extended through the provision of a net gain in habitat area or whether an initial consolidation of the resource is required in order to ensure it is of suitable quality to provide a viable habitat for associated species and to meet their requirements.

Examining the quality of the existing statutory sites, *i.e.* SSSIs is one method of determining the current state of the landscape. It is noted that the study area contains relatively few SSSIs when compared to its constituent regions – 1.49% of the study area compared to 7.12% South East, 6.66% East Midlands and 7.70% East of England (Table 3-1). However, data detailing the monitoring of site quality is scarce, particularly that which is comparable both across regions and nationally. It is proposed that the SSSI status data be considered as an indicative measure of habitat quality within the study area rather than representing a detailed assessment. The majority (87.6%) of SSSIs in the study area are in a positive condition status, *i.e.* favourable or unfavourable recovering. This suggests an ecological resource that is well managed and relatively healthy. However, 12.4% (approximately 2800 ha) of SSSI is recorded as having a negative condition status, *i.e.* unfavourable no change, unfavourable declining, part destroyed or destroyed (Figure 3-5). When the condition status percentages are compared to the regional and national picture there is little difference in the proportion of SSSIs recorded in each status category (Figure 3-6). Although this can be considered a positive result the national and regional condition of SSSIs needs to be examined to ensure that the baseline being used for comparison is itself representative of a positive situation. For regional and national SSSIs the positive condition status values

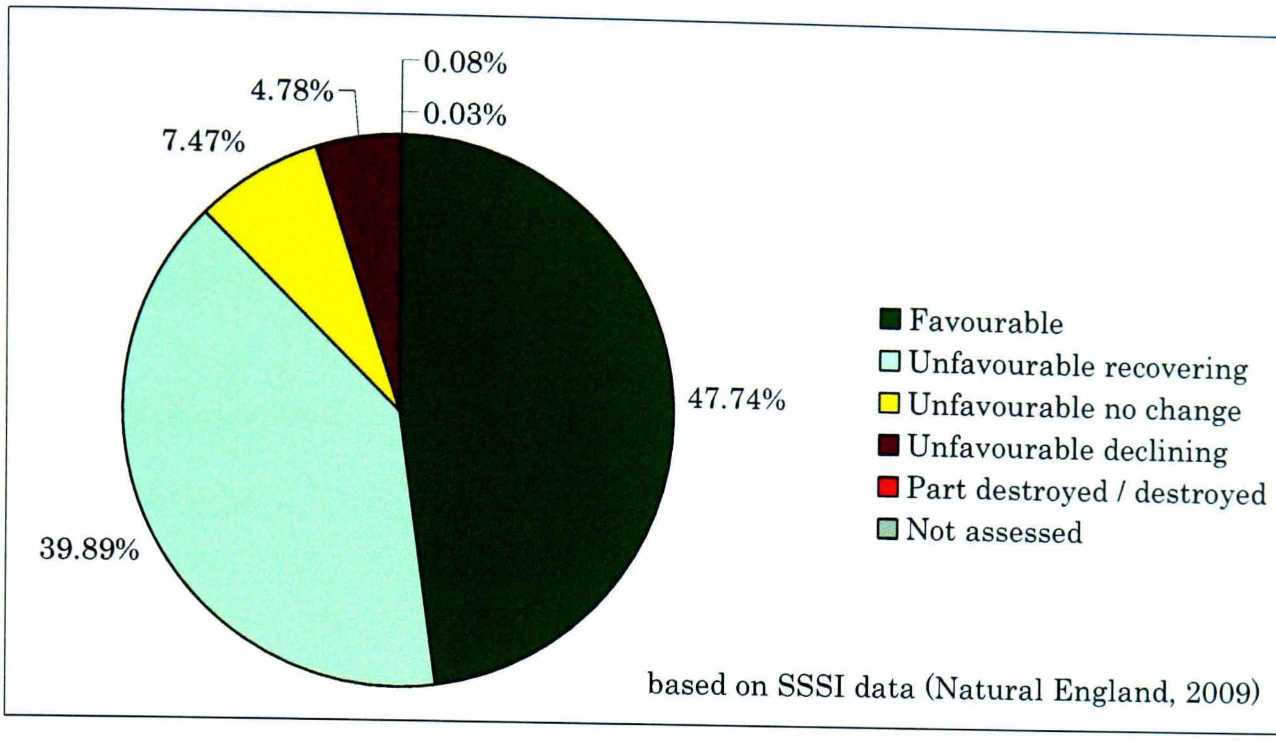


Figure 3-5 Status of SSSIs found in the study area

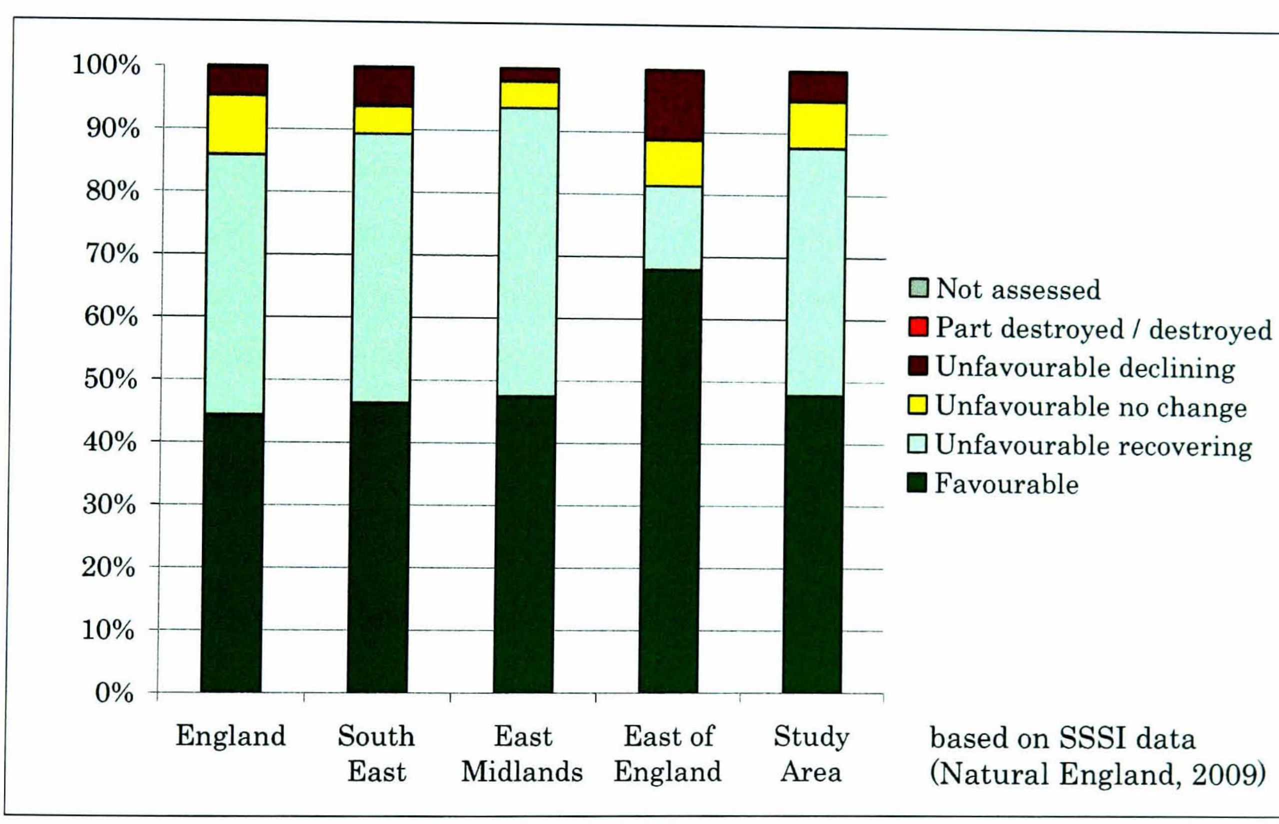


Figure 3-6 Comparison of national, regional and study area SSSI status

range from 81.4%-93.6% and the negative from 6.4%-18.6% (remaining sites are classified as not assessed). Whilst not an entirely positive scenario the relatively high values representing unfavourable recovering sites suggests a positive condition trajectory. It is important that the landscape strategy acknowledges this situation ensuring that areas of high ecological quality are retained and consolidated prior to extension and enhancement. This would be beneficial not only with respect to the continued viability of populations of species, but also to provide valuable natural greenspace resources for local communities and with respect to climate change to ensure the vital functions performed by green and open space remain available in the future.

3.1.3 Climate status

Whilst evidence of climate change is now compelling (IPCC, 2007), the location of the study area in the South Midlands has so far restricted the impacts of climate change felt there. However, it is likely that habitat and species will change in extent, character and distribution over the coming years as they respond and adapt to predicted changes (Piper *et al.*, 2006). Under the 50% probability scenario the study area is likely to experience mean daily temperature rises in both summer and winter of around 4°C, and a change in precipitation patterns (+10-30% in winter, -20-30% in summer) by the 2080s (Murphy *et al.*, 2009). Such large scale changes are likely to impact on many aspects of the landscape from species persistence to the ability to use greenspaces for recreation. Developing a landscape strategy which incorporates predicted climatic changes is therefore vital. By examining the results of the ecological network analysis and natural greenspace assessment within the context of these proposed climate changes it is suggested that necessary responses and adaptations can be incorporated into the landscape strategy developed.

3.2 Research structure

The individual methodologies developed to carry out ecological network analysis and natural greenspace assessment, and the approach used to consider climate space tracking, are detailed at the outset of the corresponding results chapters (four, five and six), in order to provide continuity of subject and context for the results. However, the linkages between the three components of the research have been examined here in light of the shared case study area.

CHAPTER **FOUR** | Methodology and Results 1

Ecological networks and their role in habitat bank location

4.0 Introduction

Habitat fragmentation is a landscape-scale process involving habitat loss and the disintegration of habitat. Often occurring as a result of social and economically driven processes it is a major threat to the retention of species, populations and associated habitat (Andrén, 1994; Markovchick-Nicholls *et al.*, 2008; Saunders *et al.*, 1991). Additionally, the quality of remaining habitat areas is often poor. It is clear that the less fragmented a habitat resource is, *i.e.* the more it resembles its original extent and quality, then the better it will be at ensuring population viability. An ecological network is a multi-species concept determining where individuals and populations exist and how they are linked into a spatially coherent but dynamic system through interactions with the wider landscape (Opdam *et al.*, 2006). Ecological network analysis and planning can be used to reduce habitat fragmentation. Defining ecological networks in this research determines the extent of current fragmentation and therefore impaired landscape function in order to develop an appropriate landscape strategy based on locating habitat banks. Habitat banks, sometimes referred to as mitigation or offset banks, are an intervention mechanism established to respond to the likely detrimental ecological effects of multiple independent built developments (Stein *et al.*, 2000). It is proposed that future developments in the MKSM study area could contribute financially to the establishment of habitat banks in order to consolidate and increase ecological network structures already present in the landscape.

4.1 Research questions

i. Which habitats are important in the study landscape, how can their functionality be assessed and what is their current quantitative and qualitative status?

ii. Do habitat patches currently function as ecological networks and what is the current extent of the ecological network resource in the study area?

iii. What economic, social and environmental factors need to be considered in spatially identifying ecological network extension areas, *i.e.* habitat banks?

iv. What ecological effect would the introduction of habitat banks have in the proposed locations?

4.2 Ecological network analysis methodology

4.2.1 Habitats and their status

Local BAPs in the study area: Bedfordshire and Luton, Buckinghamshire and Milton Keynes, Dacorum and Northamptonshire, were examined to identify habitats and species of local importance. Four habitat types incorporating ten BAP priority habitats were determined to be a focus for three or more local BAPs and were therefore considered to be of relevance to the study area: wetland habitats (reedbeds, fens and grazing marsh), lowland heath, unimproved grassland (acid, neutral and calcareous), and broadleaf and mixed woodland (lowland beech and yew, lowland mixed deciduous and wet woodland). Thirty one UK BAP species (amphibians, birds, butterflies, moths and terrestrial mammals only) associated with these habitats and recorded within the MKSM area were then identified using the National Biodiversity Network Gateway (National Biodiversity Network, 2007) (Appendix 2). A literature search was carried out and established the following important species characteristics:

a. breeding habitat preferences,
b. life cycle habitat preferences,
c. minimum breeding area requirement,
d. maximum dispersal distance, and
e. dispersal ability over non-breeding habitat.

Nine ecoprofiles were then created incorporating values for the five characteristics representing the range of values found for the 31 BAP species (Table 4-1).

Table 4-1 Characteristics of ecoprofiles developed for the study area

Ecoprofile	Breeding: habitat	Breeding: area (ha)	Life cycle[a]: habitat	Dispersal: distance (km)	Dispersal: general ability[b]	Literature
Wetland habitat – A	ditches, reedbeds	0.5	heathland	2.6	*good:* sand, forest, road *poor:* agricultural land	(Miaud and Sanuy, 2005; Sinsch, 1992; Sinsch, 1997; Stevens *et al.*, 2006; Stevens *et al.*, 2004)
Wetland habitat – B	marsh, wet emergent areas	1.0	scrub, arable fields	4.9	*good:* semi-natural grassland, water *poor:* woodland, urban / suburban	(Brickle and Peach, 2004; Cramp and Perrins, 1994; Fuller *et al.*, 2004; Gregory and Baillie, 1998; Pasinelli and Schiegg, 2006)
Lowland heath – A	heathland	1.0	coppiced woodland	0.6	*good:* unimproved grassland	(Asher *et al.*, 2001; Bergman *et al.*, 2004; Holloway *et al.*, 2003; Warren, 1987)
Lowland heath – B	heathland	4.5	unimproved grassland	4.0	*good:* sand, gravel *poor:* agricultural land	(Cramp, 1988; Mallord *et al.*, 2007; Sitters *et al.*, 1996)
Unimproved grassland – A	short semi-natural grassland	2.0	pasture	3.0	*good:* agricultural land *poor:* roads, urban / suburban	(Cramp, 1983; Green *et al.*, 2000; Thompson *et al.*, 2004)
Unimproved grassland – B	calcareous grassland	0.8	arable fields, unimproved pasture	1.0	*poor:* woodland, scrub	(Asher *et al.*, 2001; Hill *et al.*, 1996)
Broadleaf & mixed woodland – A	mature broadleaf woodland	1.0	parkland	3.0	*good:* scrub, agricultural land	(Cramp and Perrins, 1993; Freeman and Crick, 2003; Stevens *et al.*, 2007)
Broadleaf & mixed woodland – B	Ancient and broadleaf woodland, PAWS	20.0		0.5		(Bright, 1996; Bright *et al.*, 2006)
Broadleaf & mixed woodland – C	Ancient and broadleaf woodland, PAWS	0.8		1.0	*good:* scrub	(Field, 2004; Waring and Field, 2002)

[a] includes: foraging, feeding and roosting (where applicable).
[b] *'good'* dispersal ability is used where positive association is noted for a habitat type in literature, *'poor'* dispersal ability is used where negative association is noted for a habitat type in literature. Only habitat types specifically identified in literature are included, those previously stated as either breeding or life cycle habitats are classified as allowing 'good' dispersal ability.

Ecoprofiles are a composite of species representing comparable habitat requirements, dispersal abilities, and area requirements (Vos *et al.*, 2007). They are constructed from groups of species which exhibit a strong association with a particular habitat. The use of multiple ecoprofiles for the same habitat type allows a wider range of species associated with that habitat to influence the identification of ecological networks. The nine ecoprofiles constructed incorporate traits of specific BAP species (Table 4-2).

Table 4-2 Constituent BAP species of each ecoprofile

Ecoprofile	BAP species incorporated
Wetland habitat A	*Bufo calamita*
	Arvicola terrestris
Wetland habitat B	*Emberiza schoeniclus*
	Pipistrellus pipistrellus sensu lato
Lowland heath A	*Melitaea athalia*
	Carduelis cannabina
	Lanius collurio
Lowland heath B	*Lullula arborea*
	Caprimulgus europaeus
Unimproved grassland A	*Burhinus oedicnemus*
	Polia bombycina
	Heliophobus reticulate
	Hemaris tityus
Unimproved grassland B	*Hesperia comma*
	Lysandra bellargus
	Dorycera graminum
	Asilus crabroniformis
Broadleaf and mixed woodland A	*Muscicapa striata*
	Barbastella barbastellus
Broadleaf and mixed woodland B	*Muscardinus avellanarius*
	Boloria euphrosyne
Broadleaf and mixed woodland C	*Xestia rhomboidea*
	Catocala promissa
	Catocala sponsa
	Dicycla oo
	Jodia croceago
	Mythimna turca
	Pechipogo strigilata
	Rheumaptera hastata
	Trisateles emortualis

Using a Geographic Information System (GIS) existing habitat suitable for each ecoprofile was mapped based on characteristics a, b & c (Appendix 3). A continuous surface was then created representing land use type across the landscape. This land use map was constructed using Land Cover Map 2000 (CEH, 2000), UK BAP priority habitat inventories (English Nature, 2001a; English Nature, 2001b; English Nature, 2001c; English Nature, 2001d; English Nature, 2002a; English

Nature, 2002b; English Nature, 2002c; English Nature, 2002d; English Nature, 2002e; English Nature, 2003b; English Nature, 2004a; English Nature, 2004b; English Nature, 2004c; Natural England, 2008f), National Inventory of Woodland and Trees (Forestry Commission, 2002) and, where appropriate, extracted semi-natural areas from Ordnance Survey (OS) Mastermap (Ordnance Survey, 2006). Characteristics a, b, d & e were then combined to determine the permeability of each land use type for each ecoprofile in the continuous surface. Land cover types were allocated a score (1-50) based on associations between land cover type and ecoprofile (Table 4-3).

Table 4-3 Cost surface scores of land cover type allocated for each ecoprofile

Land cover type	Cost score
Breeding habitat	1
Habitat associated with other life cycle events	2
Habitat with positive associations	3-10
Semi-natural habitats but no known association	20
Habitat with negative association or man-made features	50

Scores were determined from literature, with a low score indicating a strong preference or ability for movement and a high score a barrier to movement. The land cover descriptions were compared to breeding and life cycle habitat requirements for each ecoprofile as identified in Table 4-1. Although the same score values were used for all ecoprofiles by matching the land cover type descriptions (Table 4-3), and the specific habitat requirements and barriers identified for each ecoprofile (Table 4-1), ensured that the resulting land cover maps were ecoprofile specific. A number of intentional implications of the score values can be identified:

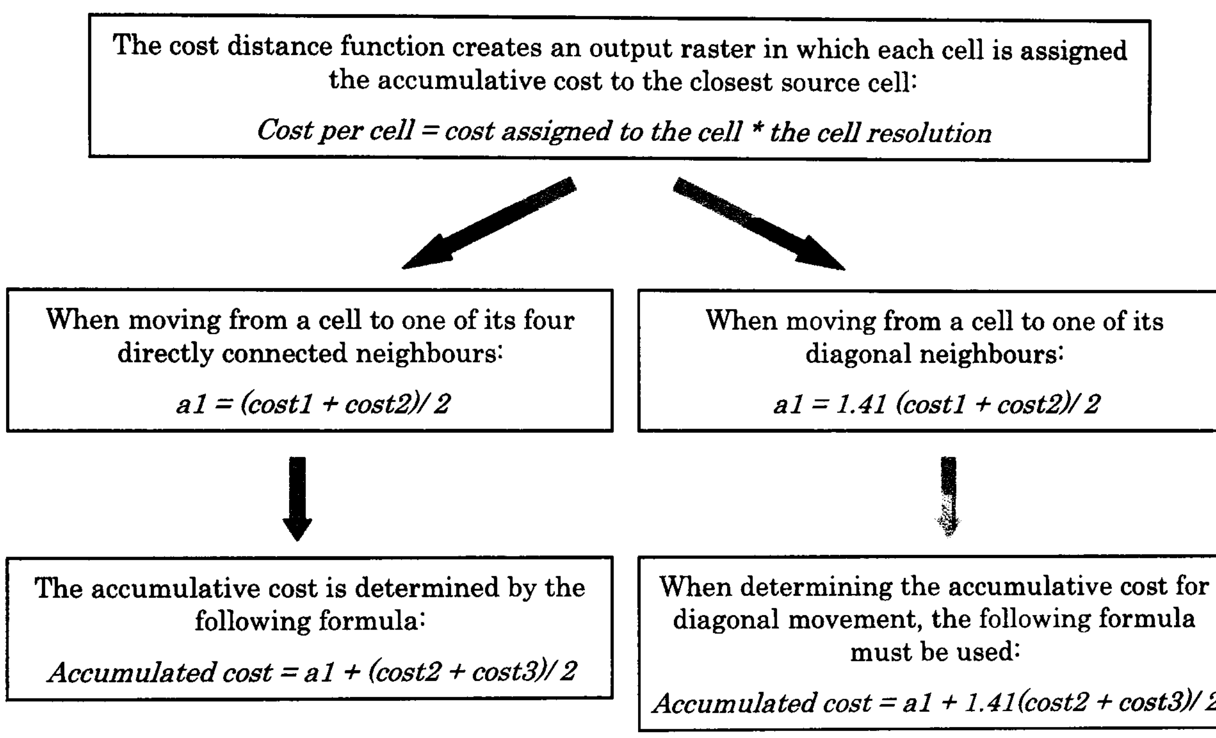
i. A score of 1 allows movement to continue through the landscape unhindered and is only restricted by the maximum dispersal ability of an ecoprofile.

ii. A score of 50 would result in a halt to movement in the direction of this land cover type. This is representative of barriers in the landscape.

The creation of unique land cover surfaces for each ecoprofile allows the characteristics of the BAP species which the ecoprofile is constructed from to be taken into account, e.g. if a bird species is included then barriers with a score of 50 are infrequently identified. A continuous land use surface map was constructed for each of the nine ecoprofiles representing the landscape as perceived by each ecoprofile.

4.2.2 From habitat patches to ecological networks

The ecological network analysis utilised a least cost path algorithm (Adriaensen *et al.*, 2003; Drielsma *et al.*, 2007; Watts *et al.*, 2005a; Watts *et al.*, 2005b). The least cost measurement is dissimilar from Euclidean measurements as, instead of calculating the straight line distance from one point to another, it determines the shortest weighted distance or accumulated travel cost from each cell to the nearest cell in the source cells (Figure 4-1). The weighted distance functions apply distance in cost units, not in geographic units (ESRI, 2007).



Adapted from (ESRI, 2007)

Figure 4-1 The least cost path approach to determining ecological networks

This determined the connectivity of the landscape for each ecoprofile by analysing the ability to travel between patches of existing habitat and through the continuous landscape up to the maximum dispersal distance set by characteristic d. This approach allowed ecological networks to be identified as spatially connected habitat of an adequate size to sustain a minimum viable population. The ecological network is therefore constructed from both habitat and non-habitat areas, with the non-habitat areas functioning as conduits. The methodology assesses two types of landscape structure, landscape composition and landscape connectivity, by incorporating distribution of patches in the landscape and the degree to which the landscape facilitates or impedes movement among resource patches (Taylor *et al.*, 1993).

4.3 Habitat bank location

Whilst a thorough assessment of the ecological networks in the study area can be completed using the methodology outlined above, economic, social and other environmental factors which will influence and affect the landscape need to be considered prior to determining potential habitat bank locations. Economic factors are of particular relevance given the nature of the study and the projected housing growth proposals. The social functions of green infrastructure, and an appropriate methodology to incorporate these into the location of habitat banks is outlined in chapter five.

4.3.1 Economic factors

Growth area assessment reports were examined for each of five growth clusters within MKSM centred on existing conurbations: Bedford, Corby – Kettering – Wellingborough, Luton – Dunstable – Houghton Regis, Milton Keynes and Northampton, (East Midlands Regional Local Government Association, 2003; East of England Local Government Conference, 2003; East of England Local Government Conference and Government Office for the East of England, 2003; South East England Regional Assembly *et al.*, 2003). Each cluster employed an assessment process whereby potential development sites were examined against criteria such as infrastructure requirements and potential for development to cause merging of neighbouring settlements. Broad indications of dwelling numbers, dwelling density and provision of none built land were then given for each site. Using this information, preferred development locations were mapped onto JCA boundaries to determine the number of dwellings planned within each.

Within the study area one growth cluster, Milton Keynes, has developed a strategic approach to funding infrastructure required as a result of development. The Milton Keynes roof tariff requires developers to contribute £18,500 per dwelling (Milton Keynes Partnership, 2006). It was felt that this was an appropriate financial instrument to link to the planning of habitat banks. For the purposes of this research, therefore, the Milton Keynes roof tariff has been applied to the entire study area. This was felt to be fitting as a number of other Local Planning Authorities (LPA) are considering and developing similar approaches (Walker, 2007). The level of financial contribution was identified through considering: overall development timescale, new dwelling numbers, new commercial floor space area, new resident numbers, and understanding of infrastructure and services required to be delivered (Milton Keynes Partnership, 2006). Through itemisation of costs the Local Delivery Vehicle (LDV) responsible for co-ordinating and planning growth was able to work with developers and landowners to agree a suitable level of tariff (Milton Keynes Partnership, 2007). As no detailed funding breakdown was provided in growth area literature the full £2590 per dwelling (14.1% of total roof tariff receipts) allocated for landscape and open space (Milton Keynes Partnership, 2007) was used to calculate the total compensatory fund available from development for green infrastructure and thus habitat banks.

It was determined that the cost of developing habitat banks could be split into two areas, land acquisition and habitat creation. Land values were determined by examining land for sale (without planning permission) in the study area. Cost, size of land parcel and type of land (grazing / paddock, agricultural, woodland, other, *e.g.* derelict, recreation, gravel) were recorded. Data were collected over a three week period in March and April 2008 (uklanddirectory ltd., 2008), resulting in 41 parcels of land being identified ranging in size from 0.05 – 40 ha. This allowed an average cost per land type per county to be calculated (Table 4-4).

Table 4-4 Cost of land acquisition by land type and by county

County	Land acquisition costs per ha				
	grazing / paddock	agricultural	woodland	other*	*Average*
Bedfordshire	£152,874	£197,684	£78,250		*£142,936*
Buckinghamshire	£42,997	£128,459		£164,737	*£112,064*
Cambridgeshire	£14,692	£44,862		£38,831	*£32,795*
Hertfordshire	£121,082	£171,782	£224,103	£41,184	*£139,538*
Northamptonshire	£288,290	£218,079		£329,474	*£278,614*
Oxfordshire		£27,149			*£27,149*
Average	*£123,987*	*£131,336*	*£151,177*	*£143,557*	

* includes land described as derelict, recreation and gravel

Considerable variation in land acquisition costs can be seen by comparing county of site location, existing land type, adjacent land types and size of site. A standard set of costs for habitat creation was determined from a Heritage Lottery Fund (HLF)

Review (Bailey and Thompson, 2007) which evaluated HLF projects from 1994-2006 with costs of habitat works determined by region and by 17 habitat categories comparable to UK BAP priority habitats. Minimum and maximum areas of habitat creation were calculated (Table 4-5) to ensure potential habitat banks determined through spatial analysis adhered to the financial limitations set by the roof tariff approach to funding habitat banks.

Table 4-5 Habitat creation budgets

Habitat type	Habitat creation costs	Minimum creation area[a] (ha)	Maximum creation area[b] (ha)
Wetland habitat	£4,204	1574	64 235
Lowland heath	£3,447	1581	78 341
Unimproved grassland	£6,492	1553	41 596
Broadleaf & mixed woodland	£7,692	1542	35 107

[a] includes habitat creation and land acquisition
[b] includes habitat creation costs only

4.3.2 Environmental factors

Whilst the location of existing ecological networks is a strong indicator of where it would be most appropriate for habitat banks to be located it was necessary to consider other environmental factors to ensure a systematic approach was developed. The study area was examined from a land use planning perspective using a series of constraints rules which were constructed following a literature search (Table 4-6). This aimed to reduce the amount of land considered suitable for habitat bank location and in this way was viewed as a 'land filtering' process. Constraints examined interactions between ecoprofiles and the landscape and introduced anthropogenic factors into the decision making process. The filtering process was carried out separately for each ecoprofile, data were then joined for ecoprofiles from the same habitat type in order to determine bank locations based on habitat type rather than for each ecoprofile. This approach fits well with the definition of ecological networks being multi-species constructs. Existing breeding and life cycle habitat preferences and existing ecological networks were then mapped onto the potential habitat banks. Locations intersecting existing ecological networks were identified. However, where this resulted in potential bank areas larger than habitat creation budgets the individual attributes of potential banks were examined (Figure 4-2). A detailed breakdown of number of patches,

Table 4-6 Constraints rules applied to the study area to identify possible habitat bank locations

Constraint rule[a]		Wetland habitat	Lowland heath	Unimproved grassland	Broadleaf & mixed woodland	Literature
1. Land is currently a UK BAP priority habitat		Applied	Applied	Applied	Applied	(Jackson, 2000; UK Biodiversity Action Plan, 2008)
2. Land is grade 1 or 2 agricultural land		Applied	Applied	Applied	Applied	(Ministry of Agriculture Fisheries and Food, 1988)
3. Land is within the detrimental zone of influence of urban areas (km)		0.25	0.40	0.12	0.40	(Blair and Launer, 1997; Hogsden and Hutchinson, 2004; Palomino and Carrascal, 2007; Stevens *et al.*, 2007)
4. Land is within the detrimental zone of influence of main roads (km)		0.25	0.12	3.00	0.30	(Bright, 1996; Bright *et al.*, 2006; Green *et al.*, 2000; Hogsden and Hutchinson, 2004; Houlahan and Findlay, 2003; Palomino and Carrascal, 2007; Reijnen *et al.*, 1996; Reijnen *et al.*, 1995)
5. Land is above 300m altitude		Applied	Applied			
6. Land is within the detrimental zone of influence of potential runoff sources (km)		Buffer 0.03				(Hickey and Doran, 2004)
7. Soils are suitable for habitat development	*texture*	clayey, sandy, loamy	sandy	loamy	loamy – sandy	(National Soil Resources Institute, 2003)
	drainage	naturally wet – impeded drainage	variable – freely draining	freely draining	impeded drainage – freely draining	
	fertility	very low – moderate	very low – low	lime rich – moderate	low – high	
8. Area of land available is ≥ minimum breeding area required by all ecoprofiles within a habitat (ha)		1.0	4.5	2.0	20.0	See Table 1
9. Land is adjacent to a major or minor river		Applied				
10. Land is adjacent to an area of ancient semi-natural woodland					Applied	(Natural England, 2008a)
11. Land is adjacent to existing ecological network		Applied	Applied	Applied	Applied	

[a] Land selected through the constraint rule process was treated in two ways: i) rules 1-6, selected land was unsuitable for habitat banks, this land was removed from further selections, ii) rules 7-11, selected land was suitable for habitat banks if all rules applicable were met.

minimum patch size, maximum patch size, total patch area, mean patch area and standard deviation was determined for each habitat type, in each JCA where development was proposed. This provided a good overview of potential options for habitat bank locations. Further prioritisation of sites for habitat banks is likely to be best assessed at the local scale based on this information and local priorities.

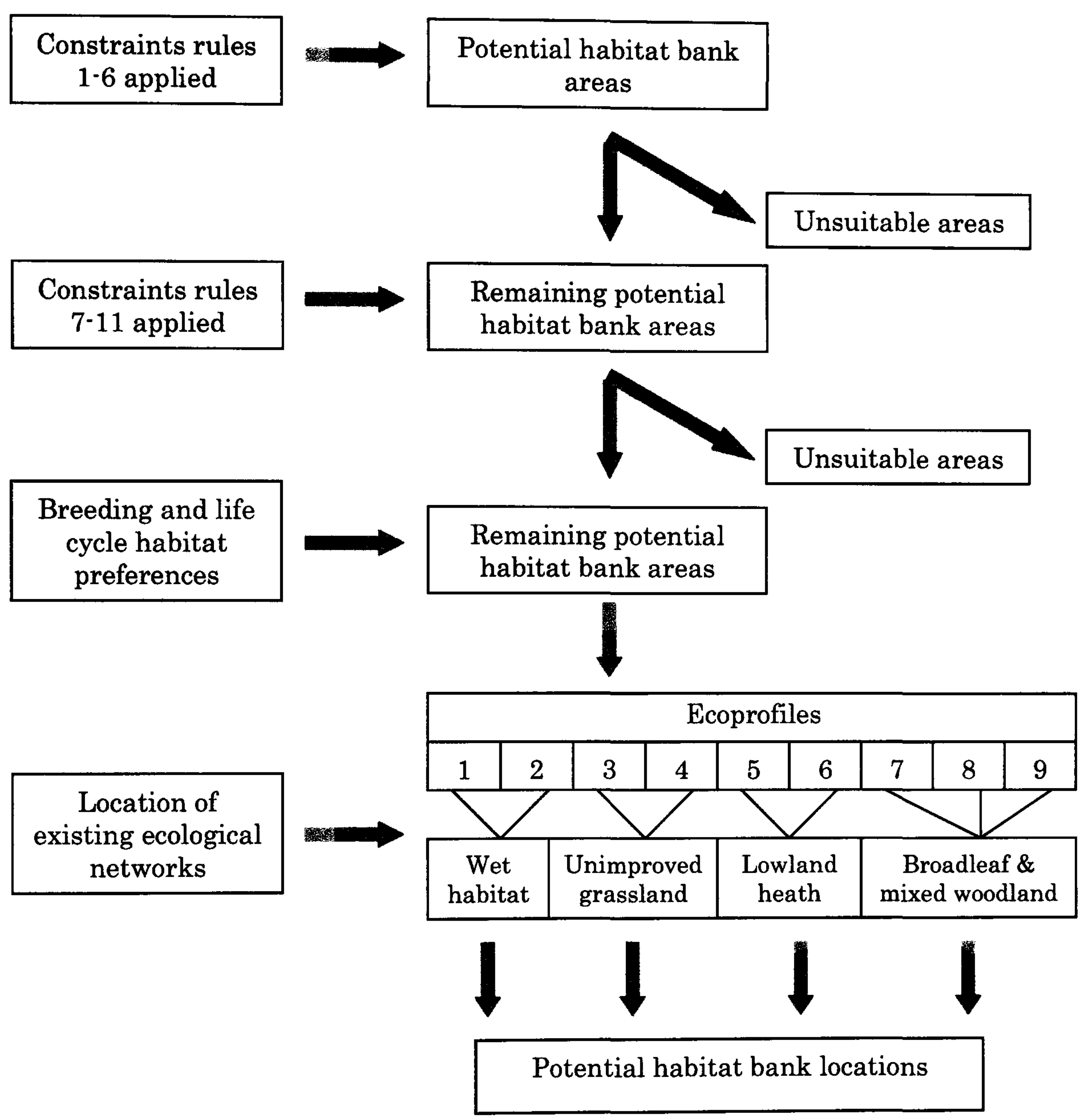


Figure 4-2 Locating habitat banks based on environmental factors

4.4 Results

Considering the extent of existing habitat and how this is configured and functions within ecological networks provides a strong basis for determining appropriate habitat bank locations. However, without incorporating other economic and environmental factors (this chapter) and social factors (chapter five) habitat banks are unlikely to fit well within a sustainable multifunctional green infrastructure system. Considering the potential ecological outcomes of habitat bank locations from a structural, functional and dynamic perspective allows the validity of the methodology developed to be assessed both in the terms of landscape ecology and in the use of the roof tariff as a suitable financial instrument.

4.4.1 The functionality of the existing habitat resource

Using ecoprofiles to assess the current functional ecological status of the study area reveals a paucity of existing habitat. A maximum of 8.92% of the study area is suitable for the ecoprofiles (Table 4-7).

Table 4-7 Fragmentation level of existing habitats

	Number of habitat patches	Total area (ha)	% of study area	% of habitat resource in study area	Average patch area (ha)
Wetland habitat - A	2540	26044.27	1.56	98.71	10.26
Wetland habitat - B	1937	25564.80	1.53	97.00	13.20
Lowland heath - A	63	1713.67	0.10	84.78	27.64
Lowland heath - B	59	1933.10	0.12	97.03	33.33
Unimproved grassland - A	737	7351.99	0.44	92.54	9.99
Unimproved grassland - B	755	4914.97	0.29	61.86	6.52
Broadleaf & mixed woodland - A	9651	67181.28	4.03	41.10	6.96
Broadleaf & mixed woodland - B	708	31544.80	1.89	19.30	44.62
Broadleaf & mixed woodland - C	23775	113307.16	6.80	51.45	4.77

Note: total areas for ecoprofiles within the same habitat type cannot be summed owing to overlaps in habitat patches determined to be appropriate for each ecoprofile. Suitability of the study area habitat resource is calculated using the ecoprofile with the highest percentage area for each habitat type.

Considering the level of fragmentation by determining the extent of existing habitat areas is strongly influenced by the requirements of ecoprofiles (Table 4-1). The ecoprofiles show varied results when considering the percentage of the study area habitat resource they are theoretically able to utilise ranging as high as 98% (wetland habitat) to as low as 19% (broadleaf and mixed woodland). For those ecoprofiles with a good habitat resource / suitability match any fragmentation issue is likely to be linked closely to losses in habitat area which could restrict the ability of populations to grow owing to an area based carrying capacity being reached.

For those ecoprofiles where suitability of the study area habitat resource is less well matched, fragmentation through habitat loss is likely to further compound the situation. The disintegration or dissection of habitat patches so that they are smaller than minimum areas required for a single population or for the patch to operate as habitat for a metapopulations results in a landscape with reduced functionality. This effect is shown clearly for the broadleaf and mixed woodland A and B ecoprofiles which both require habitat patches of a different format relative to those found in the landscape of the study area. In the case of broadleaf and mixed woodland B, minimum size of patch and the permeability of the landscape between patches of an adequate size are major factors, although it is acknowledged that the 20 hectare minimum patch size required could be considered a relatively ambitious requirement to meet consistently across any lowland English landscape. However, broadleaf and mixed woodland A has a much less demanding minimum area requirement and less strict permeability criteria. This highlights a third aspect of habitat fragmentation which plays a role in determining landscape functionality, the quality of remaining habitat patches. Whilst it was not possible to determine the quality of all habitats within the study area landscape the percentage of sites found in favourable condition when considering the status of SSSIs (47.7%) (Figure 3-5), identify a landscape with a significant proportion of poor quality habitat.

4.4.2 Identified ecological networks

Examination of the proportion of habitat which is part of an ecological network allows an assessment of the current level of habitat fragmentation. It can also be used to identify robust networks which are functioning well and those which could be strengthened and extended through the use of habitat banks. The number of ecological networks identified for each habitat type showed a large range: six lowland heath networks (1169 ha), 69 unimproved grassland networks (5079 ha), 93 wetland habitat networks (157,001 ha) 492 broadleaf and mixed woodland networks (591,252 ha). However, to understand which networks would benefit most from the addition of habitat banks a frequency distribution of network size needs to be examined (Figure 4-3).

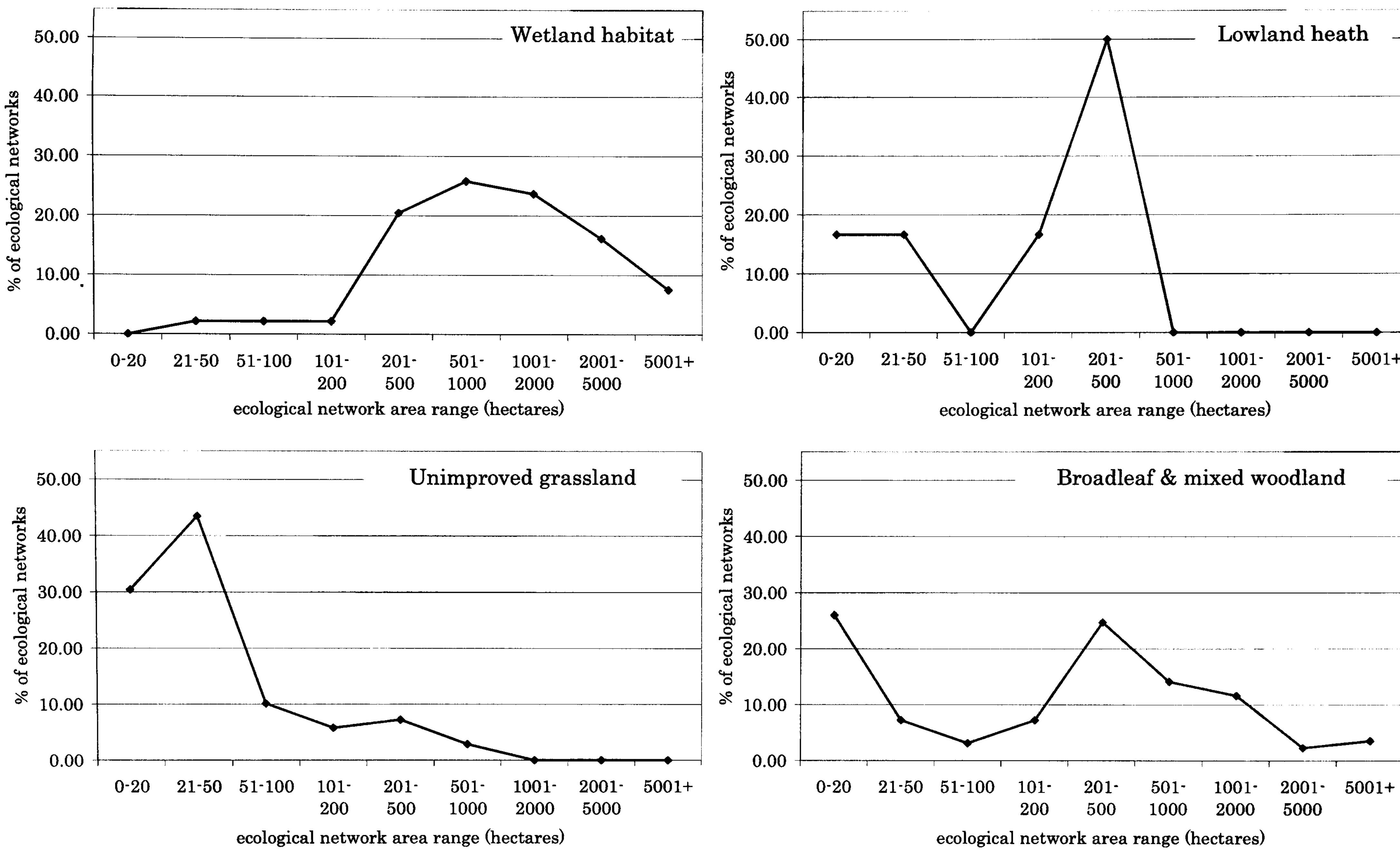


Figure 4-3 Frequency distribution of area of ecological networks

Wetland habitat networks are represented in all but one of the size categories suggesting a good range of network sizes across the landscape. These networks are strongly associated with distribution of water bodies and watercourses and as such are also relatively evenly distributed spatially. A good range of network sizes is also found for broadleaf and mixed woodland. Although just over a quarter of networks are 20 hectares or less for this habitat the same percentage is found for networks of 201-500 hectares making it of less concern. Lowland heath and unimproved grassland networks, however, display a less evenly distributed range of network sizes. Half of all lowland heath networks are less than 200 hectares. Also of concern is that only six networks were found and these showed a wide spatial distribution suggesting that consolidation of the resource would be difficult. The unimproved grassland networks identified show a negative correlation with size. As only 69 networks are found in the study area, and 74% of these are less than 50 hectares, a requirement for strengthening is clearly indicated.

All habitat types have some of their ecological networks in the lowest size ranges and therefore support the perceived need to strategically invest in the landscape in the form of habitat banks. Habitat bank creation is likely to produce a more quantifiable effect for wetland habitat and broadleaf and mixed woodland networks owing to the number of networks in the landscape and their even spatial distribution. However, habitat banks which contribute to the strengthening of lowland heath and unimproved grassland networks are likely to have a positive qualitative result owing to the currently limited networks identified for theses two habitats and the large distances between neighbouring networks.

4.4.3 Factors affecting habitat bank locations

Identification of appropriate locations for habitat banks was based on development locations, habitat bank creation budgets, existing natural and anthropogenic constraints and ecological network locations. It was determined that only six JCAs would be directly affected by development, these being clustered in the south, east and central parts of the study area. Although, it is recognised that development would be likely to have an impact on a wider area and possibly other JCAs, the six JCAs formed the focus area for identifying habitat bank locations.

The preferred sizes of habitat banks were considered from two perspectives cost of land acquisition, and optimum ecological size. The cost per hectare of acquiring different types of land was examined to understand how economies of scale may have an influence on this aspect of habitat bank creation costs (Figure 4-4).

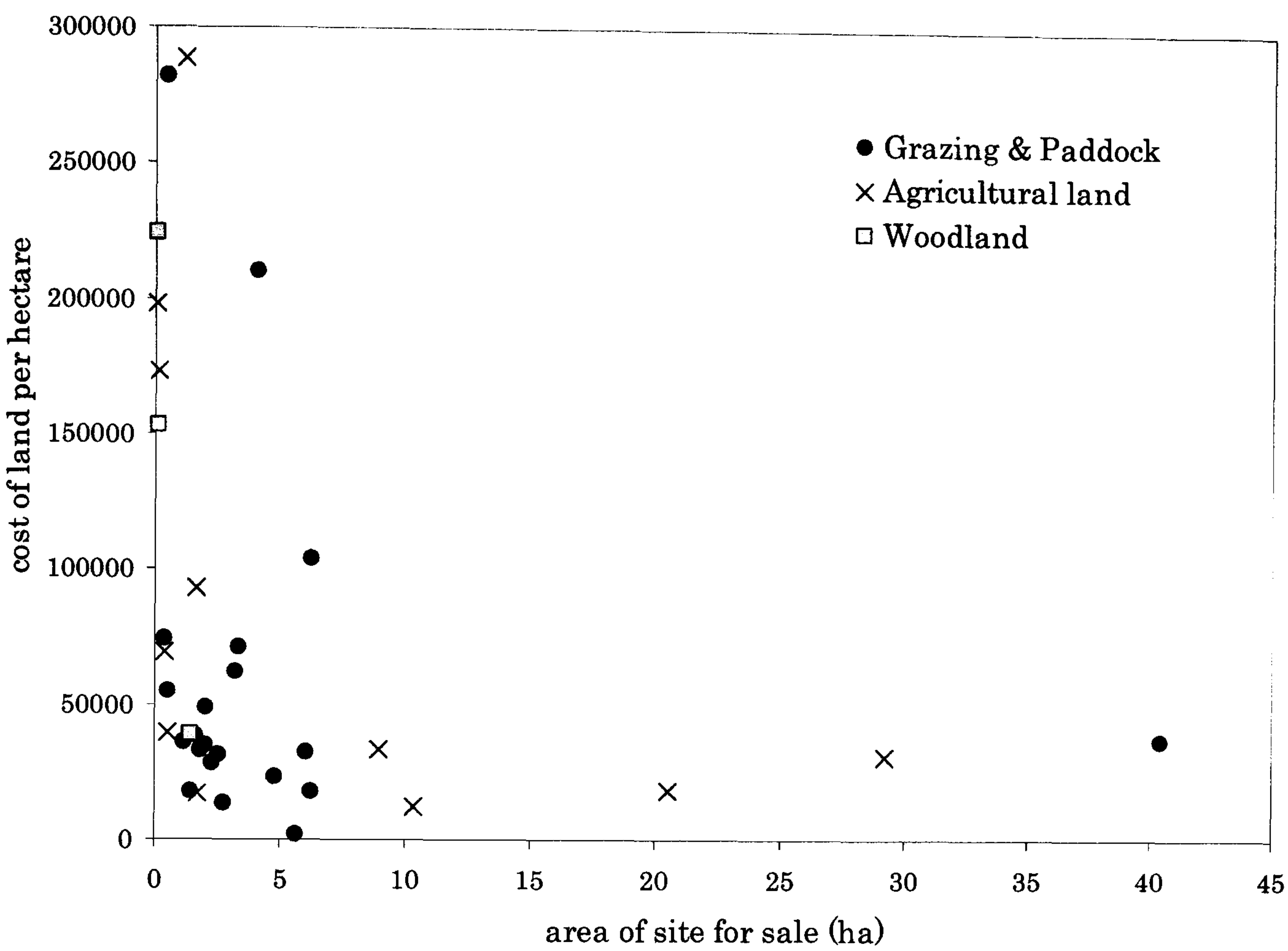


Figure 4-4 The effect of site size on cost per hectare of example acquisition land

It is suggested that habitat banks (particularly wetland habitat, lowland heath and unimproved grassland), should be located on larger sites which are currently classified as agricultural, grazing or paddock. However, sites currently identified as woodland should not be discarded as they may offer an opportunity to achieve a woodland habitat bank through conservation, restoration and management of the existing habitat rather than through the creation of new habitat. Under both scenarios other economic, social and environmental considerations as outlined previously need to be used to ultimately determine the most appropriate sites.

Ecologically optimum sites are generally agreed to be areas where populations are likely to persist in the future, *e.g.* highly connected landscapes, areas with high actual and effective populations and metapopulations, and areas with a high likelihood of further colonisation (Hanski and Ovaskainen, 2000; Moilanen *et al.*, 2005). However, a consensus on an optimum site area is harder to reach. Whilst it is accepted that in general a larger site is more likely to be beneficial to a larger number and range of species, such a site needs to have a high level of connectivity with the surrounding landscape in order to play a role within a network or provide suitable habitat for a metapopulation. It is therefore suggested that larger sites be preferred over smaller sites for habitat banks, however, size of bank should not be the only criterion used to determine ecological suitability.

4.4.4 Proposed habitat banks

Potential habitat bank sites are quickly exhausted for lowland heath and unimproved grassland, suggesting a limit to the reversal of fragmentation that could be achieved for these habitats. Conversely, sites suitable for wetland habitats and broadleaf and mixed woodland far exceeded the habitat bank area it would be financially possible to create, allowing multiple habitat bank scenarios and locations to be considered (Figure 4-5). A strong clustering of potential habitat banks can be identified for wetland habitats and broadleaf and mixed woodland focused on existing watercourses for the former and existing areas of extensive woodland for the latter. This highlights how habitat banks can contribute to strengthening the existing habitat resource within the landscape. It is not possible to determine whether a similar effect occurs for lowland heath and unimproved grassland habitat banks owing to the paucity of potential sites identified. The habitat bank locations identified also suggest possible locations and combinations of multi-habitat banks containing adjacent areas of differing habitat. This would enable a mosaic habitat to be developed which would be more likely to benefit a large range of species and take into account the life cycle and breeding habitat requirements of a number of ecoprofiles. Three multi-habitat bank types were determined to be available: wetland habitats-broadleaf and mixed woodland (2308 ha), broadleaf and mixed woodland-unimproved grassland (1361 ha) and lowland heath-broadleaf and mixed woodland (47 ha).

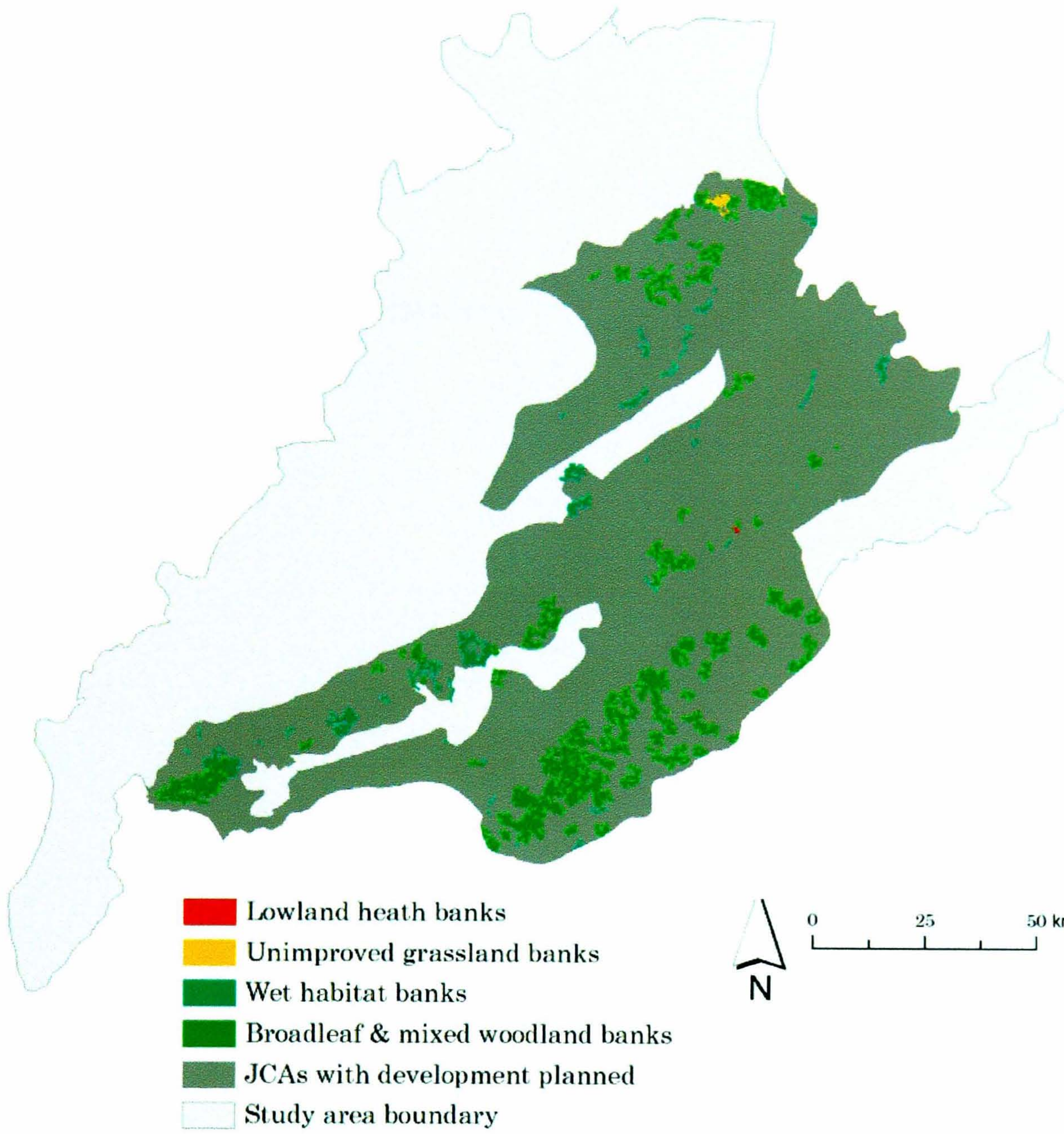


Figure 4-5 Potential habitat bank sites within JCAs with proposed development

4.4.5 The effect of habitat bank locations

To determine the validity of the proposed habitat bank locations they were reassessed in terms of their effect on ecological networks. The methodology outlined in 4.2.2 was re-employed with the habitat banks incorporated. This allowed the effect their presence would have on the landscape and their potential role in consolidating and strengthening existing ecological networks to be gauged Figure 4-6).

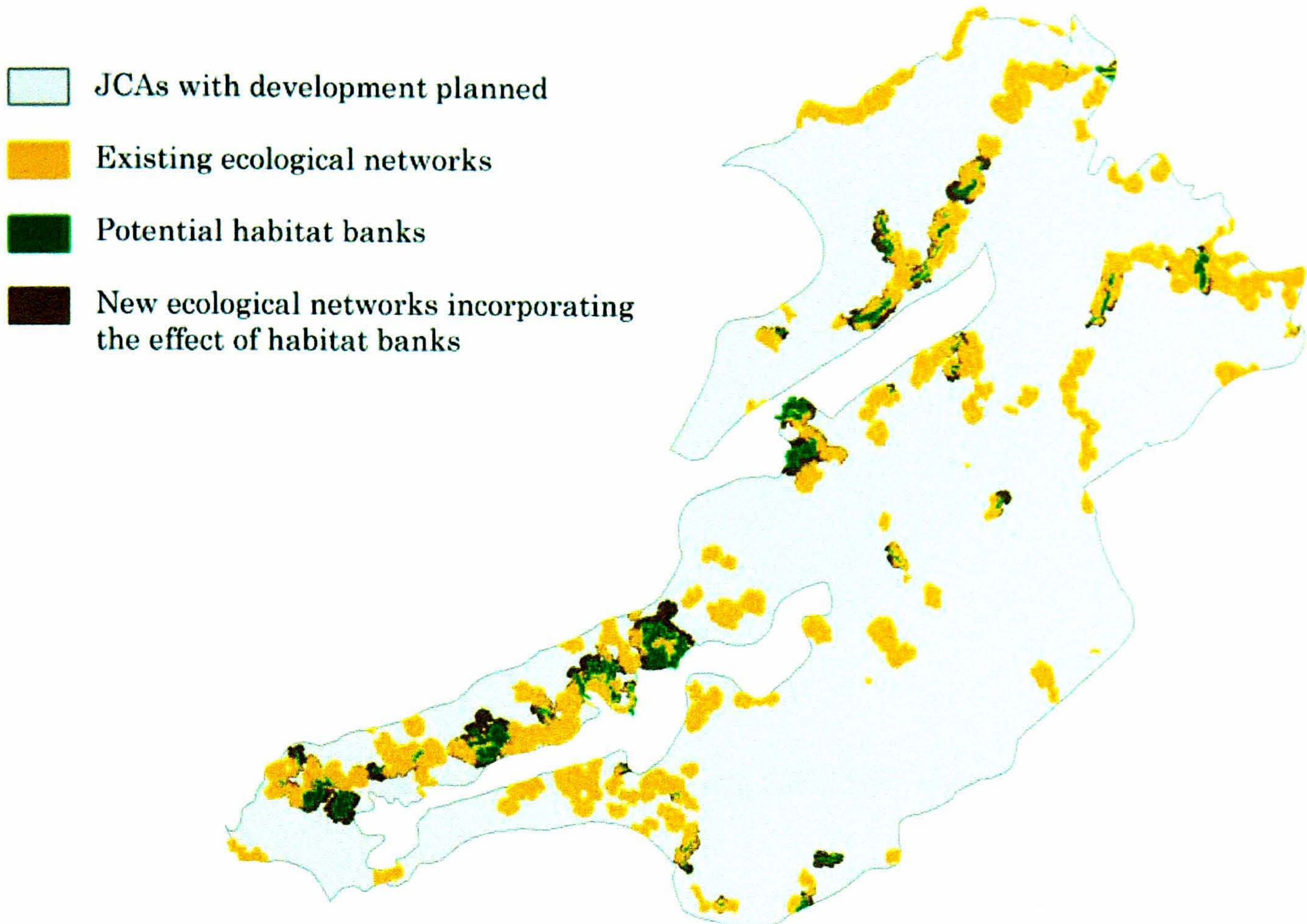


Figure 4-6a Potential habitat banks and existing ecological networks: wetland habitats

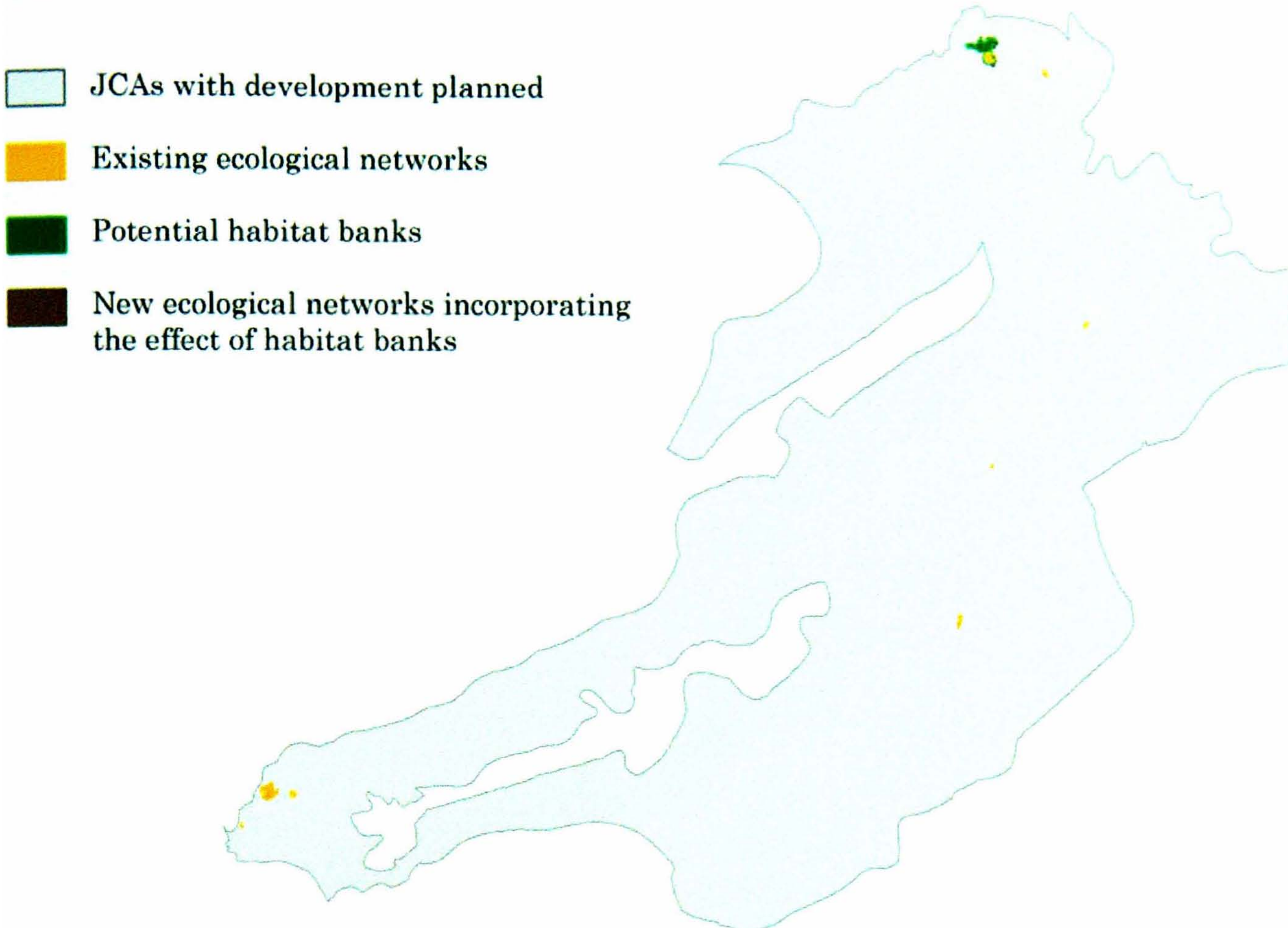


Figure 4-6b Potential habitat banks and existing ecological networks: unimproved grassland

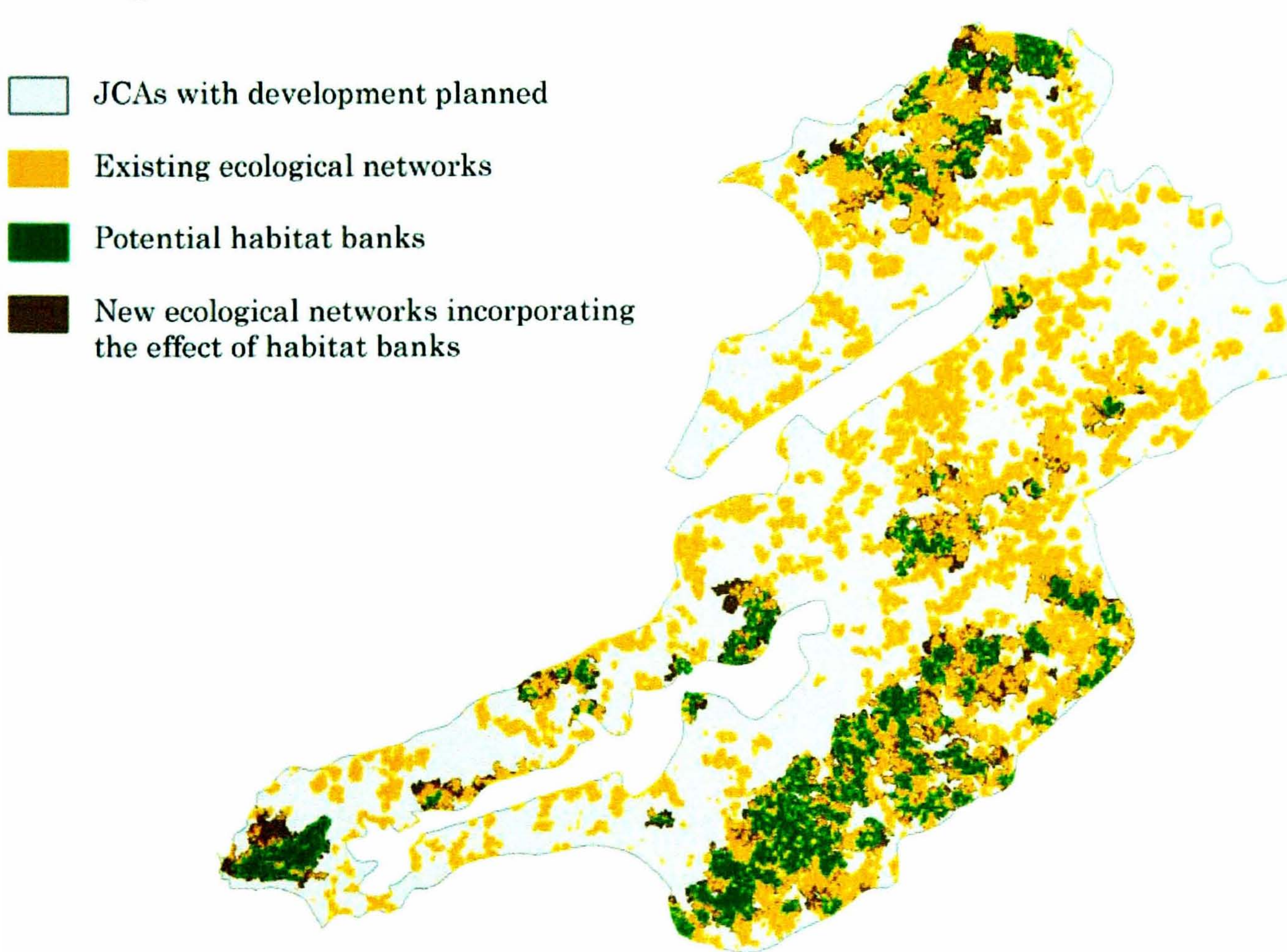


Figure 4-6c Potential habitat banks and existing ecological networks: broadleaf and mixed woodland

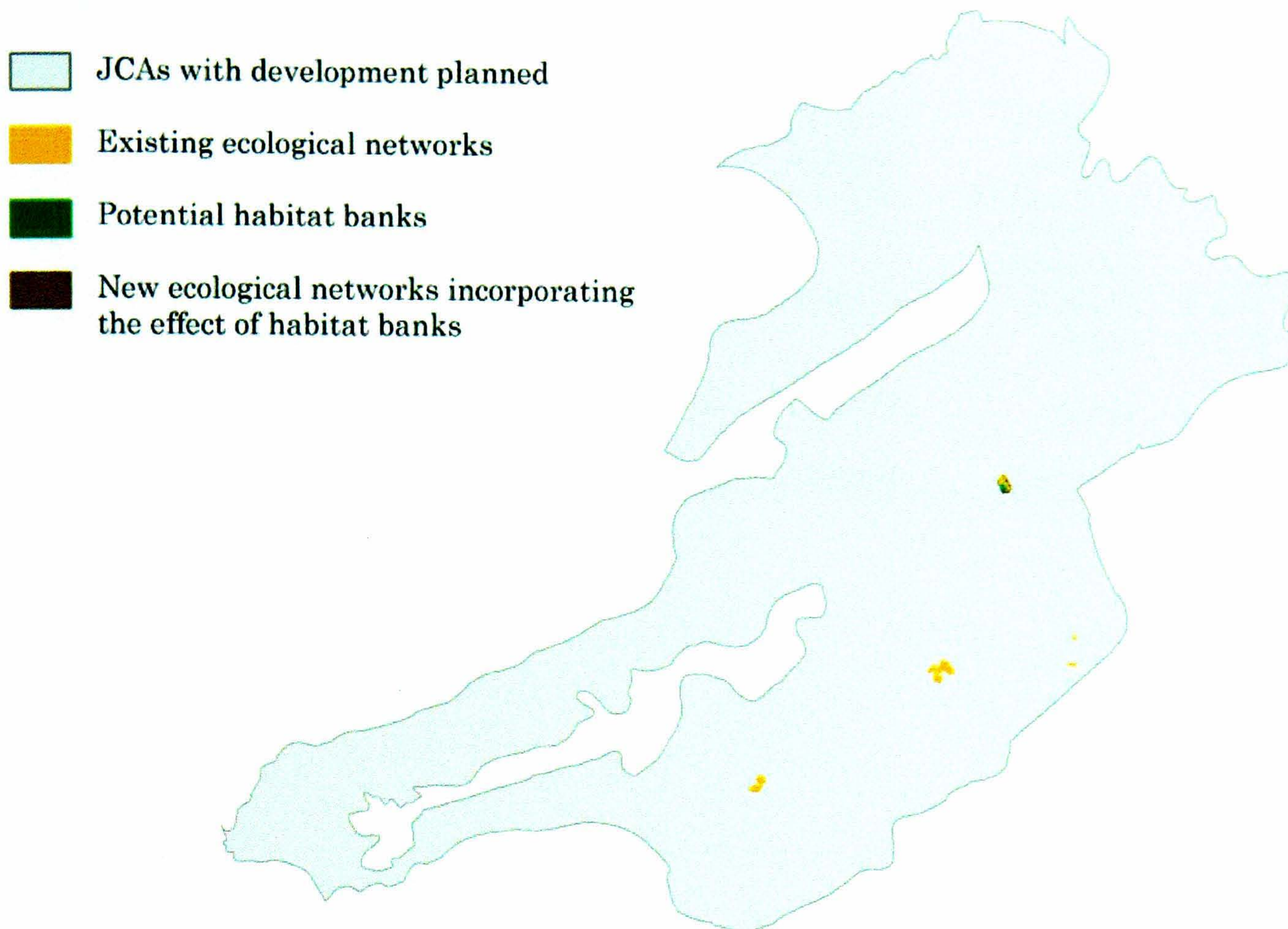


Figure 4-6d Potential habitat banks and existing ecological networks: lowland heath

The effect of the targeted habitat bank additions was positive on ecological networks of all habitat types. Wetland habitats showed an increase ratio of 1 ha habitat bank added to a 2.72 ha increase in ecological network – the highest of all habitats considered (Table 4-8). Lowland heath, broadleaf and mixed woodland and unimproved grassland, all showed an increase in ecological network area size above the habitat bank area added with ratios of 1:2.15 ha, 1:1.81 ha and 1:1.72 ha respectively.

These results highlight two important issues addressed by the methodology proposed in this research. Firstly, the importance of incorporating multiple considerations into the choice of location for habitat creation areas, and secondly, that measurement of the effect of habitat creation from a functional perspective allows a more accurate value to be placed on any intervention in the landscape.

Table 4-8 The effect of adding habitat banks to ecological networks

	Existing ecological networks (ha)	Habitat banks (ha)	Increase in ecological networks (ha)	Ecological networks incorporating habitat bank effects (ha)	Increase ratio (ha)
Wetland habitats	116315.39	8715.13	15004.65	122604.91	1: 2.72
Lowland heath	1004.59	31.09	35.77	1009.27	1: 2.15
Unimproved grassland	891.04	775.79	558.66	673.91	1: 1.72
Broadleaf & mixed woodland	293376.92	41018.54	33354.05	285712.43	1: 1.81

CHAPTER **FIVE** | Methodology and Results 2

Natural greenspace assessment and green infrastructure requirements of communities

5.0 Introduction

In areas where significant housing development is taking place there are likely to be impacts on people's access to greenspace. These impacts will be first felt by existing communities and subsequently by the new communities formed as dwellings are constructed and occupied. It is proposed that habitat banks can play a role in providing improved access to natural greenspace, linking impacts of development to the requirements of communities. It is also considered advantageous to link research carried out on locating habitat banks based on ecological network analysis and planning with the greenspace requirements of communities, so viewing the landscape as multifunctional green infrastructure. The aim is to create a stepped methodology which in combining biodiversity and recreation functions of the landscape proposes it as a green infrastructure system which can be planned sustainably. The suggested economic platform for this approach is habitat banking. A methodology has been developed which utilises funding available through the contributions of developers active in the study area. This was calculated on a per new dwelling basis as defined by the Milton Keynes roof tariff (Milton Keynes Partnership, 2006).

Although no statutory measures are in place for the provision of natural greenspace, the Accessible Natural Greenspace Standards (ANGSt) (Handley *et al.*, 2003a; Harrison *et al.*, 1995) are referred to within national planning guidance, *e.g.* Planning Policy Guidance (PPG) 17 (ODPM, 2002), and are widely regarded by local authorities and other agencies as an appropriate standard to work towards (Comber *et al.*, 2008; McKernan and Grose, 2007).

5.1 Research questions

i. What green infrastructure currently exists in the study area and how can the functionality of the resource be assessed?
ii. What level of accessibility do the current population in the study area have to the natural greenspace resource?
iii. How will the proposed developments in the study area affect green infrastructure quantitatively, qualitatively and functionally?
iv. What are the effects of introducing additional green infrastructure in the form of habitat banks on accessibility for the study area population?

5.2 Green infrastructure analysis methodology

The methodology used to examine natural greenspace and green infrastructure requirements of communities is based on a number of stages:

i. defining the existing resource,
ii. considering the population,
iii. acknowledging how changes in the landscape will affect this dynamic, and
iv. considering how habitat banks could contribute positively.

Only areas subject to development through the MKSM growth area were considered (Figure 4-5). This allows a comparison between the current situation and potential future scenarios to be examined. The focus on natural greenspace was adopted in order to fit with the locating of potential habitat banks based on spatial analysis of ecological networks.

5.2.1 Existing green infrastructure

Identifying greenspaces which make up the existing green infrastructure resource is often carried out using the Urban Greenspaces Taskforce greenspace typology (Community Forests North West, 2005; Davies *et al.*, 2006; East Midlands Regional Assembly, 2006; NW Green Infrastructure Think Tank, 2008; TEP *et al.*, 2005; Town and Country Planning Association, 2004) which defines nine categories of green and civic space (DTLR, 2002). However, it was determined that only those categories which contained greenspaces of a 'natural' type, *i.e.* areas consisting of semi-natural or unimproved habitat with an inherent biodiversity function would be incorporated into the developed methodology. Four greenspace categories: parks and gardens, amenity greenspace, natural and semi-natural greenspace and green corridors, were identified as relevant to the study area. A more detailed breakdown of the four categories (Table 5-1) was made and their extent examined using a GIS.

Table 5-1 Accessible greenspace categories used in the assessment methodology

Greenspace category	Greenspace detail	Dataset reference	
Parks, gardens and designated sites	registered parks and gardens local nature reserves national nature reserves Ramsar sites special area of conservation	(English Heritage, 2007; Natural England, 2006a; Natural England, 2007a; Natural England, 2007b; Natural England, 2008f)	Sites within this category were determined to be freely accessible to the public and were included in their entirety
Amenity greenspace	registered common land open countryside	(Natural England, 2007c)	Areas were designated under the CRoW Act 2000 (Acts of Parliament, 2000), and as such are freely accessible to the public and were included in their entirety.
Natural and semi-natural greenspace	woodland and scrub grassland and meadow lowland heath marsh, fens and reedbeds	(English Nature, 2001a; English Nature, 2001b; English Nature, 2001c; English Nature, 2001d; English Nature, 2002a; English Nature, 2002b; English Nature, 2002c; English Nature, 2002d; English Nature, 2002e; English Nature, 2003b; English Nature, 2004a; English Nature, 2004b; English Nature, 2004c)	Sites were overlaid with public rights of way and where the two intersected the site was viewed as accessible
Green corridors	public right of way (ROW) long distance footpaths tracks and minor roads	(Bedfordshire County Council, 2007; Buckinghamshire County Council, 2007; Cambridgeshire County Council, 2008; Leicestershire County Council, 2008; Luton Borough Council, 2007; Ordnance Survey, 2006; Ordnance Survey, 2008)	Access was considered from the perspective of people on foot only. This ensured public transport routes or people's access to private transport had no influence

Note: SSSIs were not included in the accessible natural greenspace data as there is no consistent policy of accessibility at these sites. However, these sites were captured under the 'semi-natural habitat' category where access was available.

5.2.2 Functional assessment of green infrastructure

Understanding the multifunctionality of green infrastructure allows its potential to contribute to sustainable communities to be determined. Approaches to describing functions and benefits of green infrastructure vary in their scale and focus. Numerous approaches (Barber, 2005; NW Green Infrastructure Think Tank, 2008) identify broad-scale functions which are suited to considering green infrastructure from a policy perspective. However, in developing a methodology to assess green infrastructure functionally in a nominated study area with a specific landscape trajectory, *i.e.* impending large scale and prolonged development, a more detailed and pragmatic interpretation of green infrastructure functions is required. A focus on the social functions complements the ecological and economic approach carried out in the earlier part of the research. Twenty functions can be identified which fall within this social remit (Coles and Caserio, 2001), and they can be grouped into seven themes: quality of life, health and well-being, education and life skills, community and local identity, security and social inclusion, recreation and sport, and support management and finance (Figure 5-1).

Assessment of the accessibility of natural greenspace is seen as one method of determining people's ability to experience the twenty identified functions. It also allows a strong spatial analysis element to be incorporated which fits with the research aim of locating habitat banks. The study area was examined to determine compliance with the Accessible Natural Greenspace Standards (ANGSt):

Rule 1: 2 hectares of accessible natural greenspace within 300 m
Rule 2: 20 hectares of accessible natural greenspace within 2 km
Rule 3: 100 hectares of accessible natural greenspace within 5 km
Rule 4: 500 hectares of accessible natural greenspace within 10 km

(Handley *et al.*, 2003a)

Two scales of enquiry were considered, a coarse scale which included the JCAs within the study area where development had been identified and a more focused area determined by a 10 km buffer placed around all development sites identified by growth area assessments (Figure 5-2). It was felt appropriate to consider two study area scales owing to the emphasis placed on urban greenspace by ANGSt.

SOCIAL CRITERIA

Quality of Life			Health and Well-Being		Education and Life Skills			Community and Local Identity				Security & Social Inclusion		Recreation and Sport			Support, Management and Finance		
1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Promotion of high quality living & working conditions	Daily recreational needs	Life strategies	Physical & emotional well-being	Retreat against stress & pace of urban life	Exploration of the natural world	Understand urban environments	Educational resource	Optimise local use	Location	Community ownership	Local identity	Fear of crime/safe use	Social inclusion	Recreation network	Sports facilities	Family/community events	Management regimes	Training	Financing

Source: Adapted from (Coles and Caserio, 2001)

Figure 5-1 The social functions of green infrastructure

This approach also allowed a more localised assessment to be made of areas and communities likely to be directly influenced by proposed housing developments but remaining within the context of the overall MKSM study area.

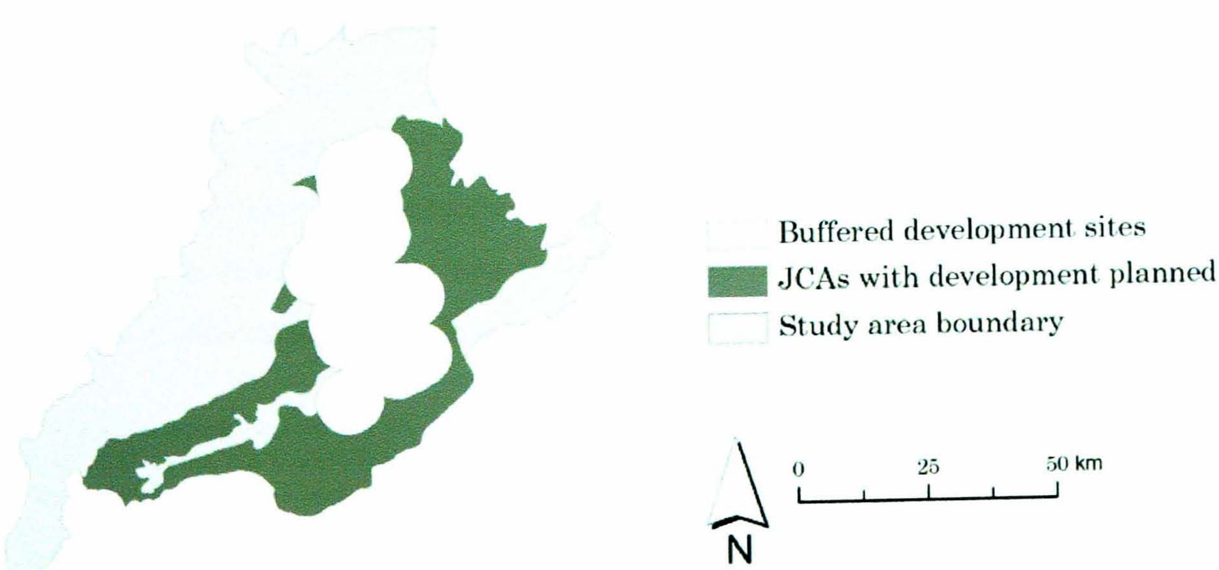


Figure 5-2 Study area scales used in the greenspace assessment methodology

5.2.3 Community access to green infrastructure

The purpose of this work was to firstly establish the current accessible natural greenspace provision at the two study scales, and secondly to establish whether targeted planning of additional greenspace provision could increase accessibility of natural greenspace for the study area population. Two approaches to assessing natural greenspace provision were considered:

i. *Buffered greenspace approach*

Using a GIS a buffer was placed around existing accessible greenspace. The size of the greenspace site and buffer represented the four ANGSt rules, *e.g.* rule 1 was represented by considering all accessible greenspace over 2 hectares and a buffer of 300 metres was applied around its perimeter. Data representing the appropriate study scale populations[1] were then added in order to assess the proportion of the population in the study area that would be able to access a natural greenspace site within a particular ANGSt rule.

ii. *Network analysis approach*

This approach was based on the use of an origin-destination cost matrix (Comber, 2008; Comber *et al.*, 2008) within a GIS. This allows the measurement of the distance between multiple origins and multiple destinations with the output detailing the total distances from each origin to each destination. Green corridors and access routes (accessible on foot)[2] within the study area were used to create a network dataset which detailed impedance to movement. An origin point dataset defining all starting points in the analysis was created, this represented where the population[1] of the study area may begin a journey to an accessible natural greenspace. A destination point dataset which identified entrance points into accessible greenspaces was then created. Separate files were created to represent each of the four ANGSt rules.

[1] The population was based on the assessment of Output Areas (OAs) in the study area. 2001 Census OAs (Office of National Statistics, 2003) are constructed from clusters of adjacent unit postcodes. They are designed to have similar population sizes and be as socially homogenous as possible - based on tenure of household and dwelling type. OAs in England and Wales have an average size of 124 households which results in a population of around 300 people (Vickers and Rees, 2007)

[2] Considering access to natural greenspace on foot raises some issues as at least two of the ANGSt recommendations are likely to require people to use public or private transport owing to the distance to sites being considered i.e. 5 km or 10km. However, availability of transport for the population could not be determined.

The two approaches used allowed different levels of detail to be examined, and also reflect the range of approaches currently being used in the application of ANGSt by local authorities and other agencies. It was considered that by comparing the results of the two assessment methods comparisons between this and studies carried out by others could be examined with the aim of proposing a suitable methodology for the application of ANGSt, a point which is yet to be adequately addressed.

In order to reflect likely changes to the study area resulting from proposed developments the population was considered in two stages, firstly the existing population and its spatial distribution and secondly the projected increased population following proposed housing development and subsequent occupation. The existing population was determined using 2001 Census OAs (Office of National Statistics, 2003) within the study area. In order to project the increase in population as a result of development, and place this potential new population in the landscape, likely development sites were identified through the MKSM growth area assessment reports (East Midlands Regional Local Government Association, 2003; East of England Local Government Conference, 2003; East of England Local Government Conference and Government Office for the East of England, 2003; South East England Regional Assembly *et al.*, 2003) produced by each local authority affected. The boundaries of these sites, number of dwellings they would contain and average household size in each local authority (Office of National Statistics, 2001) were used to calculate the population increase for each development site. This value was divided by the England and Wales OA population size average to determine the number of additional OAs which would be created as a result of development. The additional 846 OAs were located randomly within the existing OAs whose boundaries intersected the development site. The number of OAs added at each development site reflected the additional population likely as a result of the proposed development.

In order to examine the effectiveness of the greenspace assessment methodology constructed to date testing was carried out. This was based on the addition of greenspace into the study area on a random basis. The use of randomly placed additional greenspace allowed the sensitivity of the GIS modelling approach to be examined. All potential additional greenspace sites were merged into a single dataset where a random, none repeating number was allocated to each site. The sites were then ordered by the random number. It was determined that 1562ha of additional greenspace could be created at an average cost of £172,866 per ha. The financial constraints were set out by the Milton Keynes roof tariff approach which determined approximately £2600 per new dwelling to be available to spend on greenspace and landscape, providing a total available to spend of £270,042,354 (Milton Keynes Partnership, 2007). The area of each potential greenspace site was examined and a cumulative area total created until the financial resources available were exhausted. These sites were then added to the GIS and the greenspace analysis using the two assessment methods was carried out.

Following the sensitivity analysis two alternative approaches to locating additional accessible greenspace were developed, biodiversity led and recreation led. The biodiversity led approach was developed in order to implement area based targets set out in local BAPs. It utilises the habitat banks identified by the ecological network analysis as being most suitable for the extension and consolidation of existing networks within the study area. UK BAP targets were determined for lowland beech and yew woodland, lowland mixed deciduous woodland, upland oak woodland, wet woodland, lowland calcareous grassland, lowland meadows, upland hay meadows, lowland dry acid grassland, fens, reedbeds, coastal and floodplain grazing marsh, lowland heath and purple moor grassland (UK BAP Partnership, 2008). The habitat targets were determined to be:

- broadleaf and mixed woodland (create an additional 250 ha),
- lowland heath (create an additional 40 ha),
- wetland habitats (create an additional 55 ha and restore 92 ha), and
- unimproved grassland (create an additional 122 ha and restore 150 ha).

In order to identify where this additional biodiversity led greenspace would be located, the favoured locations for habitat banks determined in the ecological network assessment were used. The cost of additional greenspace was considered using the Milton Keynes roof tariff values and the land acquisition and habitat creation costs identified in Chapter four. BAP targets were matched against habitat banks of suitable types until either the available banks or finances were exhausted (Table 5-2).

Table 5-2: Biodiversity led greenspace addition areas for the two scales of assessment

JCAs with development planned					
Habitat type	BAP target (ha)	Available (ha)	Average cost (£/ha)	Greenspace added area (ha)	Greenspace added cost(£)
Wetland habitats	147	8715.13	£171,612	686.85	£117,871,702
Lowland heath	40	31.09	£170,855	31.09	£5,311,881
Unimproved grassland	272	387.66	£173,899	387.66	£67,413,686
Broadleaf & mixed woodland	250	41018.54	£175,099	452.70	£79,267,317

Buffered development sites					
Habitat type	BAP target (ha)	Available (ha)	Average cost (£/ha)	Greenspace added area (ha)	Greenspace added cost(£)
Wetland habitats	147	2556.27	£171,612	686.85	£117,871,702
Lowland heath	40	31.09	£170,855	31.09	£5,311,882
Unimproved grassland	272	0.00	£173,899	0.00	-
Broadleaf & mixed woodland	250	13839.61	£175,099	837.72	£146,683,934

With the exception of lowland heath and unimproved grassland at the smaller study scale it was possible to identify suitable habitat banks which could provide greenspace in excess of the BAP targets identified. This suggests that the use of habitat banks is well matched as a delivery mechanism for UK BAP targets.

The recreation led approach bases additions of greenspace on those habitats best suited to recreation and the largest area of additional accessible greenspace that can be achieved within financial constraints. Natural greenspace most suitable for recreation was identified using three criteria: greenspace types with a high disturbance threshold, greenspace types found most attractive by recreational users and greenspace with the best hectare per pound sterling ratio. Because of the types of species associated with lowland heath and unimproved grassland and the vulnerability of these habitat types within the study area, they were determined to be inappropriate habitats to promote as recreational natural greenspace. Wetland habitats and broadleaf and mixed woodland were identified as being more resilient to recreational use. These habitat types more frequently feature when people's use of green and open space is examined (Coles and Caserio, 2001; Tyrväinen *et al.*, 2007; Van Herzele and Wiedemann, 2003). Wetland habitats and broadleaf and mixed woodland were therefore taken forward as land cover types which should be targeted for increasing greenspace provision. The cost of greenspace creation was also considered. It was felt that the largest amount of greenspace by area should be planned in line with the potential funding available. Using land acquisition and habitat creation costs derived for the MKSM study area (Bailey and Thompson, 2007), average costs were determined to be £171,612 for wetland habitat and £175,099 for broadleaf and mixed woodland per hectare. Using the same financial approach as the biodiversity led greenspace increase it was determined that 1530 ha of additional greenspace, 760 ha of wetland habitats and 770 ha of broadleaf and mixed woodland could be created. The spatial location of this additional greenspace was guided by locations of habitat banks identified through ecological network assessment. The percentage of the study area within each JCA was determined. The same percentage was then applied to the wetland habitat and broadleaf and mixed woodland to be located. Sites were found within the appropriate JCA until either all locatable greenspace had been allocated or potential sites had been exhausted, in the latter case the adjacent JCA was then searched for available sites.

In order to accommodate all combinations of the study scales, assessment methods, population changes, testing, greenspace addition approaches and ANGSt rules, multiple scenarios were developed (Figure 5-3). Each scenario was run using the appropriate separate network, origin and destination datasets.

5.3 Results

Through the consideration of social functions of green infrastructure and the level of that functionality in the study area as determined by accessibility assessment, the location of habitat banks with a multifunctional purpose can be identified. However, first examining how alternative natural greenspace addition scenarios influence accessibility for the study area population ensures that the methodology developed is suitable and complements the environmental based assessment carried out in chapter four.

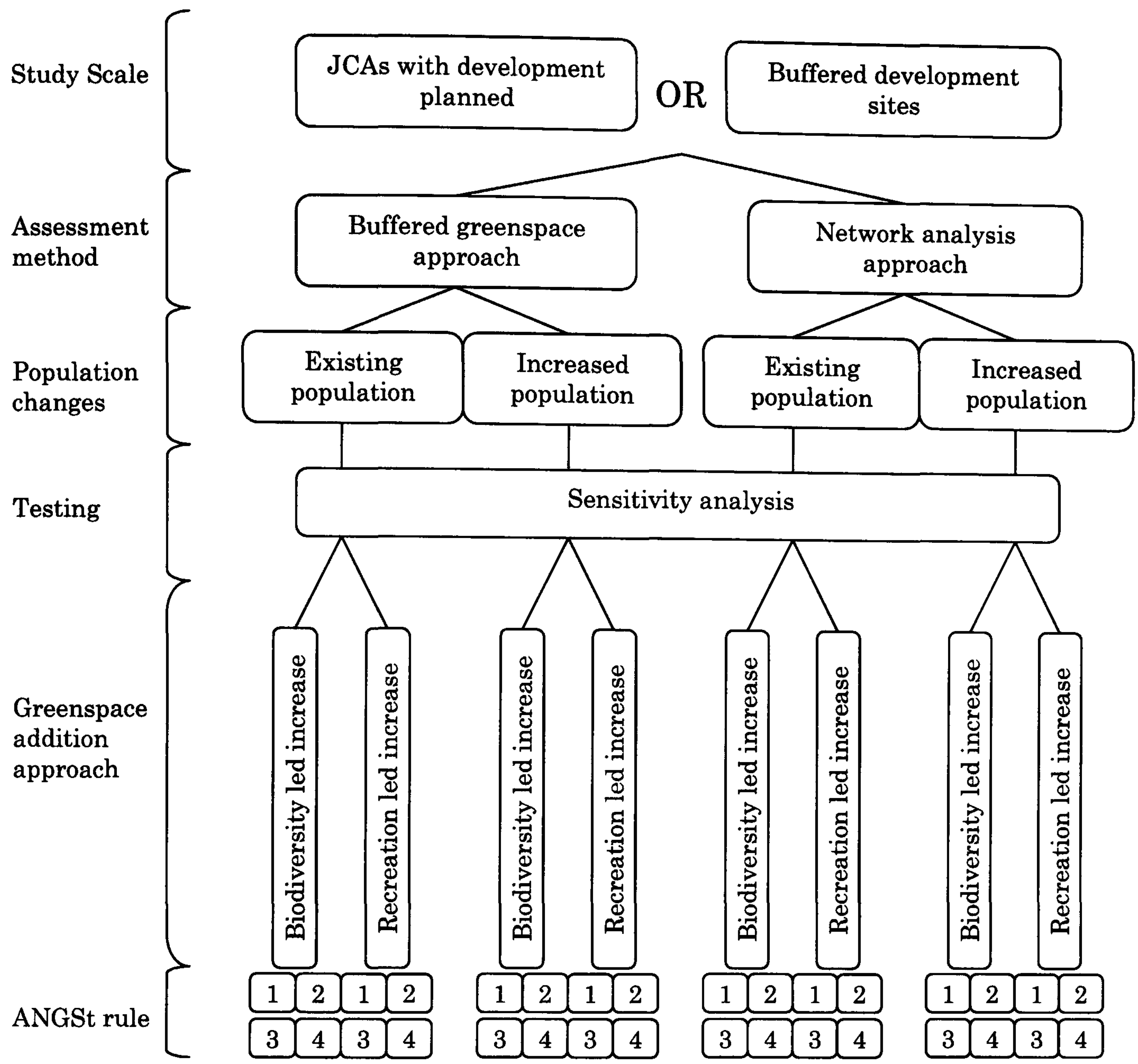


Figure 5-3 Parameters used to create alternative accessible natural greenspace scenarios

5.3.1 Accessible green infrastructure

The four categories of greenspace which comprise the accessible portion of the study area green infrastructure resource account for 9.7% (161,645 ha) of the overall study area, 12.8% (99,854 ha) of the JCAs subject to development, and 12.7% (44,781 ha) of the buffered development sites. However, both the quantitative and spatial distribution of the categories and types show an uneven spread with some types such as woodland and scrub making up around 40% of the overall resource (Figure 5-4), and therefore likely to be used more frequently by the population.

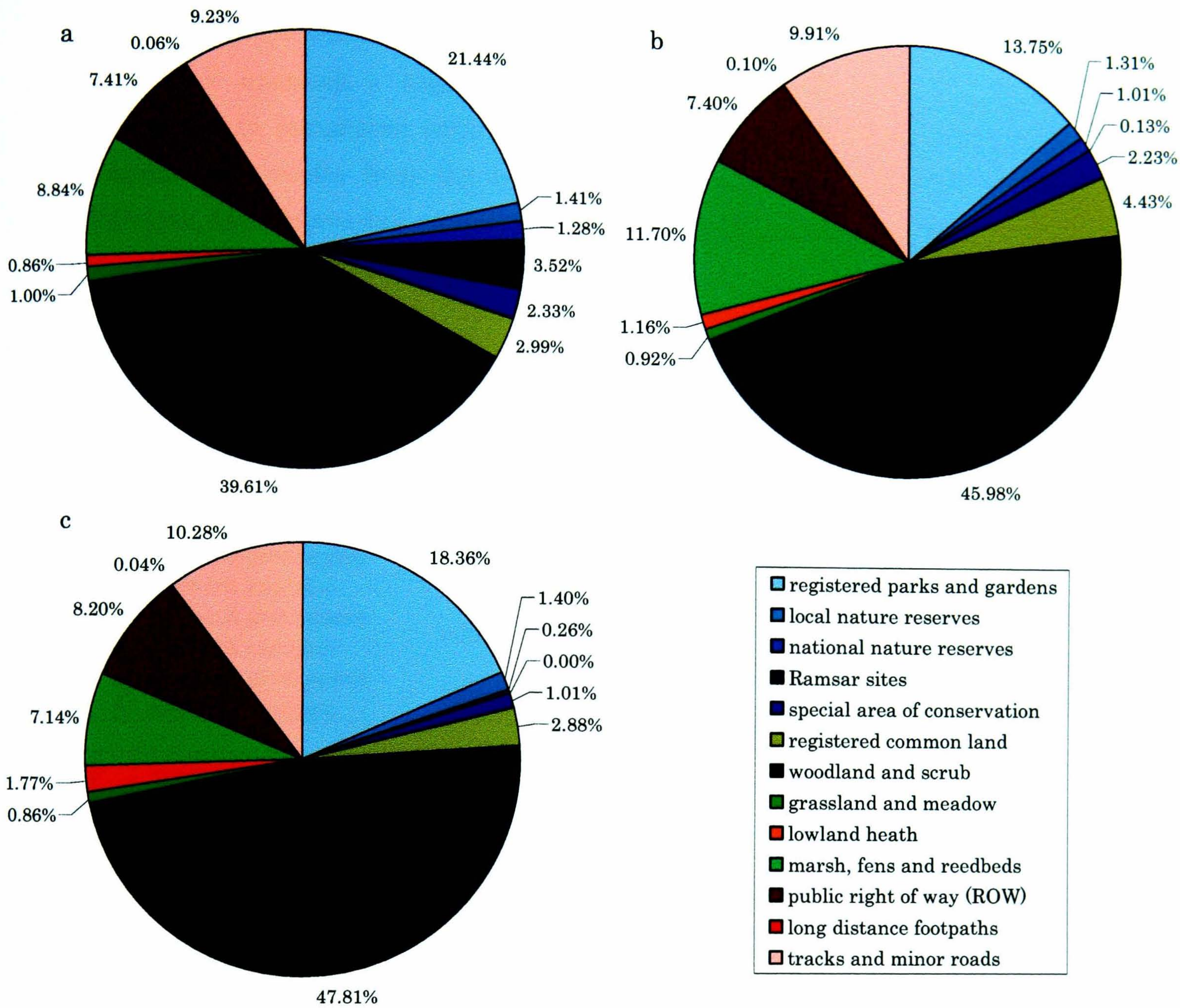


Figure 5-4 Prevalence of accessible greenspace types at the three study scales: a) full study area, b) JCAs with development planned and c) buffered development sites

Although these areas of greenspace have been determined to be potentially accessible for the population they need to be assessed using the buffered accessible greenspace and network analysis in order to determine more accurately the population's level of accessibility from the perspective of ANGSt. The two methods result in apparently very different levels of accessibility (Table 5-3). The pattern of accessibility found with regard to the four rules proposed in the ANGSt highlights differences between the two assessment approaches. The buffered greenspace approach determines a higher proportion of the population to have access to 20 and 100 hectare sites (rules two and three), whereas the network analysis approach identifies that more of the population has access to 2 and 20 hectare sites. There is also a difference when the two study scales are considered. Using the buffered greenspace approach a higher percentage of the population enjoy accessible greenspace in compliance with ANGSt at the buffered development site scale, whereas the results from the network analysis approach show the opposite. However, both approaches confirm that ANGSt are not currently met for all of the population at either study scale.

Table 5-3 The current accessible greenspace situation at the two study scales

	Assessment method	ANGSt rule	Origin points	Destination points	origins complying number	%
JCAs with development planned	buffered greenspace	rule 1	12451	2774	2068	16.61
	network analysis				507	4.07
	buffered greenspace	rule 2	12451	2774	7694	61.79
	network analysis				456	3.66
	buffered greenspace	rule 3	12451	2774	7713	61.95
	network analysis				151	1.21
	buffered greenspace	rule 4	12451	2774	5062	40.66
	network analysis				19	0.15
Buffered development sites	buffered greenspace	rule 1	4781	791	667	13.95
	network analysis				89	1.86
	buffered greenspace	rule 2	4781	791	3442	71.99
	network analysis				110	2.30
	buffered greenspace	rule 3	4781	791	3204	67.02
	network analysis				52	1.09
	buffered greenspace	rule 4	4781	791	2710	56.68
	network analysis				12	0.25

When the level of accessibility set out by the ANGSt is considered spatially the differences in the two assessment approaches can be clearly identified (Figure 5-5). The buffered greenspace approach is a coarse approach and does not consider how the population may be restricted in their movement through the study landscape. This results in the majority of the study area fitting into at least one ANGSt rule and therefore a higher percentage of the population seemingly having access to natural greenspace. The network analysis approach is more realistic and as such produces a relatively restricted result. However, the latter approach allows spatial patterns to be identified more readily, *e.g.* the large areas of accessible woodland in the South East of the study area.

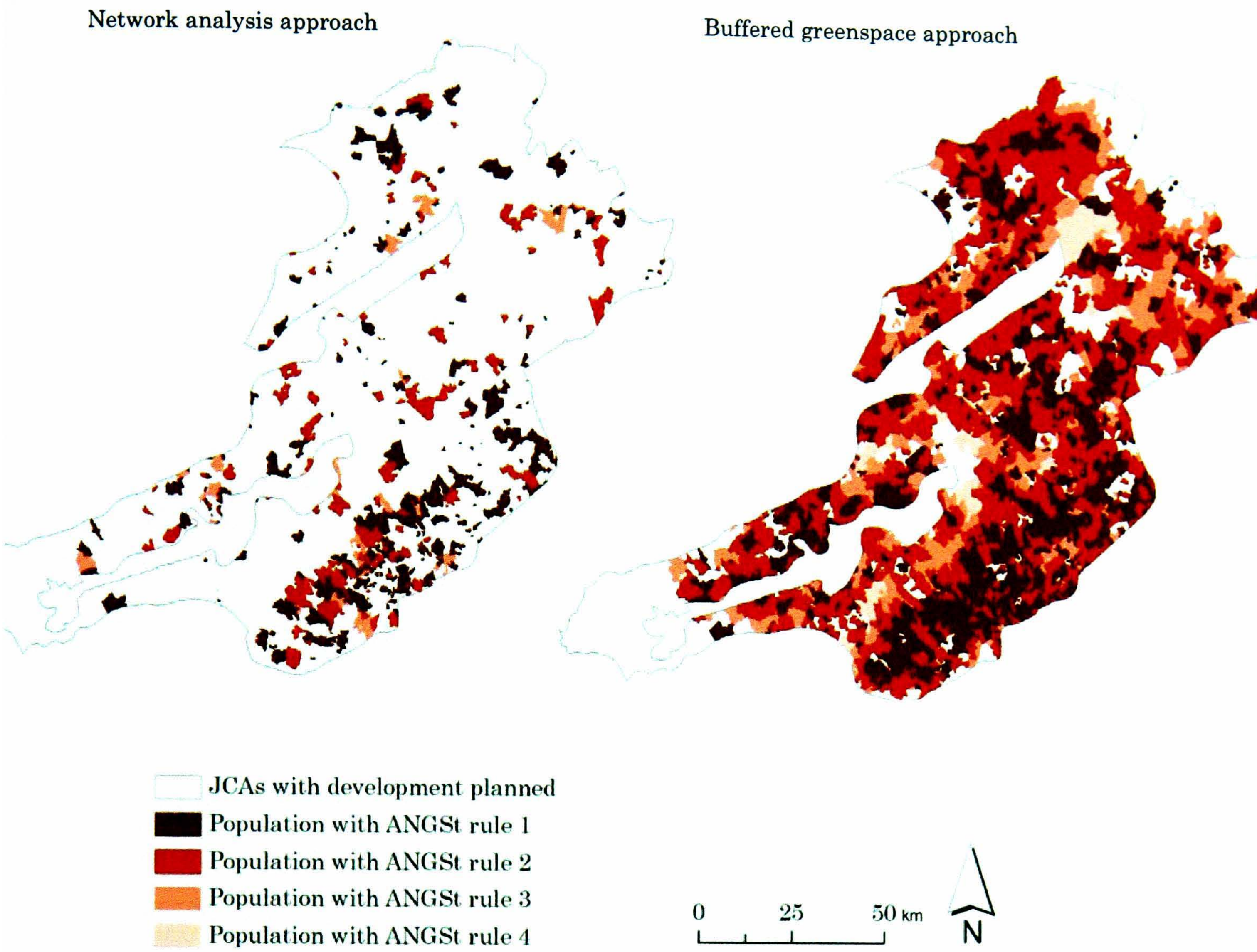


Figure 5-5 Spatial distribution of the human population with access to natural greenspace.

5.3.2 Factors affecting accessibility levels of green infrastructure

By focusing on the part of the study area where development is planned a number of factors can be identified that will have a negative influence on the percentage of the population with access to natural greenspace inline with the ANGSt. These factors can be divided into two groups, population changes and landscape changes. The increased population of approximately 250,000 will not only have an effect on the percentage of the population with access to greenspace but also the density of the population using accessible natural greenspace sites. Whilst the effect on the percentage of the population with access to greenspace can be calculated and considered spatially, the density of population use cannot be measured without surveying the existing use of individual sites and extrapolating preferences for site size and type from the existing population to the additional population resulting from developments. Therefore, the former measure will form the focus of this research. The effect of increase in population does not show a simple relationship (Figure 5-6). Using the buffered greenspace approach results in either very similar (JCAs with development planned scale), or a slight increase (buffered development locations), in the percentage of the population having access to natural greenspace (average increase of 1.8%). The results from using the network analysis assessment method are more readily visible with a slight decrease (JCAs with development planned scale), and clear decrease (buffered development locations), in the percentage of the population with access to natural greenspace found (an average decrease at the buffered development site study scale of 0.5%).

Where development is planned on greenfield and currently open brownfield sites a quantifiable impact will be felt within the landscape resulting from the net loss of open space. Whilst it is conceded that not all of this land would currently be publicly accessible it is likely that it will contribute to the identifiable social functions of green infrastructure. The development sites identified in the study area cover approximately 9,335 hectares of which 280 hectares are identified as being part of the accessible greenspace resource being considered. Although this is only 1.1% of the total accessible greenspace resource in the buffered development sites study scale the location of these impacted sites (close to existing and proposed increases in population) make this apparently small percentage significant. Readily accessible greenspace sites, *i.e.* those in closest proximity to populations, are identified as being of prime importance for the fulfilment of the social functions of greenspace (Balram and Dragićević, 2005; Coles and Caserio, 2001). The removal of such sites, subsequent change of land use from open space to built development and associated increases in population results in an exacerbation of this issue. Whilst development planned to adjoin existing conurbations is practical from the perspective of fitting into grey infrastructure and existing services the use of accessible greenspace for such development sites cannot be considered a sustainable approach irrespective of whether that land has formal designation or not.

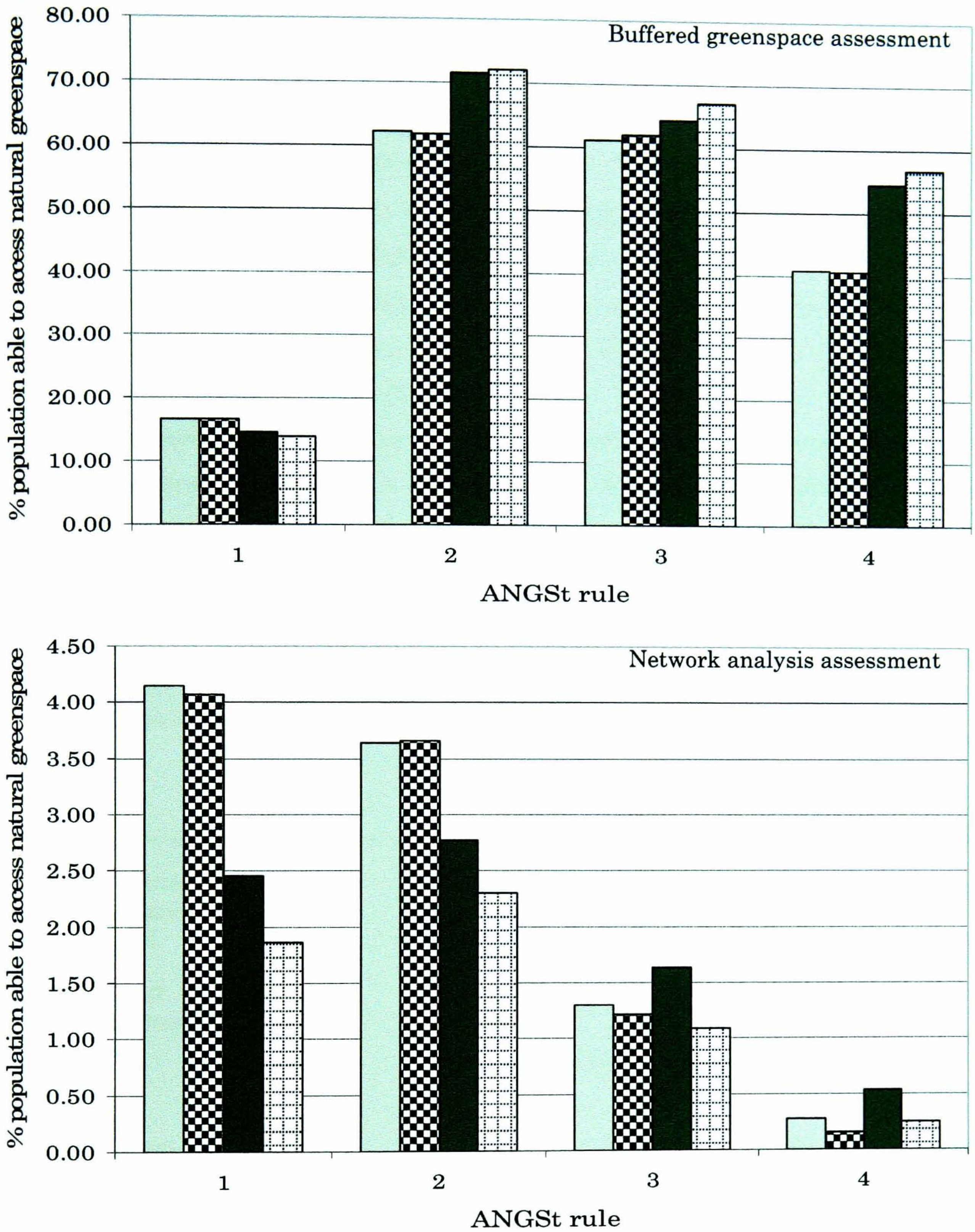
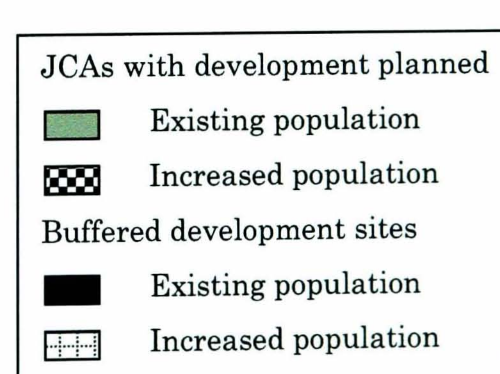


Figure 5-6 The effect of population change on fulfilment of ANGSt



5.3.3 The effects of habitat banks

Considering the introduction of additional accessible natural greenspace in the form of habitat banks from two perspectives, biodiversity led and recreation led, allows a multifunctional approach to greenspace location to be examined. When compared against the baseline of existing accessible greenspace it is possible to determine that the addition of greenspaces in all cases leads to an increase in the percentage of the population able to access greenspace as set out by ANGSt (Table 5-4). However, larger increases in percentage are seen for rules two and three, which is indicative of the size of the habitat banks added to the accessible greenspace resource. Whilst the percentage increases seen in all combinations of greenspace strategy, study scale and assessment type, appears relatively low when this is used to represent the actual population noteworthy increases of up to 12,000 people are identified.

Table 5-4 The effect of adding greenspace to the landscape on the percentage of the human population able to access natural greenspace at the level specified in ANGSt.

		Assessment type	ANGSt Rule 1 population increase %	ANGSt Rule 1 population increase number	ANGSt Rule 2 population increase %	ANGSt Rule 2 population increase number	ANGSt Rule 3 population increase %	ANGSt Rule 3 population increase number
Biodiversity led greenspace increase	JCAs with development planned	Buffered greenspace	0.10	3600	0.32	12000	0.00	0
		Network Analysis	0.06	2100	0.09	3300	0.00	0
	Buffered development sites	Buffered greenspace	0.25	3600	0.84	12000	0.00	0
		Network Analysis	0.15	2100	0.23	3300	0.00	0
Recreation led greenspace increase	JCAs with development planned	Buffered greenspace	0.08	3000	0.10	3900	0.32	12000
		Network Analysis	0.05	1800	0.03	1200	0.02	600
	Buffered development sites	Buffered greenspace	0.21	3000	0.27	3900	0.84	12000
		Network Analysis	0.13	1800	0.08	1200	0.04	600

Note: The results for rule 4 have been omitted as no change in the percentage of the population complying with this rule were found in any of the combinations of assessment. This is due to the unavailability of habitat banks added being of an adequate size i.e. 500 hectares or over.

There is a clear difference in the outcomes of the two approaches to increasing greenspace with the recreation led approach resulting in a much larger increase in the population able to access natural greenspace. One simplistic reason for this is the quantitative increase in accessible greenspace this strategy allowed, 1,377 hectares compared to 1,106 hectares using the biodiversity led approach. The decision as to whether the additional increase in the population able to access natural greenspace based on recreation, needs to be considered in light of i. the overall objective of achieving multifunctional habitat banks and ii. the wider concept of sustainable landscape planning.

CHAPTER SIX | Methodology and Results 3

Impacts of climate change on spatial landscape strategies

6.0 Introduction

The detailed manifestation of climate change in the UK remains uncertain although it is now acknowledged that changes to the structure and function of the landscape are inevitable and will lead to new challenges for biodiversity conservation (EEA *et al.*, 2008). Impacts on species are already being felt with changes in phenology, species ranges and species abundance recorded (Walmsley *et al.*, 2007; Walther *et al.*, 2002). Examining and predicting the extent and pattern of future climate derived ecological change should therefore be considered in landscape planning and strategy development; without changes being anticipated then appropriate landscape adaptation cannot be undertaken (Berry *et al.*, 2002; Hannah *et al.*, 2002). The role of connectivity in ensuring landscapes remain viable despite changes in climate is considered to be of prime importance (Vos *et al.*, 2008). It is widely suggested that habitat fragmentation can exacerbate climate change impacts by reducing the ability of species to adapt, particularly through the shifting of geographic ranges (Best *et al.*, 2007; Travis, 2003). Prior to the development of landscape strategies which incorporate adaptation to climate impacts the response of species to landscape structure and function under alternative climate scenarios must be examined. A model which incorporates climate change scenarios alongside ecological system data, *e.g.* habitat use, colonisation and dispersal abilities allows potential impacts to be investigated and likely scenarios to be explored.

Linking together climate scenario modelling with other parts of this research allows an overview of the function of green infrastructure in the study area to be achieved. Examining the current structure and function and linking this to potential future demands on, and changes to, green infrastructure allows a spatial strategy for the location of habitat banks to be developed. The results from all three areas of research are linked by ecological baseline data. If ecological networks are determined to be the frame upon which other types of green infrastructure depend then ensuring this landscape strategy is underpinned by ecological baseline data should mean a valid result is achieved.

6.1 Research questions

i. How might climate change scenarios affect current habitat and landscape function?

ii. Can landscape functionality be retained under changing climate scenarios?

iii. How could habitat banks contribute to retention and safeguarding of functionality?

iv. Can an optimum strategy be developed which incorporates the functionality required in the landscape economically, socially and environmentally?

6.2 Climate change impact methodology

The methodology developed to determine the likely impacts of climate change and how these can be incorporated in a strategy for the landscape is based on a chain of models. Each model addresses specific pressures and demands on the landscape: climate changes, existing use of the landscape and changes to availability and use of habitat.

6.2.1 The scale of data

Climate change is monitored and measured by many organisations worldwide. However, in order to understand patterns of change a large scale perspective must be taken. Climate is the characteristic weather conditions of a country or region; the prevalent pattern of weather in a region throughout the year, in respect of variation of temperature, humidity, precipitation, wind etc., especially as these affect human, animal, or plant life (The Oxford English Dictionary, 2009). The European scale is therefore appropriate for the analysis of climate change impacts on potential species distributions as well as being compatible with existing sources of species information (Harrison *et al.*, 2006). Using raw climate data at a more detailed scale such as the MKSM study area would result in an over reliance on changes to weather, *i.e.* the condition of the atmosphere at a given place and time with respect to heat or cold, quantity of sunshine, presence or absence of rain, hail, snow, thunder, fog, violence or gentleness of the winds etc. (The Oxford English Dictionary, 1989), rather than climate. The European scale allows an overview to be established with a more detailed picture being achievable for North West Europe, biogeographic regions or even the UK as a whole. However, in considering how results may impact on landscape function, qualitative conclusions can be drawn from the European and UK scales for smaller scale areas such as the MKSM study area. In fact, incorporation of such generalised results into analysis of the MKSM landscape is necessary if the proposal of habitat banks as a landscape adaptation mechanism is to be considered valid.

6.2.2 Climate change scenarios

The Hadley Centre, using a Global Climate Model (GCM) (`HadCM3`), along with the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES), have simulated a range of potential changes in climate. Using this the UK Climate Impacts Programme (UKCIP) have identified for the three time periods 2020, 2050 and 2080, four climate change scenarios (UKCIP02); low emssions (B1), medium-low emissions (B2), medium-high emissions (A2) and high emissions (A1F1). These scenarios represent temperature rises of between 0.79°C (2020s Low) and 3.88°C (2080s High) (Hulme *et al.*, 2002). In order to determine the effects of climate change on the landscape it was necessary to relate these data to landscape structure and function. The approach of 'climate space' or 'climate envelope' modelling has been widely used for this purpose (Bakkenes *et al.*, 2002; Baselga and Araújo, 2009; Berry *et al.*, 2002; Brooker *et al.*, 2007; Davis *et al.*, 1998), and through sequential use of multiple scenarios allows temporal changes to considered. The climate envelope encompasses the spatial extent of a species range both physically and climatically. Sequential climate envelopes allow required species range shifts as a result of climate change to be identified.

An existing model, Spatial Estimator of the Climate Impacts on the Envelopes of Species (`SPECIES`) (Pearson *et al.*, 2002), identified the current climate envelopes of 32 UK BAP species using European distribution and baseline bioclimatic data

(Berry *et al.*, 2007). This led to the publication of a probabilistic climate suitability surface for each species and, by incorporating future soil water availability, growing degree days and temperature indices, future potential climate space suitability surfaces were also developed (Walmsley *et al.*, 2007). Seven of the BAP species modelled using SPECIES were also incorporated into the ecoprofiles developed for the MKSM study area making it possible to use the climate envelope results as indications of how ecoprofiles previously used in this research may be impacted by climate change. The SPECIES approach is based on an Artificial Neural Network (ANN), a type of model which, it is suggested, produces generally more accurate predictions of species range shifts than other model types such as generalized linear models (GLM), generalized additive models (GAM), and classification and regression tree analysis (CART) (Harrison *et al.*, 2006), see Appendix 4.

6.2.3 Climate change impacts modelling approach

The structure of the landscape and how it is perceived by species, *i.e.* its traversability and quality, must be incorporated into a modelled system. Landscape was considered in this way in the ecological network approach (chapter four), whereby ecoprofiles were determined to be able to move through and reach additional habitat patches, subsequently forming networks of patches, *i.e.* ecological networks. This ability was determined by distance from the starting patch to neighbouring patches, intervening landscape structure, size of habitat and potential disturbance factors. As the consideration of climate change was carried out at a European scale an automated model was required to incorporate landscape structure and to handle the large datasets and ensure continuity. The GRIDWALK model (Schippers *et al.*, 1996), developed to simulate animal dispersal whilst taking account of landscape heterogeneity and linear barriers, was used. GRIDWALK allowed landscape use and connectivity to be linked to the climate envelopes of species modelled using SPECIES (Appendix 4). The exchange probability of individual habitat patches was determined by considering patch size, surrounding landscape and life history traits of individuals. Unlike the cost surface approach the dispersal distance of individuals is not set at a maximum but is a connectivity probability, regulated separately for each combination of patches with a value for leaving and arriving.

The third element in the modelling approach draws together climate and landscape factors and considers how species can disperse to and colonise habitat patches (Appendix 4). Individuals can only colonise new suitable habitat areas where climatic conditions become favourable if these areas are within reachable distance from a currently populated area (van Rooij *et al.*, 2007). Taking the movement model `SmallSteps` (Baveco, 2002), a vector based correlated random walk model, additional data was incorporated to allow a colonisation simulation model (`RANGESHIFT`) to be created. Three ecoprofiles were used in this model to represent the conditions required by a larger number of species found in wetland habitats (wetland), unimproved grassland (grassland) and broadleaf and mixed woodland (woodland). This allowed the approach to combine ecological network analysis and geographic scales most suitable for considering the impacts of climate change on biodiversity, *i.e.* the Atlantic European biogeographic region (EEA, 2008) and UK (Figure 6-1).

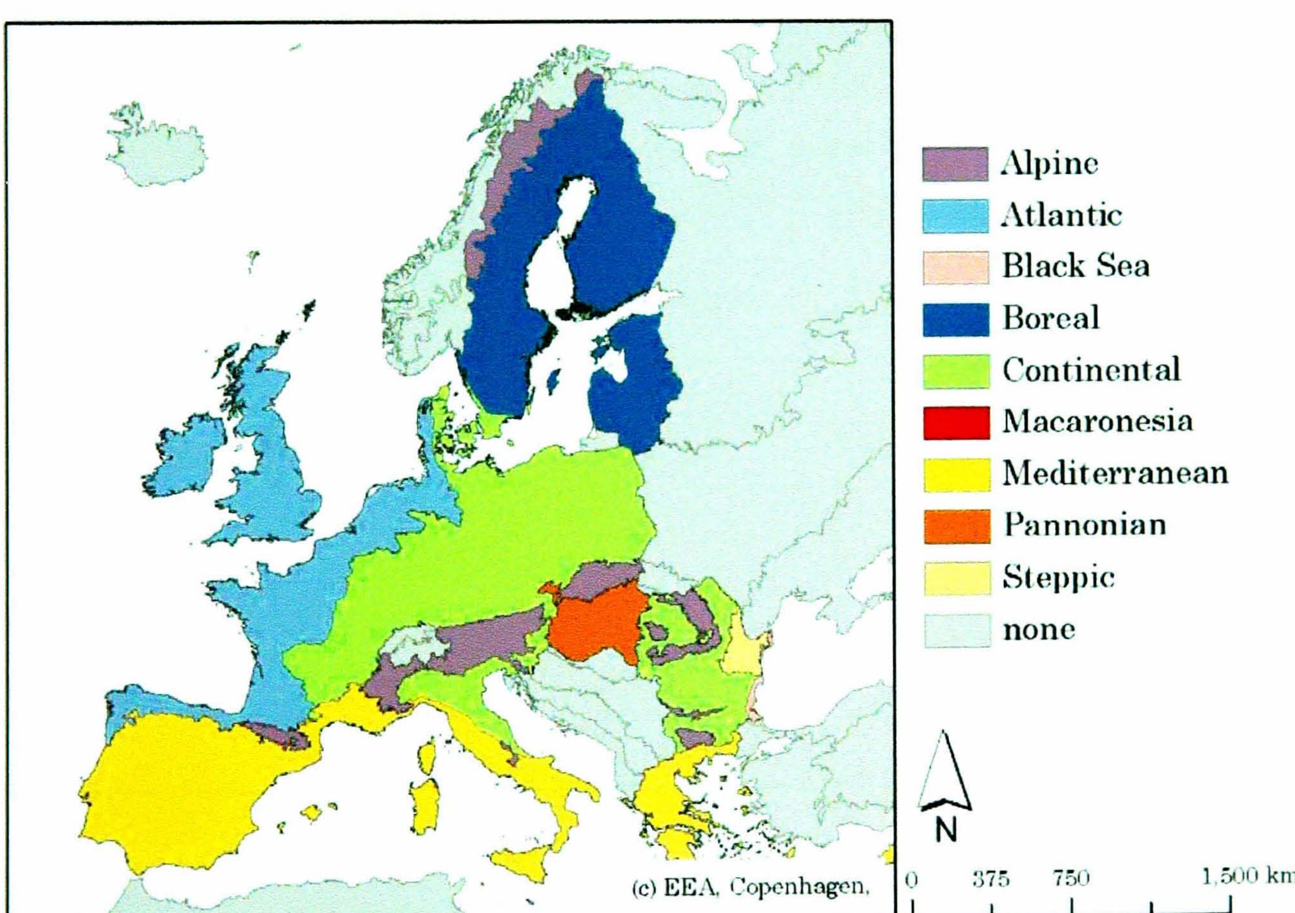


Figure 6-1 The Atlantic region study area in the context of all European biogeographic regions.

RANGESHIFT requires two types of spatial inputs, landscape elements and individuals. Landscape input was based on the European CORINE landcover data (EEA, 2005). Individuals were defined by the number of reproductive units available in the reproductive habitat in the current landscape (Table 6-1). Following this, numerous rules were specified for each input type to allow movement and population behaviour to be regulated, these were:

i. landscape quality,
ii. landscape element connectivity,
iii. boundary treatment,
iv. movement types,
v. mortality,
vi. reproduction,
vii. occupancy, and
viii. dispersal.

Table 6-1 Characteristics of ecoprofiles developed for RANGESHIFT.

Ecoprofile	Breeding		Dispersal		Mortality	Literature
	area per reproductive unit (ha)	mean clutch size	distance (km)	probability	probability	
Wetland habitat	5.0	4.4	15.0	0.5	0.56	(Antonov *et al.*, 2006; Robinson, 2005; Schulze-Hagen *et al.*, 1996)
Unimproved grassland	5.0	4.2	15.0	0.5	0.52	(Cramp, 1988; Evans *et al.*, 2005a; Evans *et al.*, 2005b; Hotker, 1988; Pedroli, 1978; Robinson, 2005)
Broadleaf & mixed woodland	20.0	4.0	10.0	0.5	0.40	(Holzkämper *et al.*, 2006; Kosinski and Ksit, 2006; Kosinski and Winiecki, 2005; Michalek and Winkler, 2001; Pasinelli, 2000; Pasinelli, 2001; Pasinelli, 2006; Pasinelli, 2007; Pettersson, 1985; Robles *et al.*, 2007a; Robles *et al.*, 2007b; van Adrichem *et al* 2007; Wiebe, 2006)

The quality of the landscape was determined by examining the type of habitat and how this related to the breeding habitat specified for the ecoprofile, and the size of habitat patch. This allowed the number of reproductive units available in each patch to be identified. The ability of landscape elements to allow or hinder dispersal, *i.e.* to act as a corridor, barrier or neutral was determined based on the results of GRIDWALK. When individuals are released in the model and complete a correlated random walk they pass from one element to another. Each element's connectivity value is assigned based on the probability of an individual moving into that element from the adjacent landscape element. This is linked to the landscape quality value already assigned. In addition, a boundary transition value is also specified. Boundaries are considered to be the locations where a decision is taken - will individuals cross the boundary between two landscape element types and how will movement continue? (Baveco, 2002). The value is determined for each combination of landscape element pairs showing the probability that an individual will move between two landscape elements. A probability of one is given for neighbouring elements of the same habitat quality, *i.e.* a transparent boundary.

Movement type is specified for individuals using two coefficients, step length and turning angle. The values for each are taken from reference data resulting from a one minute walk. Mortality is considered in two ways first, a mortality rate is specified for each ecoprofile (Table 6-1) which is applied after each time step, *i.e.* each year. Secondly, when dispersal does not occur but all territories within the inhabited patch are full any remaining individuals are terminated. Reproduction of individuals was determined by a coefficient identified through literature searches and then halved to take account of the female only model (Table 6-1), with a reproductive period occurring in each time step. The occupancy of the landscape by individuals was initially defined by the results of SPECIES. This specified which areas of the landscape were suitable for the ecoprofiles to inhabit. The population count occurs initially, based on the number of reproductive units, and at each time step following the reproductive and mortality events. This allows the growth in population to be assessed and can then be matched to the spatial dispersal progress of individuals in the landscape. At each major time step, *i.e.* years 2020 and 2050 SPECIES provides new climate envelopes for each ecoprofile. An assessment can then be made to determine whether individuals are able to reach this suitable climate from their initial starting point. Dispersal of individuals is considered in two ways, density dependent and density independent, with RANGESHIFT being run once for each dispersal type. Density dependent dispersal (ddd) ensures that juveniles colonise their natal habitat patch until all reproductive units are filled before any individual is allowed to disperse to a new patch. Density independent dispersal (did) allows juveniles to disperse both within the natal patch and to new patches. It is possible for individuals to disperse to patches with no free reproductive units, however, breeding only occurs in line with the reproductive unit number allocated to each habitat patch. The dispersal probability applied to individuals is based on that found in literature (Table 6-1).

6.2.4 Scales and sensitivity in RANGESHIFT

In order to investigate whether ecoprofiles were able to track changes in suitable climate space over the time period 2000 (current) to 2050 two landscapes were considered, the UK and an area of North West Europe corresponding to the Atlantic European Biogeographic region (EEA, 2008). Whilst these are larger in scale than the MKSM study area considered in the ecological network and natural greenspace assessments they include this smaller study area, provide the context for any changes which might occur there and fit well with the scale of climate data and predicted potential climatic changes. The same procedure was used for both landscape scales, with starting points for the initial population of individuals matched to currently suitable climate space. All patches within the suitable climate space envelope were used as starting patches, with the number of individuals in the starting population being equal to the number of reproductive units. In each of the time periods, current – 2020 and 2020 – 2050, four runs of RANGESHIFT were carried out, each with 100 repeats:

i. density dependent dispersal UK scale,
ii. density dependent dispersal NW Europe scale,
iii. density independent dispersal UK scale, and
iv. density independent dispersal NW Europe scale.

In order to examine the ability of RANGESHIFT to respond to different scenarios, sensitivity testing was carried out. Values were changed to consider different mortality and dispersal probabilities, both higher and lower than those which were identified for each ecoprofile. Mortality probability showed the highest sensitivity, with high values severely restricting colonisation, whereas changes in dispersal probability allowed colonisation to occur but this took place at a much slower rate. Following the sensitivity analysis, changes were made to the timing of the mortality event and the time point at which patch occupancy and population size data were collected. These changes allowed a more accurate number of individuals to be recorded for each year and allowed the rate of colonisation to be better understood.

6.2.5 Sustaining functionality in the landscape

The impacts of climate change on the functions of landscapes are already found. The particular effects found in the study area need to form the backdrop to any multifunctional strategy for the landscape. However, the scales and accuracies of data used within the three major topics investigated in this research, ecology, greenspace and climate change, make anything other than a simplistic combination of findings unwise. A 5 km^2 scale was determined to be appropriate for use in combining data as this allows a broad indication of appropriate habitat bank and accessible greenspace location whilst acknowledging the large scale of the original climate change data. The multifunctional assessment was carried out using a GIS. Data from the three topic areas were treated in a similar way with the aim of creating an additive raster grid which would show areas where all conditions could be met to retain or achieve a multifunctional landscape.

i. Areas identified as having suitable climate space for each of three ecoprofile types, woodland, wetland and natural grassland were extracted at the time stamp 2020 and 2050. These data were converted into raster files with a cell size of 5 km^2.
ii. Habitat banks sites suitable for each ecoprofile type: wetland habitats, lowland heath, unimproved grassland and broadleaf and mixed woodland identified through the ecological networks assessment process were extracted. The data was then converted into raster files with a cell size of 5 km^2.
iii. The results of the network analysis carried out at the JCAs with development planned study scale were extracted. Areas where the population was able to access natural greenspace according to application of ANGSt were isolated. The results of the final model run were used which identified populations with accessibility post MKSM development, population increase and introduction of additional accessible greenspace areas (using the biodiversity led approach). These areas were converted to raster files with cell size 5 km^2.

Data showing results for the different habitat types were separated, as were the two time stamps. This allowed multifunctionality to be assessed at two time periods and compared to that found currently. Using a raster calculator, individual layers were compiled into a multi raster layer which allowed areas to be identify where all criteria were met, *i.e.* areas were identified as habitat banks linked to existing ecological networks, areas were accessible to the local population in line with ANGSt, and areas were within the zone of suitable climate space. This process was repeated for the wetlands, natural grassland and woodland habitat types and the two time stamps. The lowland heath habitat was not included in this final multifunctional assessment as it had not been part of the suitable climate space assessment. This meant it was not possible to make the comparison of current landscape functionality currently with that found at 2020 and 2050.

6.3 Results

The climate change scenarios utilised in this research represent predictions developed using the best available data. It is clear that such changes are likely to have significant impacts on the landscape and the habitats and species it supports. Examining the changes to the climate envelope over the fifty year study period for each ecoprofile shows there to be a large spatial difference in the areas currently suitable and those that will be so by 2050 (Figure 6-2). This is prior to consideration of whether the species informing the ecoprofile would be able to keep pace with such changes. Although there appears to be little difference in the climate space between the current and 2020 time periods, by 2050 this is no longer the case. There is a clear and significant shift in suitable climate space for all ecoprofiles. The woodland ecoprofiles climate space contracts in mainland Europe whilst extending further in the UK, whereas the climate space for the wetland ecoprofile contracts eastwards without any extension areas. The grassland ecoprofile shows a large contraction with only minimal extension areas in the east of Scotland. Such dramatic changes in climatic suitability are also likely to lead to changes in the physical landscape as species are no longer able to survive in their existing locations. Ecosystems are comprised of assemblages of species which require similar conditions, as these conditions change ecosystems and their constituent habitats may shift geographically or change in species composition (Opdam and Wascher, 2004).

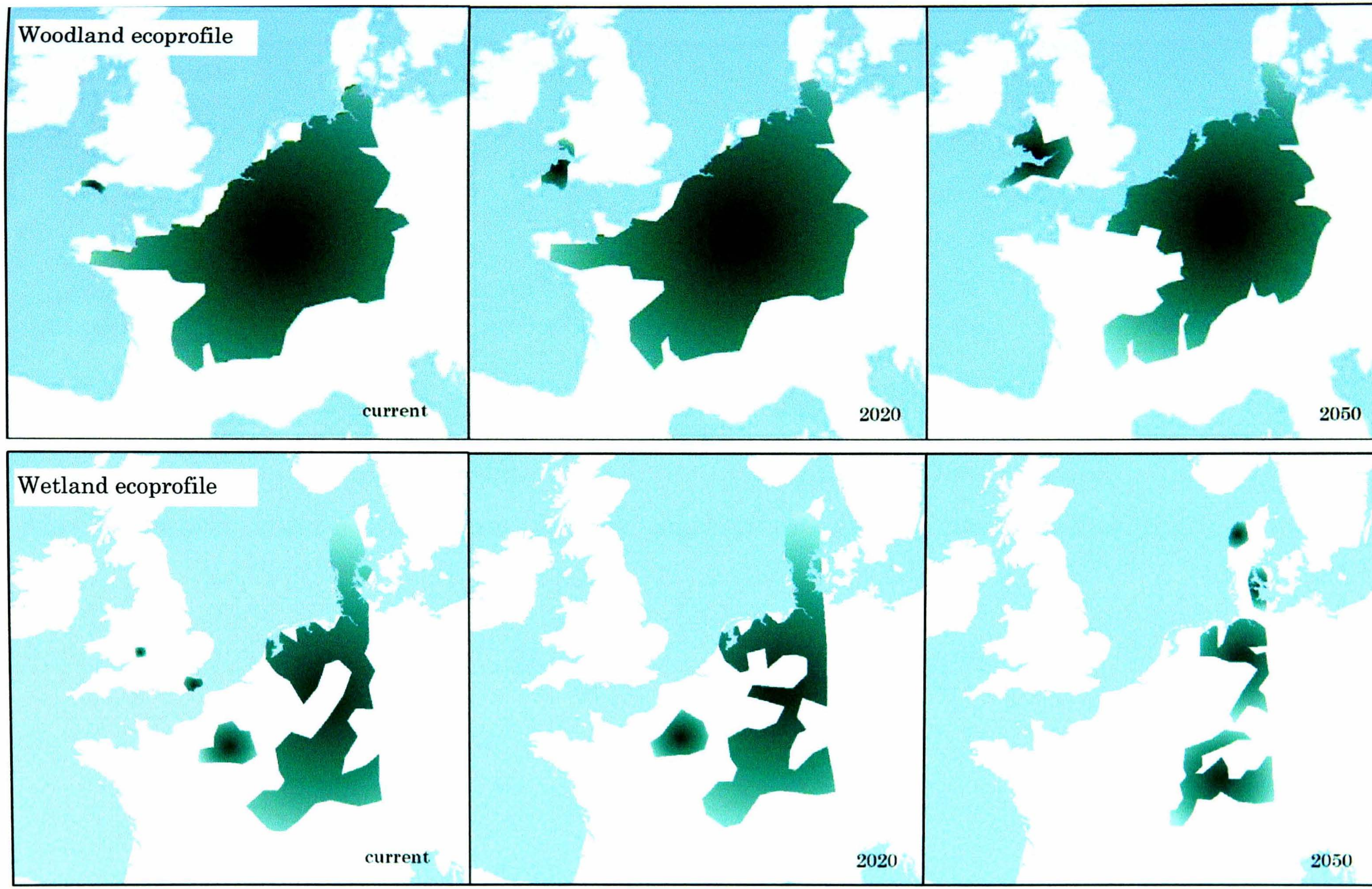


Figure 6-2 Areas of the Atlantic European biogeographic region with suitable climate space for ecoprofiles at three time periods

Figure 6-2 Areas of the Atlantic European biogeographic region with suitable climate space for ecoprofiles at three time periods

6.3.1 The ability of ecoprofiles to keep pace with climate change

Relying only on the extent of suitable climate space to predict whether species adapt to changes in climate assumes that species distributions always mirror climatic limitations, however, species may respond to shifting climate conditions by a shift in their realised niche (Opdam and Wascher, 2004). Using RANGESHIFT the dispersal and colonisation abilities of each ecoprofile can be examined. Habitat patches large enough to contain a reproducing population, *i.e.* reproductive units (RU), were identified and the first breeding event taking place in each was recorded. If individuals were able to disperse from existing habitat patches and reach and colonise patches identified as having suitable climate in the next time period they were determined to be able to keep pace with the changes in climate to some extent. A number of measures were developed to consider whether ecoprofiles were able to keep pace with the changing climate.

6.3.2 Colonisation of habitat patches

The percentage of habitat patches colonised by each ecoprofile in each time period provides information on both the connectivity of the landscape and the ability of the ecoprofile to colonise (Table 6-2).

Table 6-2 The percentage of habitat patches (RU) colonised by ecoprofiles under differing dispersal and climatic conditions

Ecoprofile	Model run	% RU colonised Current - 2020	% RU colonised 2020 – 2050
Woodland	ddd UK	58.8	98.0
	did UK	55.0	97.3
	ddd NW Europe	97.9	98.8
	did NW Europe	99.0	98.8
Wetland	ddd UK	0.0	0.0
	did UK	0.0	0.0
	ddd NW Europe	53.6	0.0
	did NW Europe	53.6	0.0
Grassland	ddd UK	59.6	51.3
	did UK	59.6	51.3
	ddd NW Europe	52.4	50.6
	did NW Europe	52.4	50.6

Each ecoprofile behaves differently to the shifts in suitable climate space reflecting both the structure and connectivity of the landscape and the likely ability of particular species to adapt to changes. An increased ability to disperse and colonise is seen for the woodland ecoprofile with almost all suitable habitat patches being colonised by 2050. The two types of dispersal mechanism produce similar results at both scales of the study. However, the overall trend in the results differs between study scales. In the UK the results from current to 2020 reveal large areas of the landscape which the ecoprofile cannot colonise. This issue appears to be overcome in the period 2020 – 2050 where most of the landscape becomes colonised. Within this second time period the number of suitable habitat patches increases by around a third, which is likely to have increased the connectivity between patches allowing more widespread colonisation of the landscape to take place. At the North West European scale the ecoprofile colonises the majority of habitat patches in both time scales. Whilst this may suggest an ability to keep pace with changes in suitable climate space, there is an underlying issue. The change in suitable climate space at this scale means that the number of habitat patches available for colonisation decreases by 21% between 2020 and 2050. This results in the woodland ecoprofile exhausting the supply of suitable habitat patches owing to no further possible expansion and could lead to a reduction in the population size the landscape is able to support.

The results for the wetland ecoprofile are more marked. Although the ecoprofile has suitable climate space in the UK at the moment it is not able to colonise any additional patches in the future predicted climate space. Whilst species represented by this ecoprofile may be able to survive in their current habitats any retreat of suitable climate space north may result in population mortality. A similar scenario is found at the North West European scale. Despite initially being able to colonise habitat patches in newly suitable climate space between 2020 and 2050 this ability recedes and the ecoprofile is no longer able to shift its range to keep pace with the change in climate.

The grassland ecoprofile's ability to colonise is very similar, using both dispersal mechanisms and at both scales with approximately half of the suitable habitat patches being colonised in each case. However, the changes in climate space result in a significant reduction in the amount of habitat available for colonisation in the UK; a 61% reduction is seen between current and 2050 and at the North West European scale the habitat patches available are reduced by 79%. This reduction in the carrying capacity of the landscape would lead to a reduction in population size of species found in such grassland habitats.

6.3.3 Achievable colonisation distances

Colonisation distance, *i.e.* mean total distance of colonisation events, can be used to consider the potential an ecoprofile has to be able to keep pace with a changing climate and to identify if lags are beginning to develop (Figure 6-3a and 6-3b). This analysis was also used to identify differences between the two dispersal methods. When RANGESHIFT was run using density dependent dispersal (ddd) larger colonisation distances were achieved overall. However, considering the time periods separately, the results are less well defined. Comparing the distance ecoprofiles are able to travel to colonise with the movement of suitable climate space allows the ability of the ecoprofile to keep pace with climate change to be understood. Suitable climate space movement is varied for each ecoprofile with extensions occurring in some directions and retractions elsewhere, usually to different extents. The suitable climate space for the woodland ecoprofile extends north by 207.44 km over the fifty year total study period whilst this ecoprofile is only able to colonise up to 144.69 km (NW Europe). However, the retraction of suitable climate space at the southern edge of the range is only 0.34 km, resulting in the woodland ecoprofile being able to stay within the suitable climate space throughout. Changes in suitable climate space occur in all directions and whilst the woodland ecoprofile is relatively unaffected longitudinally, latitudinally there is a contraction on the European continent of 356.45 km from the west with no corresponding extension to the east resulting in a smaller overall suitable range. The wetland ecoprofile is able to colonise in the current – 2020 time period but fails to colonise beyond 2020. In the study period 2020 – 2050 suitable climate space contracts massively from the west by 563.2 km and from the south by 44.8 km. Alongside overall reduction in size of climatically suitable habitat the connectivity of patches is reduced, further hampering colonisation efforts. The consequence of this limitation of climatically suitable space is a paucity of newly available habitat patches and the initial population being detained within their natal patch. The grassland ecoprofile in the UK appears to be able to colonise habitat patches well in both time periods. However, the total distance achieved masks two issues, a shift and reduction in suitable climate space and a reduction in the number of available habitat patches. Changes in suitable climate space is large with reductions of over 40% in the UK and over 60% in NW Europe. Whilst a colonisation distance of approximately 45 km in the time period 2020 - 2050 is achieved, even the short term trajectory of this ecoprofile does not appear positive.

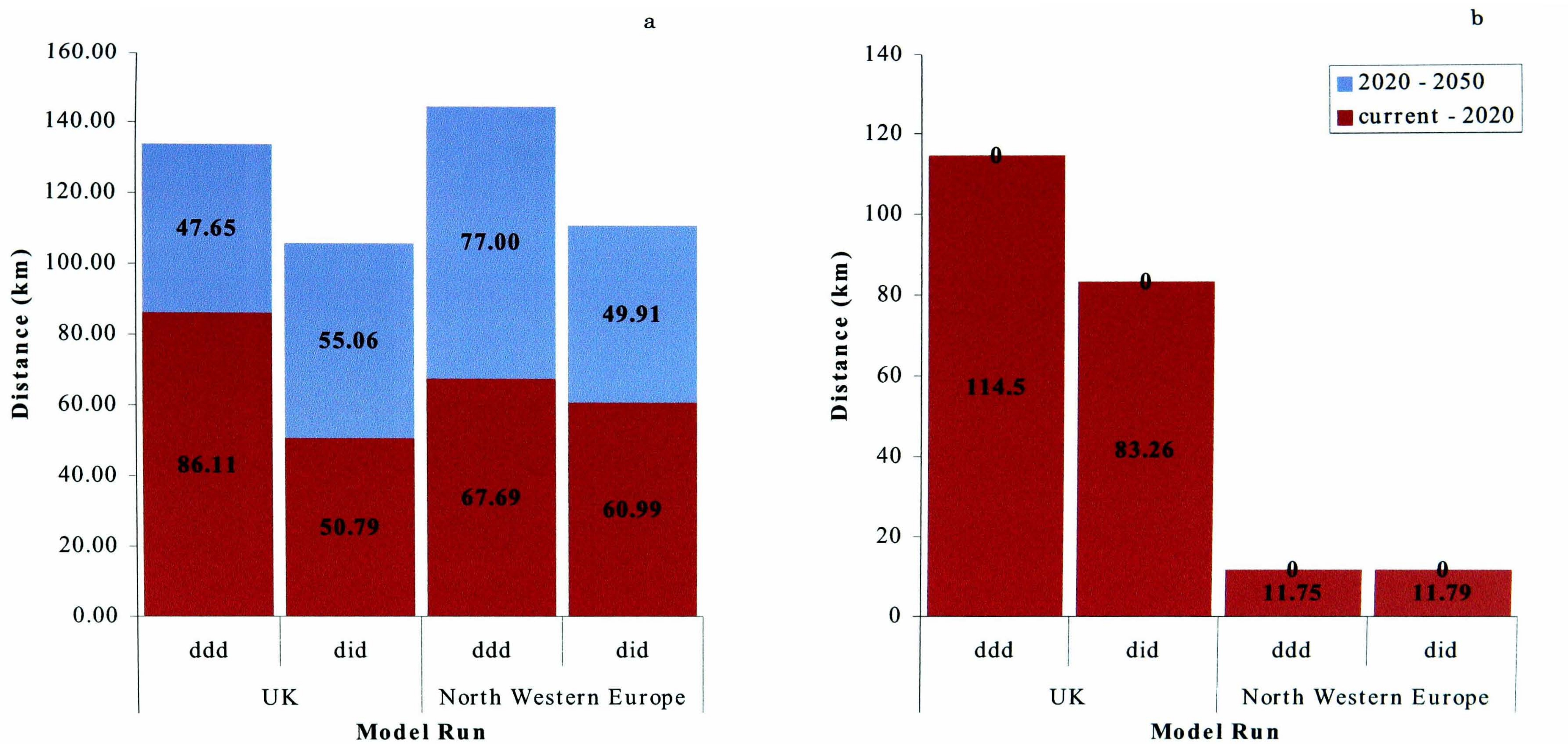


Figure 6-3a The mean total distance of colonisation events occurring in each time period and overall: a. woodland ecoprofile, b. wetland ecoprofile

c

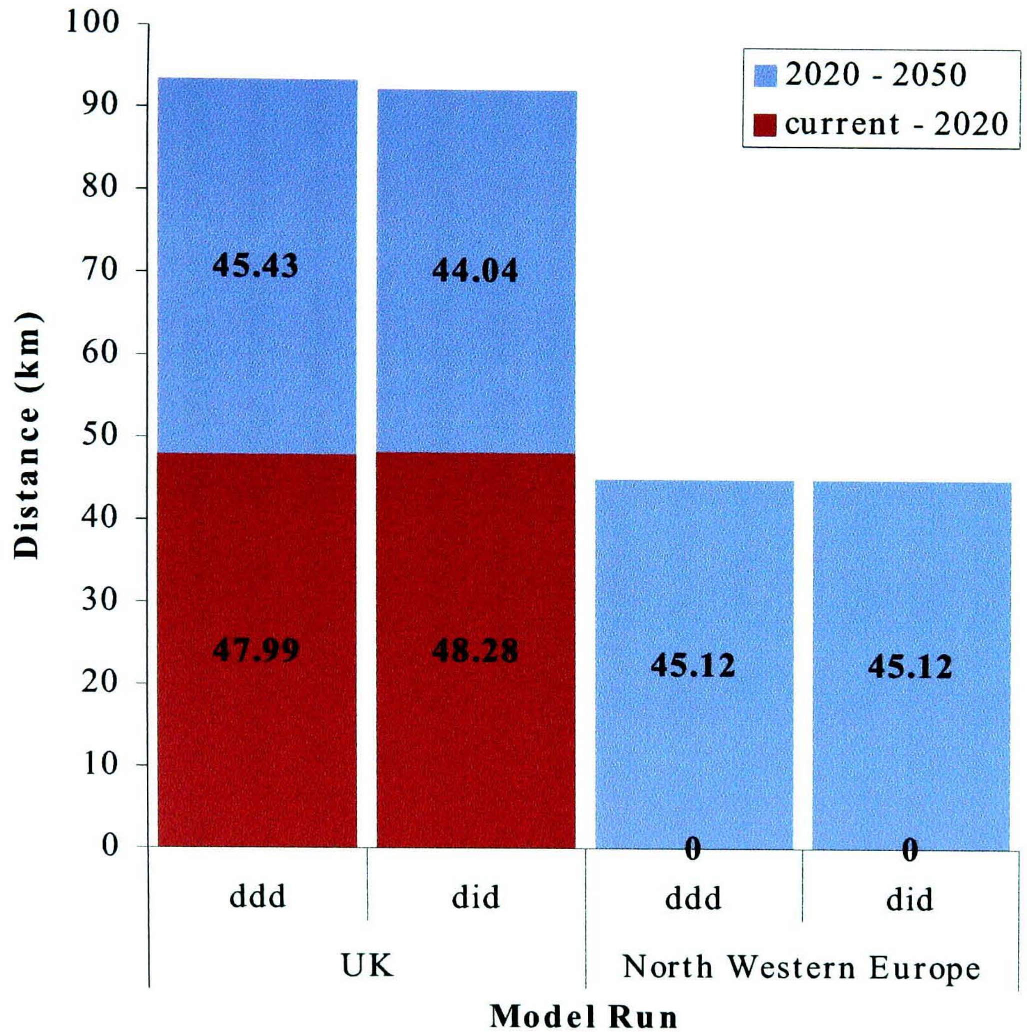


Figure 6-3b The mean total distance of colonisation events occurring in each time period and overall: c. grassland ecoprofile

6.3.4 Frequency of ecoprofile colonisation

Results showing the entire landscape allow the overall ability of ecoprofiles to track changes in suitable climate space to be examined. However, examining how individuals behave in a more localised way provides a better insight into how habitat banks could be used as a landscape adaptation mechanism. Considering the frequency of colonisation events between habitat patches allows clusters of activity to be identified. It is also possible to determine areas which, although climatically suitable, cannot be reached by individuals (Figure 6-4a and 6-4b). This analysis was carried out in the UK only in order to relate the climate change results to the study area used in the ecological and greenspace assessments. The woodland ecoprofile has strong clustering of colonisation movements in both time periods. There are colonisation movements, however, which link these clusters, particularly in the current – 2020 time period. This suggests that over time individuals are able to make movements into the wider landscape by dispersing and colonising new patches, reproducing, and their offspring going on to colonise additional patches. However, such activity depends on there being suitable habitat patches in the landscape between clusters of activity, and the requirement for individuals to frequently disperse from their natal patch. The colonisation frequency of the grassland ecoprofile in the current – 2020 time period is clustered in the east of the UK and East Midlands. The number of individual colonisation movements is high with 68% of the UK colonisation movements occurring in this cluster (current – 2020 time period only). The colonisation ability of the ecoprofile changes, however, in the 2020 – 2050 time period due to a significant reduction of over 40% in the suitable climate space. The eastern UK and East Midlands are particularly affected resulting in the ecoprofile being unable to make colonisation movements at the same pace as changes in suitable climate space occur. The ecoprofile remains in the north east of England and east of Scotland but is severely restricted. The results for this ecoprofile are particularly dramatic given its widespread ability to disperse and colonise in the earlier time period. The wetland ecoprofile, whilst having habitat in suitable climate space initially, was only able to reproduce within habitat patches currently occupied. This resulted in movement only occurring between habitat patches designated as 'starting patches' and colonisation of new habitat was determined not to have occurred in either time period.

Figure 6-4a The frequency of colonisation movements for the woodland ecoprofile in the UK

Figure 6-4b The frequency of colonisation movements for the grassland ecoprofile in the UK

6.3.5 Habitat patch connectivity

The frequency of colonisation movements by ecoprofiles is strongly regulated by the connectivity of habitat patches. However, frequency of colonisation events does not provide enough detail to determine connectivity. Where there are many habitat patches where colonisation could occur the number of times an individual patch is colonised reduces, whereas if only a small number of patches are accessible the same movement event occurs frequently. Taking the colonised patches and considering the number of source patches colonising individuals originate from can, however, be used to determine how well connected an individual habitat patch is within a habitat network. Using this connectivity value specific areas of the landscape can be identified which form critical nodes within a network. Conversely, areas with habitat patches which are sparsely connected can be identified and, over the time periods considered, levels of existing and predicted connectivity can be determined (Figure 6-5a and 6-5b). It has been identified that changes in habitat suitability and the ability of species to survive and adapt to climate change are likely to be exacerbated by low landscape connectivity or high fragmentation (Opdam and Wascher, 2004; Travis, 2003; Vos *et al.*, 2008). Therefore, if areas which are shown to have low connectivity can be identified, landscape adaptation activity can be focused there. By focusing on the MKSM study area used in the ecological network and greenspace assessments it is possible to consider how predicted climate change may further impact on issues previously identified in the landscape. Connectivity is assessed qualitatively owing to the fact that only simple measures, *e.g.* number of source patches colonising individuals originate from, have been used. The woodland ecoprofile is only able to colonise patches on the edges of the wider study area in the current – 2020 period. However, the number and connectivity level of colonised patches increases during the 2020 – 2050 period. All but two patches are considered to have medium or high connectivity in this latter time period suggesting that the landscape is well suited to the requirements of the ecoprofile. As the suitable climate space progresses a larger proportion of the MKSM study area becomes climatically suitable for the woodland ecoprofile. This suggests that mechanisms such as habitat banks which can contribute to a larger and better connected woodland resource should be informed by requirements of newly colonising species and where they may require habitat. The current – 2020 time can only be considered for the grassland ecoprofile, beyond this time suitable climate space in no longer found in the MKSM study area. Within this first time

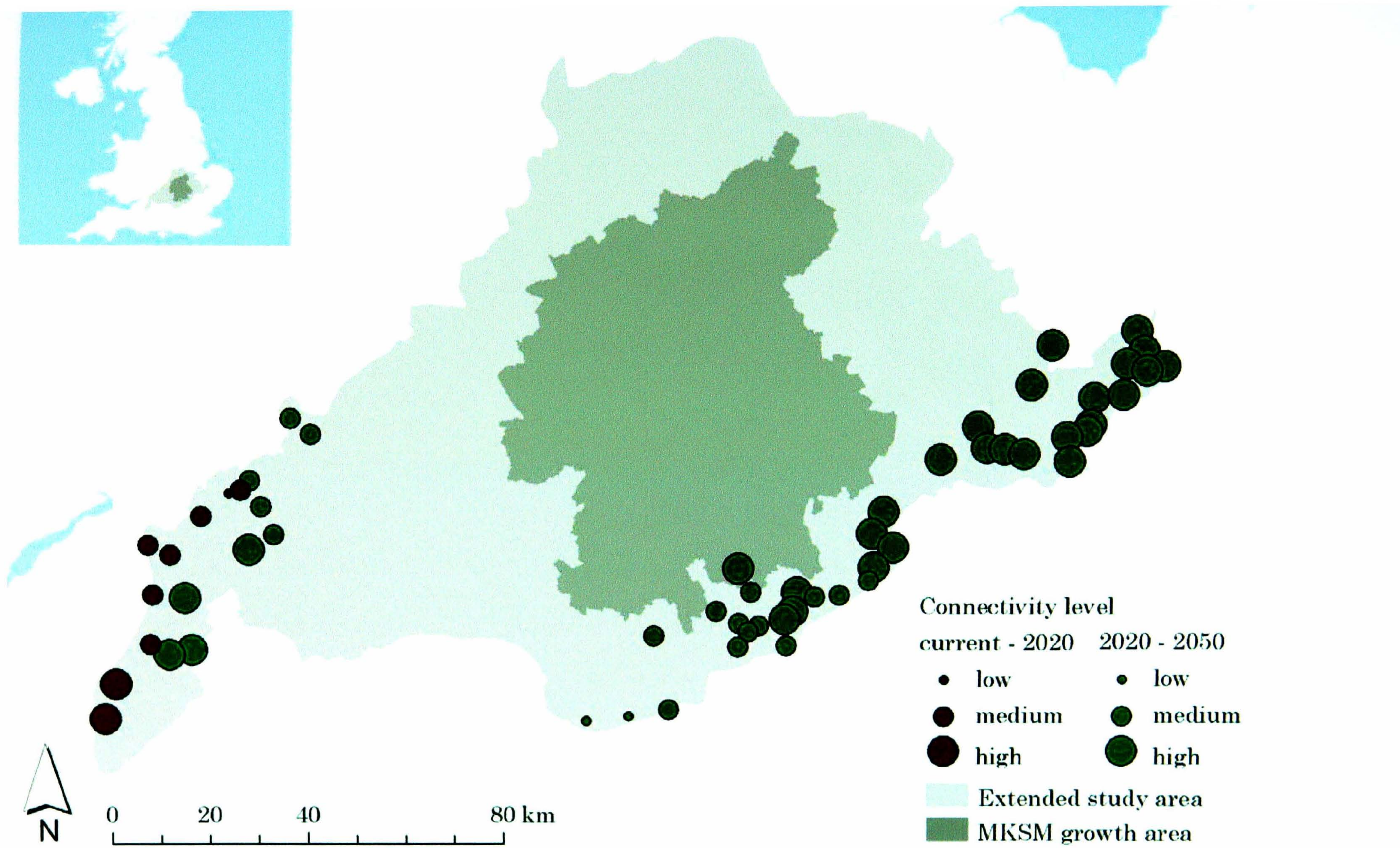


Figure 6-5a The connectivity level of woodland ecoprofile habitat patches in the MKSM study area over the two time periods

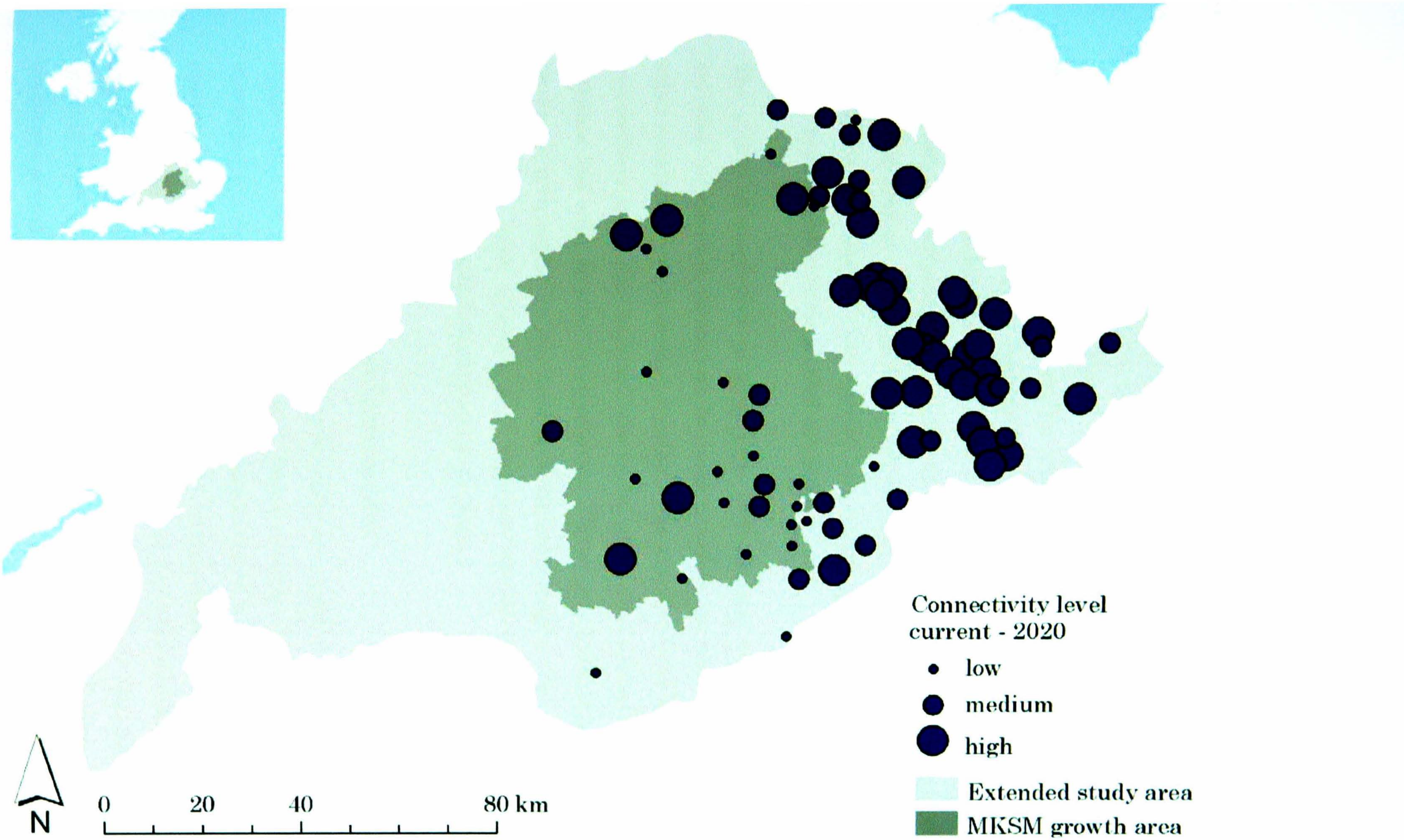


Figure 6-5b The connectivity level of grassland ecoprofile habitat patches in the MKSM study area over the current - 2020 time period

period there are a large number of colonised habitat patches within the study area. The connectivity levels found are spatially explicit with the majority of western patches having a low connectivity level whereas patches in the east are dominated by medium and high connectivity levels. Despite the appearance of a well connected network of habitat patches in this time period, by the 2020 – 2050 period the MKSM study area is no longer found in the suitable climate space of the grassland ecoprofile. This suggests that even if available habitat were increased in size and connectivity the ecoprofile would still be unable to maintain a viable population. This outcome highlights the necessity for long term predictions of climate space and landscape suitability. It identifies the importance of landscape adaptation mechanisms which manage landscapes to ensure long-term viability of vulnerable species. The inability of the wetland ecoprofile to colonise new habitat within the UK suggests such species may have significant problems in retaining viable populations in their current ecosystems.

The ability of ecoprofiles to keep pace with changes in suitable climate space provides information which can be used to determine how climate change may affect ecosystem and landscape function. The results of the woodland, grassland and wetland ecoprofiles highlight the varied response of species requiring differing habitats and with diverse characteristics. Whilst only qualitative conclusions can be drawn with regard to the impacts of climate change on the MKSM study area these provide both positive and negative spatial indicators which can be incorporated into habitat bank location strategies.

6.3.6 Combined landscape functionality

Temporal changes in landscape functionality are clearly demonstrated through the combination of the landscape assessment models. This makes it possible to determine the critical time by which action needs to be taken to ensure that a positive landscape functionality trajectory is maintained or achieved. Owing to the ecological, social and economic background to the models developed the results provide an intriguing insight into the sustainability of the landscape in the medium term. Although the results are coarsely grained they allow the impacts of landscape pressures outlined in this research to be identified Figure 6-6a-d.

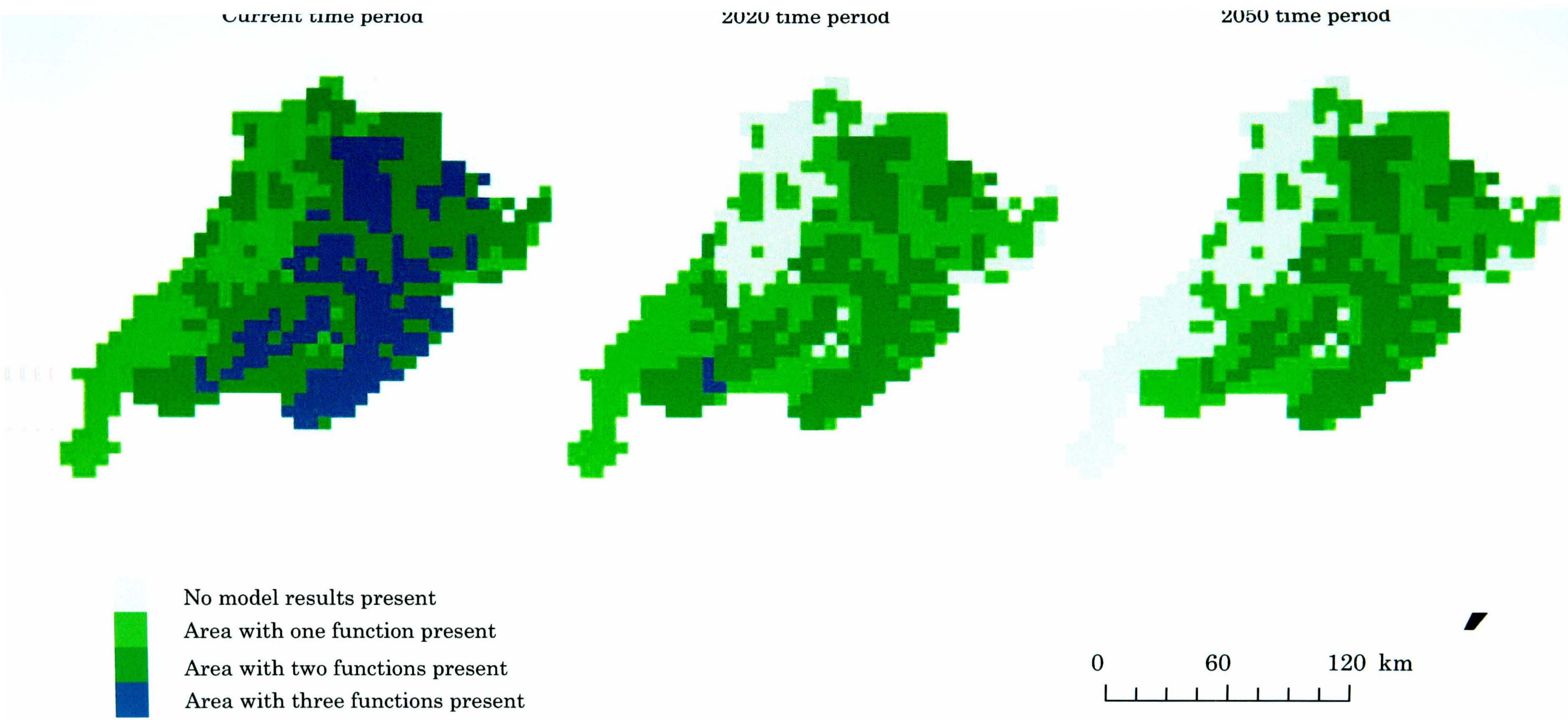


Figure 6-6a Combined model results for woodland showing multifunctional areas in three time periods: current, 2020 and 2050

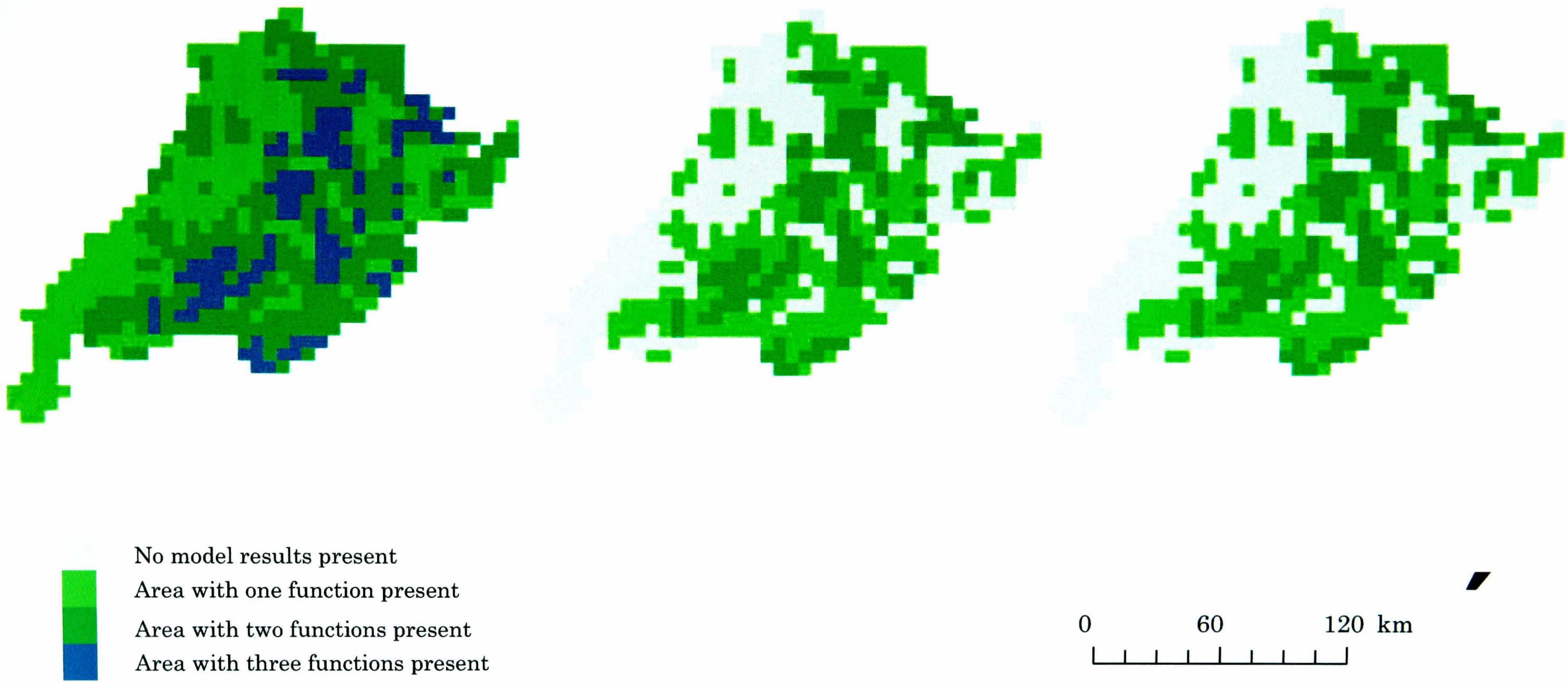


Figure 6-6b Combined model results for wetland showing multifunctional areas in three time periods: current, 2020 and 2050

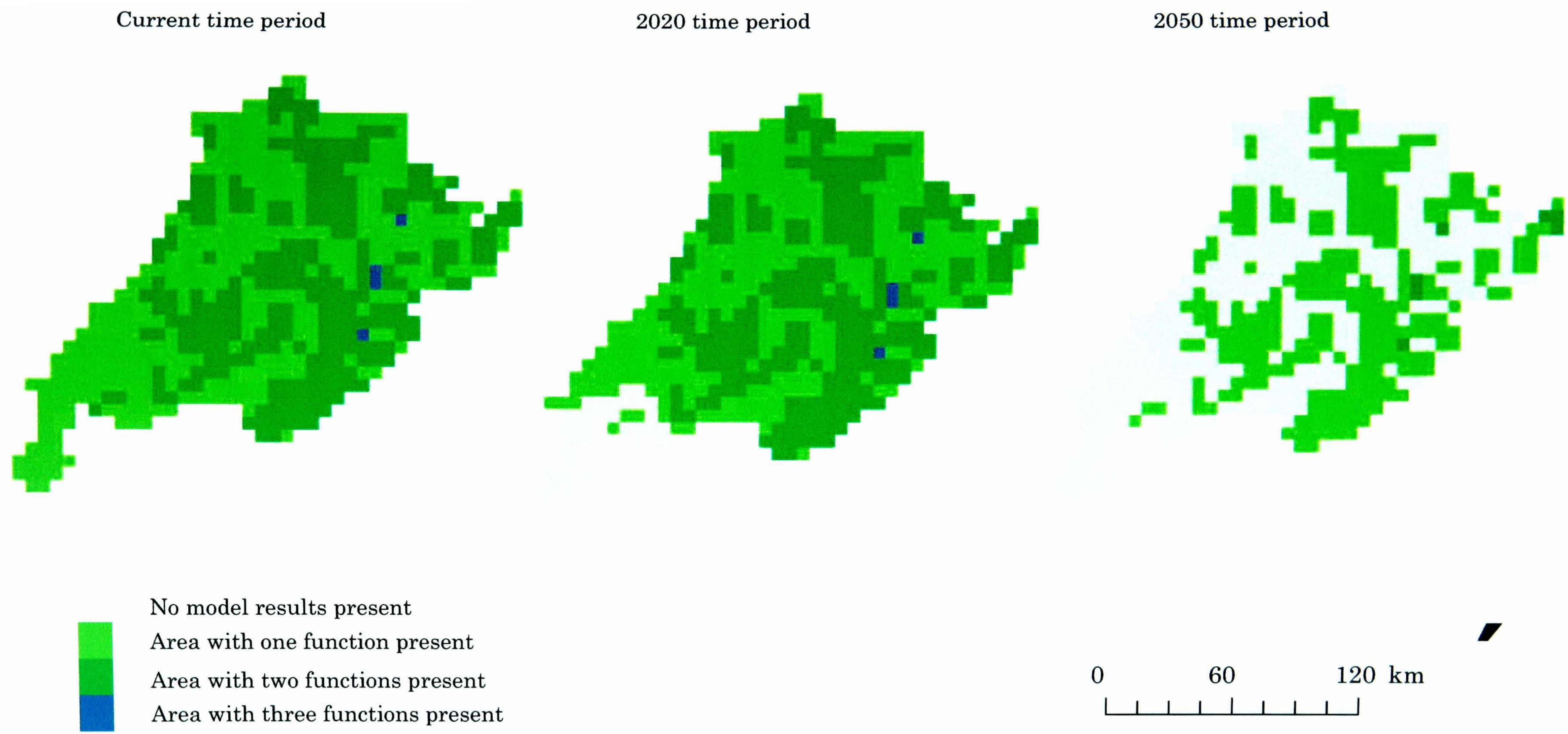


Figure 6-6c Combined model results for grassland showing multifunctional areas in three time periods: current, 2020 and 2050

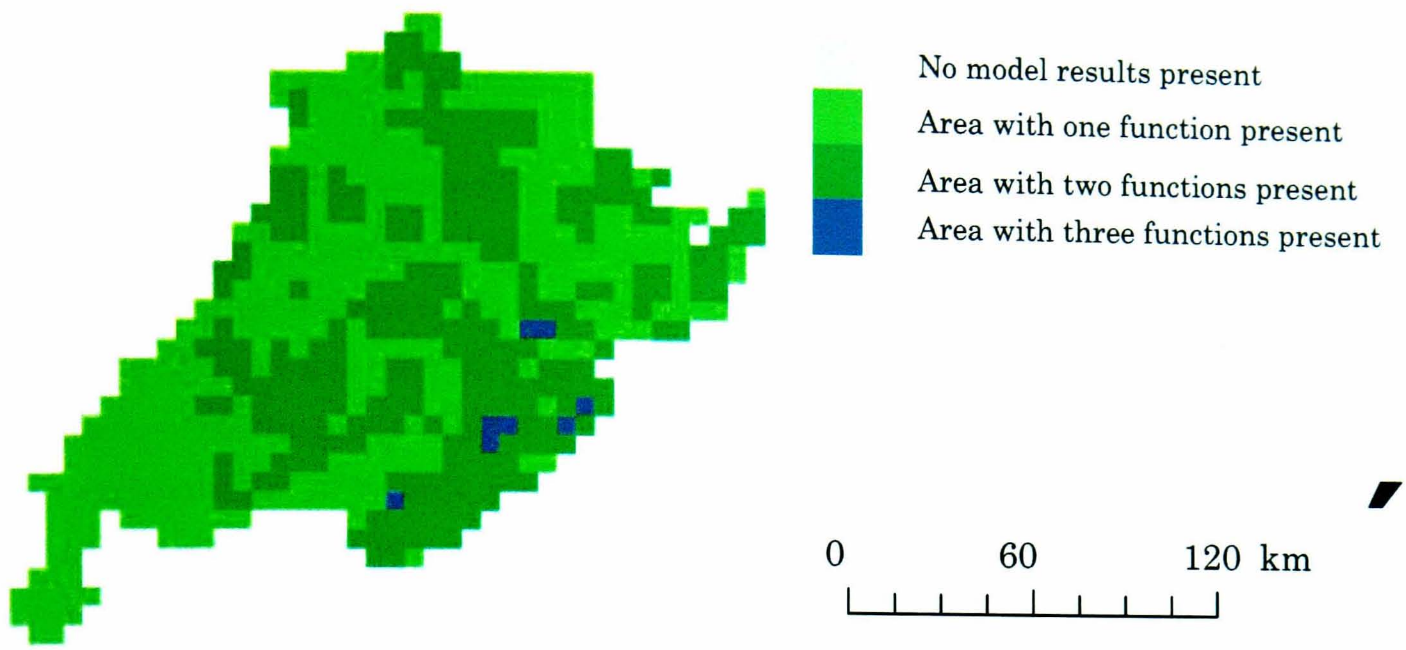


Figure 6-6d Combined model results for lowland heath showing multifunctional areas in the current time period

Areas of multifunctionality are determined to be those locations where all three models created combine. Whilst a weighting of models has not been carried out the results of the climate change modelling are viewed as crucial and limiting factors in identifying multifunctionality. Therefore, where climate space is not suitable, even if the ecological and greenspace models coincide, the areas are considered to be of limited functionality. This ensures that the importance of landscape trajectory and future functionality becomes a central part of any evaluation of the landscape, and the importance of projecting forward and planning with the future requirements of the landscape in mind, an issue which has been part of the model development process throughout this research.

The results for the woodland models (Figure 6-6a), reveal large and well connected areas of multifunctionality in the landscape in all but the west of the study area in the current period. However, by 2020 this situation has deteriorated with no multifunctional areas identified and a significant area where such limited functionality is seen that no models produce positive results, again in the west. Interestingly, between 2020 and 2050 the suitable climate space found in the study area allows a small part of the landscape to regain its multifunctionality. However, the most of the landscape remains with limited functionality and, given there is a period between current and at least 2020 when functionality is lost from the landscape, it should be considered whether it would be possible in a real landscape to regain functionality with this depicted relatively ease.

The level of multifunctionality seen in the results for wetlands is relatively high (Figure 6-6b), although individual areas are not particularly well connected. The western area of the study area is again unrepresented within the multifunctional areas. This is more readily seen in the subsequent time period as no model has positive results in this area. In fact all multifunctionality is lost from the wetland landscape by 2020 and this continues to be the case in 2050. Whilst not unexpected, given the suitable climate space identified in Figure 6-2, this is particularly concerning given that wetlands were a habitat type with multiple options for habitat banks and were seen to be relatively robust ecosystems which would be able to withstand visitor pressure and provide access to natural greenspace areas.

The combined assessment of grassland areas within the landscape is of particular concern. In the current period an extremely limited area is identified as multifunctional (Figure 6-6c) and, whilst this area remains so in 2020, there is a clear functional retreat beginning from the south of the study area. By 2050 the reduction in functionality has become apparent across the entire study area revealing a landscape with significant ecological issues.

It is only possible to consider the current situation for the lowland heath results owing to a lack of corresponding climate change data. A small area of multifunctionality can be identified suggesting, at least currently, that the locating of habitat banks can provide a valued addition in the landscape.

It is clear that a multifunctional landscape is influenced by more than ecological, greenspace and climate change factors. There are more explicit economic and social factors which have not been considered here. However, the initial driver of this research, large scale and prolonged housing development leading to increased built land and an increased population, mean that many of these issues have been considered implicitly throughout the research. This research has proposed one approach to considering how a landscape strategy can be informed by functionality in the landscape and as such brings together individual distinct and separate fields into a common approach where results can be compared and combined.

CHAPTER SEVEN | Discussion

7.0 Introduction

By examining the various functions performed by the landscape, and mapping pressures both current and future, a real understanding of the trajectory of, and inputs required in, the landscape can be achieved. Explicitly examining the connections and synergistic value of functions in developing a spatial strategy in the landscape addresses the three aims of this research:

1. To examine habitat banking as a practical response to habitat fragmentation and the degradation of functional landscapes.
2. To determine the nature of green infrastructure expansions required in order to compensate for landscape changes and to address future requirements.
3. To develop a methodology which determines optimal solutions for habitat bank and wider green infrastructure location and development.

7.1 Ecological networks in the landscape

A methodology has been developed in this research to determine habitat fragmentation in the landscape and to link this to a spatial response strategy based on the inherent ability of landscapes to operate in a multifunctional way. Landscape ecology research is often focused exclusively on landscape elements and structure (Forman, 1995; Hobbs, 1997), whereas it is ecological processes and functions which produce the composition and configuration of habitat patches (Bélisle, 2005) and thus provide a more complete understanding of the landscape. The use of ecological networks as a way of considering flows and functions in the landscape has become popular in Europe and the UK over the past twenty years or so. A number of national and regional ecological networks have been proposed, *e.g.* (Cheshire County Council, 2004; van der Sluis *et al.*, 2003) as a way of reversing habitat and species decline. Agents of these networks have included umbrella species, generic focal species and single species (Verboom and Pouwels, 2004; Watts *et al.*, 2005a). However, a transparent and easily transposable approach which still takes account of locally relevant species, habitats and biodiversity policy and targets has often eluded landscape ecology researchers. The methodology developed in this research, which uses multiple sub-regionally defined ecoprofiles to represent multiple habitat types, allows detailed life history traits to be incorporated into the approach whilst not relying on a single species to represent an entire habitat type. The use of multiple habitats allows the way habitats are used by species and interact in the landscape to be understood and strategies to be developed which take account of this.

7.1.1 Functionality within the current landscape

Fragmentation of habitats found in the study landscape reveals evidence of small habitat patch size, dissection of habitat and low habitat quality. Over 52% of SSSIs in the study area are not in a favourable condition (Natural England, 2009) suggesting that a large proportion of habitats are likely to have quality issues. The proposal to incorporate habitat banks in the landscape as a method of quantitative and qualitative improvement alongside their role as sites for amelioration of negative development impacts therefore appears valid. The spatial location of banks is of primary importance in order to address qualitative fragmentation issues rather than to just provide an increase in habitat area. The use of nine ecoprofiles representing four habitat types allowed a considerable proportion of the landscape to be examined in a comparable way. The quantity and size of existing ecological networks (see section 4.4.2) suggest that habitat banks could have different roles for each habitat type, allowing particular weaknesses to be addressed.

The low number and small size of lowland heath (6 networks totalling 1169 ha) and unimproved grassland (69 networks totalling 5079 ha) ecological networks suggest an increase in overall size is required initially in order that they remain able to support existing populations. The quantity of these habitats found in the study area is a major limiting factor in defining ecological networks. Where ecological networks can be identified they suffer from lack of connectivity as individual networks are spatially disparate. Increasing the quantity of suitable habitat in a spatially targeted way would assist in creating a larger habitat resource whilst linking together networks to form larger structures. Larger networks based on functional cohesion of habitat patches allow individuals to utilise more sites, increasing the potential for species to form metapopulations (Opdam, 2002). The identified wetland habitat and broadleaf and mixed woodland ecological networks are numerous, and, particularly in the case of broadleaf and mixed woodland, are generally large in area (mean network size 1201ha). Habitat banks in these cases are likely to be of greatest value in trying to achieve qualitative improvements, as it is implicit within ecological and landscape planning that good habitat quality is required to ensure functional habitat patches, networks and landscapes (Thomas *et al.*, 2001; Vos *et al.*, 2007).

7.1.2 Habitat bank locations and implications

The outcome of this approach is to propose a series of habitat bank locations for each habitat type alongside a quantitative understanding of how existing ecological network functionality would be affected. By targeting habitat bank locations only within JCAs directly affected by development it is proposed that banks created will be of greatest ecological and social relevance. JCAs have been used as ecological units in numerous landscape planning strategies (Bailey *et al.*, 2006; Handley *et al.*, 1998; Lee and Thompson, 2005; Swanwick, 2004), because they define areas with the same landscape, wildlife, natural and cultural characteristics (The Countryside Agency and Scottish Natural Heritage, 2002). This research identifies that, by offsetting adverse ecological impacts resulting from development within the same JCA, the resulting habitat bank is more likely to be able to support similar habitats and species assemblages to those affected. The results reveal many options for locating broadleaf and mixed woodland and wetland habitat banks. This will allow multiple bank sites to be operational simultaneously and for large amounts of in-kind, off-site mitigation works to be achieved. However, the results for lowland heath and unimproved grassland suggest a similar level of opportunity would not be available. Habitat banks need to be located in the few areas deemed to be abiotically and biotically compatible for such habitat creation and development according to landscape constraints (Table 4-5). The overall area of such sites is low, only 31.09 ha for lowland heath and 775.79 ha for unimproved grassland compared to that available for the other habitat types. However, it should be recognised that lowland heath accounts for only 0.04% of the study area and, owing to the biodiversity value of such sites, it should be possible for developments to be planned to avoid impacts on this habitat. The nature of the lowland English landscape, however, means that this is not the case for unimproved grassland which accounts for over 8% of the study area and is likely to be negatively impacted by development of the scale expected in the MKSM. One option highlighted by this research, and in keeping with many authors' interpretations of ecological networks (Jongman, 2004; Opdam *et al.*, 2006), is multi-habitat or mosaic banks. Many species require more than one habitat type during different stages of their life cycle, *e.g. Bufo calamita* (natterjack toad) requires aquatic sites for reproduction and terrestrial habitats for post-breeding activities such as feeding and aestivation (Miaud and Sanuy, 2005). Such an approach would also be of greater benefit to bank administrators who would be able to offer credits to offset impacts occurring on a wider range of habitat types. Importantly then, this research has identified over 3700 ha of land which would be suitable for the creation of mosaic banks incorporating all four habitat types considered.

Despite the issues identified in locating lowland heath and unimproved grassland habitat banks, when the area of available bank sites is compared to the BAP habitat creation targets for the four local BAPs (LBAP) in the study area (UK BAP Partnership, 2008), it is possible to meet and indeed surpass by a considerable margin all but the lowland heath target. It would only be possible to achieve 75% of the LBAP lowland heath target using the habitat bank sites identified. This result demonstrates that the potential habitat banks have to address ecological and policy requirements as they are able to consolidate and strengthen ecological networks and habitat resources in line with BAP policies and targets, alongside the provision of a location which ameliorates negative ecological impacts from developments.

Determining how the addition of habitat banks in the landscape affects functionality in respect of ecological networks is necessary to quantify the success, or otherwise of the approach developed. The effect on ecological networks was positive for all habitat types considered. Addition of the habitat banks allowed an overall increase in size of each ecological network type to be achieved in excess of the bank area added. The ratio of increase (the effect of one hectare of habitat bank added compared to the increase in size of ecological network in hectares) is positive in all cases: wetland habitats (1:2.72), lowland heath (1:2.15), broadleaf and mixed woodland (1:1.81), and unimproved grassland (1:1.72). If connectivity levels are already relatively high the addition of habitat area in a spatially targeted way results in increased spatial cohesion, this is demonstrated for the wetland habitat and broadleaf and mixed woodland landscapes. Where connectivity is less evident, the size of habitat banks this approach was able to introduce into the landscape resulted in a noteworthy increase in overall ecological network size. The reassessment of the landscape to determine the likely effects of introduced habitat patches as carried out here is often not reported. The reported ratios of increase of ecological network size achievable using this method is large, representing a good ecological return on the investment made and highlighting the value of spatial targeting. Where monitoring has been carried out this is often at the single species scale, *e.g.* reinstatement of lowland heath plant-pollinator interaction (Forup *et al.*, 2008), rather than considering implications for entire ecological networks or landscape systems as a whole. There is a recognised requirement to compare predictions developed through spatial models with impacts in the real landscape. Opdam for example, suggests monitoring schemes are set up to consider habitat networks over a long time period (Opdam, 2002). However, the difficulty of achieving this and the cost and monitoring effort required often restrict the level to which this is achieved. In this respect the use of habitat banking would be additionally beneficial as it would require such monitoring in order to determine that adverse impacts had been adequately compensated for.

7.2 Accessible green infrastructure

The importance of ensuring people have access to green infrastructure can be viewed from two broad perspectives: personal - access allows enjoyment, exercise and pleasure derived from outdoor activity, and to promote wider understanding – access and associated education allows the importance of conservation of habitats and associated species, responsible management of the landscape and other economic functions to become known. As members of the local community it is particularly important that local people have an understanding of pressures which may lead to landscape change to enable them to become stakeholders in their local environment and able to express their values, preferences and interests (Agbenyega *et al.*, 2009). A considerable body of research exists which has considered how people use the landscape around their homes, *e.g.* (Bishop, 1992;

Neuvonen *et al.*, 2007; Ode and Fry, 2006; Skärbäck, 2007; Van Herzele *et al.*, 2005). Much of this has been concerned with ensuring parity of access for all sectors of society and particularly in ensuring lack of information or cultural traditions do not prevent active use of green infrastructure, and as such often has a qualitative focus (Barbosa *et al.*, 2007; Comber *et al.*, 2008; Lafortezza *et al.*, 2009). Attempts have been made, however, to quantify the ability of communities in accessing greenspace, most notably through the Accessible Natural Greenspace Standards (ANGSt) (Handley *et al.*, 2003; Harrison *et al.*, 1995). However, whilst these standards are well regarded by non-governmental organisations and local authorities with remits for access to greenspace, they remain non-statutory. This has resulted in no agreed methodology for their application in the landscape. Methods which have been used can be divided into two types: i. the construction of a simple buffer around the greenspace in question followed by counting the number of people living within the buffer zone, *e.g.* (McKernan and Grose, 2007), and ii. use of address points as the starting point to measure how far greenspaces are from individuals' homes, *e.g.* (Barbosa *et al.*, 2007; Comber *et al.*, 2008). Limitations exist with both methods meaning that results are rarely comparable between studies. It is usual to select only one method for use in a study and therefore the differences between methods and the results they give have been difficult to quantify.

This research has used two methods in order to apply ANGSt to the MKSM landscape. Whilst the explicit purpose of this approach is to derive current and future levels of access to natural greenspace enjoyed by local populations, an underlying aim is to advance the debate on methodologies used in the application of ANGSt. Comparing the most simplistic approach with a more rigorous, real landscape method and considering the difference in results achieved allows possible patterns of accessibility to be determined which can be considered in the light of findings from other studies. This research moves beyond an assessment of the current state of accessible greenspace to consider what impact new development, increased population and newly accessible greenspace would have on the overall level of natural greenspace accessibility in the landscape and what links can be made to wider green infrastructure functions.

7.2.1 Current greenspace accessibility

The potential for access to greenspace within the study area, *i.e.* 9.7% of the whole study area, 12.8% of the JCAs directly affected by developments and 12.7% of the buffered development sites, is comparable to that found in other studies. McKernan and Grose (2007) found between 2% and 13% of their area to be accessible greenspace and Comber *et al.* (2008) termed 25% of their study area accessible greenspace. Using two study scales, JCAs with developments planned and buffered development sites (Figure 5-2), allowed areas that are likely to be directly affected, *i.e.* through net loss of accessible greenspace, to be compared to areas predicted to suffer indirect effects such as an increase in greenspace usage. Examining the current percentage of people with access to natural greenspace (Table 7-1) within the two study scales provides a baseline from which the effects of the planned development can be examined.

Table 7-1 Current levels of accessible natural greenspace at the two study scales using two measurement methods

ANGST rule	JCAs with development planned		Buffered development sites	
	buffered greenspace (%)	network analysis (%)	buffered greenspace (%)	network analysis (%)
rule 1	16.61	4.07	13.95	1.86
rule 2	61.79	3.66	71.99	2.30
rule 3	61.95	1.21	67.02	1.09
rule 4	40.66	0.15	56.68	0.25

The results show a complex pattern of association between analysis method and study area combined with frequency of greenspace size. In the larger study area (JCAs with development planned) the buffered greenspace analysis method gives results which are directly associated with the size of the buffer applied, with the percentage of the population with access to greenspace increasing as the buffer size is increased. However, at some point between rules three and four the frequency of very large accessible greenspace sites produces a limiting effect. The paucity of such sites can be seen to restrict the percentage of the population able to access greenspaces. The same pattern can be seen in the buffered development site study area, although the limiting effect occurs earlier, between rules two and three. It is suggested that the smaller the study area the sooner this large site limiting effect will be seen. The buffered greenspace approach can therefore provide information (albeit relatively crude) about the frequency of large sites within the study area. The results using this assessment method are comparable to those reported by

McKernan and Grose (2007) particularly for ANGSt rules one, two and four where difference in values between the two studies were as little as 3.39% (rule 1 – JCAs with development planned) (*ibid.*). This is interesting as the McKernan and Grose study extended to the South East region as a whole (taking in a portion of the MKSM study area). The fact that the results were so comparable raises questions over the buffered greenspace approach as an appropriate assessment. It is possible that this approach is so insensitive to differences that the resulting values provide little insight into the actual level of natural greenspace accessibility. Whilst the simplistic and quick nature of the approach is favourable it is important that such reasons are not weighted more importantly than the validity of results. The network analysis results from the larger study area reveal a negative relationship between the increasing requirement of the ANGSt rule and level of access identified. This reflects both the frequency of accessible greenspace sites of different sizes, with a larger number of smaller sites found, alongside the ability to travel through the landscape and access sites. However, in the smaller study area the same relationship cannot be found. It appears that fewer people are able to access sites of two hectares than sites of 20 hectares. This could be an indication of a lack of small accessible greenspace sites or that there are a higher than usual number of 20 hectare sites. A similar effect can also be detected in the buffered greenspace approach which indicates that the reason is unlikely to be related to a lack of walking routes leading to sites and is more likely an expression of the number of 20 hectare sites.

Overall, the two assessment methods give markedly different results making it very difficult to compare the values or suggest patterns which hold for all. However, when the results are considered spatially the effects of different approaches to travel to greenspace sites becomes clear. Whilst positive results from the network analysis cluster around accessible greenspace sites, the results from the buffered greenspace approach are spread widely across the study areas suggesting a level of access that it optimistic at best (Figure 5-5). Despite this the buffered greenspace approach has to date been the more frequently carried out assessment method. Three reasons are suggested for this;

i. The assessment method is relatively straightforward allowing it to be a time and skills efficient assessment method.

ii. Assessment using the network analysis approach requires the use of specialist spatial analysis software and more detailed access data, and is therefore likely to be more time consuming and thus expensive.

iii. The results from the buffered assessment method give the impression that access to natural greenspace is relatively high with the lowest figures found for smaller local sites. Such a result may be more acceptable politically given that a two hectare site can be created and managed more readily than a 500 hectare site. Such a focus on local sites also fits well with the current local government planning agenda of Local Area Agreements and subsidiarity.

It is, however, clear that both assessment methods reveal that access to natural greenspace is not universally enjoyed and therefore it is likely that investment is required in order to improve this situation. However, the results from the network analysis method, whilst stark reading for those charged with providing and improving access to greenspace, may be of more use in brokering investment deals with developers than the more optimistic results from the buffered greenspace approach as they show a clear and urgent need for change.

7.2.2 Greenspace accessibility in a changing landscape

Comparing the baseline results with a likely future scenario assists in identifying particularly vulnerable areas where accessible natural greenspace is in short supply. The proposed development sites within the MKSM amount to over 9000 hectares, of which 280 hectares can be currently identified as accessible natural greenspace. Alongside this the increase in population associated with the development sites is approximately 250,000. However, when the two assessment methods were applied to this new landscape situation the buffered greenspace approach shows an increase in the percentage of people with access to greenspace at most ANGSt levels. This highlights again the simplicity of the approach, and is really identifying that the increase in population is located within the buffer areas of existing accessible greenspace. This raises the issue of site carrying capacity. Whilst it is not attempted to place a maximum value on the number of people a single accessible greenspace site is able to accommodate, owing to the range of influencing factors, *e.g.* vegetation and habitat types, size and shape of site, use of site (frequency, length of use, time of use and type of use), and popularity of site, it is clear that there is a maximum which if exceeded would lead to a deterioration of site quality. The results from the network analysis show a decrease in levels of access ranging from -0.07% to -0.59%, with only one increase in accessibility seen (0.02% rule 2 JCAs with development planned). These decreases strengthen the argument for strategic investment in green infrastructure at all levels alongside improvements in access to the existing natural greenspace resource.

By introducing the previously identified habitat banks as areas of additional accessible greenspace, an understanding of the impact such areas would have on the balance of the green infrastructure system can be investigated. Two approaches to increasing accessible greenspace were used resulting in 1530 hectares (recreation led approach) and 1558 hectares (biodiversity led approach) being added. The effects of these additions were seemingly small with a maximum of 0.84% (rule 2 and rule 3 buffered development sites) increase in population achieved when measured by the buffered greenspace assessment, and 0.23% (rule 2 buffered development sites) achieved using the network analysis method. Despite the apparently small values this increase represents up to an additional 12,000 people with access to greenspace. However, the values do not match the decrease seen as a result of the combined loss of accessible greenspace and increase in population previously reported. This underscores the significant effects that are likely as a result of the MKSM developments. It is clear that increases in access to natural greenspace can only be achieved through the addition of accessible greenspace of an appropriate size in an appropriate location and linked to the improvement of access routes. Whilst the 280 hectares of accessible greenspace lost during development is only a fifth of that added through habitat banks the location of the original greenspace gave much more value for money than the habitat bank locations in terms of the percentage of the population able to access natural greenspace.

It is clear that from a sustainability perspective the development of new housing areas should occur adjacent to existing conurbations in order that they can take advantage of many services. However, it also needs to be recognised that such a location will undoubtedly result in many small, local accessible greenspaces being lost. The effects of location aside, an increase in population of around a quarter of a million people will always lead to a dramatic change in the way the landscape and greenspaces are used and it would be unwise to think investment in green infrastructure is not necessary in order to provide these new communities with greenspaces to access and enjoy.

7.3 Climate change implications on the landscape

Development of a landscape strategy today without an explicit consideration of climate change is unwise. However, the accuracy of climate data and its use in predictions must be clearly acknowledged with the caveat that the best available data were used at the time of assessment. A considerable amount of research has been carried out into the development of climate change models to consider how ecological systems will be affected (Davis and Shaw, 2001; Parmesan and Yohe, 2003; Sala *et al.*, 2000; Walther *et al.*, 2002). Of particular interest is whether species are likely to gain or lose suitable climate space (Walmsley *et al.*, 2007). This provides an indication of where conservation effort in the landscape is required. One widely applied approach is that of climate envelopes which determine areas of climatic suitability for particular species. However, their use has drawn a number of criticisms particularly where climate envelopes have not been coupled with an examination of the underlying landscape and likelihood of species presence (Baselga and Araújo, 2009; Davis *et al.*, 1998). The MONARCH project was the first large scale attempt to consider climate change in the UK linked to its effect on species persistence. Making the link between suitable climate, suitable habitat and therefore the existence or otherwise of particular species relies heavily on the predictive capacity of models. However, if the presence data of species being considered is based on observed distributions this provides a sound baseline from which to consider how changes may occur in the future. The BRANCH project was developed across North West Europe in order to assess impacts of climate change on species and habitats and to develop strategies for adaptation (Berry *et al.*, 2007). BRANCH considered the combined effects of climate change and habitat fragmentation to determine areas with climate-proof ecological networks. The approach developed in this research extends the BRANCH methodology by considering whether species, given suitable climate space and habitat areas, are able to keep pace with climate change, *i.e.* if their dispersal and colonisation cycle can synchronise with predicted changes in climate.

7.3.1 Suitable climate space availability

Considering how the change in climate will affect suitable climate space over the study period, current – 2050 reveals interesting results particularly for the UK (Figure 6-2a and 6-2b). At the moment, suitable climate space for the woodland ecoprofile is very limited currently however, as time progresses larger portions of the UK become suitable (Figure 6-2a). If the trajectory is extrapolated beyond 2050 there is a strong likelihood that suitable climate space for woodland ecoprofiles will become more evident in the MKSM study area. This result does not take account of all woodland species as there are many woodland species able to thrive in the UK currently, but considers those used in the development of the woodland ecoprofile. Three other woodland species considered in the MONARCH project are shown to be more widespread currently in the UK and have a similar level of resilience in light of climate changes with all showing gains in climate space up to the 2050 time stamp (Walmsley *et al.*, 2007). Harrison *et al.* (2003), determined a mixed response ranging from large gains to little change when considering the impact on climate space for three woodland bird species in the UK, and more generally Kirby *et al.* (2005), whilst determining that frequencies of both specialist and generalist woodland species are likely to increase, identified that woodland species assemblages as they occur at present are likely to breakdown and re-assemble in new ways. The results given here then represent climate space suitability for the woodland ecoprofile in addition to the climate being suitable for existing species, in this way the ecoprofile would be a species gain for the UK.

The wetland ecoprofile shows the UK to be devoid of suitable climate space after 2020, with only a small amount identifiable in the current period (Figure 6-2a). However, climate change predictions have highlighted the likelihood of precipitation pattern changes which in many cases regulate water levels in areas of wetland habitat. Such changes may result in significant impacts on habitat and their associated species. However, Dawson *et al.* (2003), found three wetland plant species were able to gain in suitable climate space even under the UKCIP 2050 High scenario, which predicts a summer soil moisture reduction of 30% across large parts of England (Hulme *et al.*, 2002). It is suggested that if water levels are lowered alongside potentially higher evaporation rates due to higher temperatures plants may experience greater water stress, so despite being able to survive species are likely to be within a compromised habitat (Acreman *et al.*, 2009; Dawson *et al.*, 2003). Therefore, whilst the ecoprofile considered in this research has a lack of suitable climate space it is likely that other wetland species may not suffer to such an extent.

The suitable climate space found in the UK for the grassland ecoprofile is extensive both currently and in 2020 (Figure 6-2b). However, the model indicates that by 2050 there would be a large decrease in area and a pronounced easterly shift. Whilst the remaining areas of suitability previously formed part of the wider extent suggesting at least some original population persistence would be achievable, there are two isolated patches. When considering the effects of climate change on grassland in other research there is a mixed response. Ruderal species were identified to have decreased on grassland sites, suggesting increased winter precipitation (a widely reported impact of climate change in the UK) may favour shallow rooted grasses and allow a more characteristic natural grassland population to remain (Morecroft *et al.*, 2009). However, on ex-arable grassland (circumstances which may be encountered should grassland habitat banks be created on currently arable land) it was determined that increased incidence of summer drought would be likely to have serious implications for the establishment and successional development of ex-arable grassland (Morecroft *et al.*, 2004). Even where habitats have the opportunity to expand into a larger area of suitable climate space the issue of species ability to migrate needs to be considered. Berry *et al.* (2003) identify certain lowland calcareous grassland species that have specific habitat requirements which are not easily met in new areas further north.

7.3.2 Dispersal and colonisation in the landscape

Beyond the examination of the presence or absence of suitable climate space there is a need to determine whether ecoprofiles are able to disperse and colonise the landscape, in order to track or keep pace with climate change. Considering how the functionality of the landscape in 2020 and 2050 compares to that found in the current period requires an understanding of how the ecoprofile is able to disperse through the landscape and when colonisation is achieved. The colonisation ability of the ecoprofiles considered can be broadly divided by habitat type into majority colonised (woodland), around half colonised (grassland) and very little colonisation possible (wetland) (Table 6-2). By the 2050 period the woodland ecoprofile is able to colonise over 97% of habitat patches at both the UK and North West European scales. This is an increase from that which is possible in 2020 and reflects an increase in connectivity of habitat patches in the landscape. Although the colonisation ability of the grassland ecoprofile remains similar in both 2020 and 2050 (around 50%) this masks a dramatic reduction in habitat patch area (-61% in the UK and -79% in North West Europe). Considering the effects of climate change on the colonisation ability of the wetland ecoprofile reveals a serious issue, particularly in the UK where no colonisation events are predicted to occur in either 2020 or 2050. Even in North West Europe the level of colonisation found in 2020 (53.6%) falters by 2050 resulting in no new colonisation. Although there is suitable climate space available, the ecoprofile has colonised all areas in the current period and is predicted to do so in the 2020 period. This suggests the habitat patches available are at, or nearing, their carrying capacity and will limit reproductive capacity. Whilst other species may not suffer such striking results the situation encountered for the ecoprofiles considered in this research are likely for some species. It should also be noted that the characteristics of the ecoprofile used were not overly onerous, *e.g.* the woodland ecoprofile was determined to require breeding habitat patches to be five hectares or above. The results therefore give a clear picture of how quickly a seemingly functional landscape can become dysfunctional and result in conservation concern.

An indicator of whether ecoprofiles are able to keep pace with climate change is the total distance they are able to travel over the full time period compared with the mean movements seen in suitable climate space (Table 7-2).

Table 7-2 Summary of mean total distance (km) of colonisation events occurring over the entire study time period

Ecoprofile	UK Total (ddd)	UK Total (did)	NW Europe (ddd)	NW Europe (did)
Woodland	133.76	105.85	144.69	110.90
Wetlands	114.50	83.26	11.75	11.79
Natural grassland	93.42	92.32	45.12	45.12

The difference between the two approaches to dispersal considered in the climate model show density dependent dispersal largely resulting in a slightly longer dispersal distance. The difference is more pronounced in the UK for all ecoprofiles and in North West Europe for woodland only. Whilst it is possible to determine a single mean distance of colonisation for each ecoprofile, the changes occurring in suitable climate space are multi-directional, *e.g.* in North West Europe the suitable climate space for the woodland ecoprofile extends northwards by over 200 km, retracts from the south by only 0.34 km, contracts from the west by over 300 km and no corresponding eastern extension can be identified (Figure 6-2a). So, although the ecoprofile is largely able to remain within its current range longitudinally, overall there is a large scale reduction in suitable climate space. The results for the wetland ecoprofile, particularly in North West Europe, identify serious issues: a severe restriction of suitable climate space, nearing of landscape carrying capacity and apparent reduction in connectivity of remaining habitat patches as seen by a lack of colonisation events. Whilst the situation for the wetland ecoprofile is serious, the results are very clear. However, the grassland results appear to show an ability to colonise and an adequate mean distance of colonisation achieved over the time period. A number of issues are masked by these results: a large scale change in suitable climate space, both a reduction in area and spatial shift particularly in the UK and areas of North West Europe such as France, Belgium and the Netherlands, alongside which is a reduction in the number and area of habitat patches (greater than -40% in the UK and greater than -60% in North West Europe). The long term viability of wetland and grassland ecoprofiles is therefore, questionable. Although the results do not represent the expected fate of all wetland and grassland species they do provide a trend which cannot be ignored.

The structure of the landscape is thought by many researchers to be of utmost importance when considering how species are able to adapt to changes in climate space. Research has focused on considering the negatively synergistic effects of climate change, habitat fragmentation and low landscape connectivity (Opdam and Wascher, 2004; Travis, 2003). Areas of high connectivity, as measured by frequency of dispersal and colonisation activity in the UK, determined where ecoprofiles were currently able to move through the landscape with relative ease. The results for the woodland ecoprofile (Figure 6-4a) reveal clusters of activity followed by movements which allow new patches to be colonised, followed by clustering of activity in these new areas. This shows a robust and advancing colonisation pattern which strongly correlates with the pace of movement of suitable climate space suggesting the ecoprofile is able to track climate change, at least in the UK. However, the grassland results identify an ecoprofile which initially appears to be well placed to track climate change (Figure 6-4b). The large cluster of activity in the East

Midlands and east of England appears robust (68% of all activity in the UK occurs here), however, over the period 2020 to 2050 changes in suitable climate space result in this area becoming reduced and isolated and a requirement for the ecoprofile to move northwards. Some activity does continue in the north of England and in Scotland, however, it is clear that the ecoprofile is losing the battle to track suitable climate space changes. There is no wetland ecoprofile presence in the UK beyond the current period and therefore the frequency of its activity cannot be assessed.

The ability of ecoprofiles to track climate change is strongly influenced by the connectivity of habitat patches within the landscape. It is necessary that patches are able to span climate temporally. Whilst the suitable climate space information used in this research is simplistic, in that distinct time periods are used, even when climate is considered incrementally species need to be able to move gradually and therefore require a high level of landscape connectivity. This highlights the importance of identifying gaps in the landscape or areas of currently low connectivity in order that conservation effort can be focused there. The long term trajectory of ecoprofiles must also be considered and it should be acknowledged that a landscape may only provide suitable climate space for a limited period; this can be clearly seen in the results of the grassland ecoprofile which shows very fast paced suitable climate space change.

The ability of ecoprofiles to track changes in suitable climate space gives an indication of how species may react to climate change. The use of the 'future' landscape is likely to be very different from how we see species currently occupying areas. This strengthens the validity of investment in habitat fragmentation reduction and an overall improvement in the quality of existing habitats. Whilst movement is likely to be gradual without a higher level of connectivity, so called 'bottle necks' will occur and hinder the ability of species to adapt to their changed circumstances (Vos *et al.*, 2008).

7.4 A sustainable approach to habitat bank location

The development of a methodology which considers a multifunctional approach to landscape assessment has clear benefits. The explicit acknowledgement that drivers and pressures on a landscape are multitudinous, yet need to be both recognised and assessed, represents a starting point in the development of a sustainable landscape approach. However, the complexity of the landscape system means that there is a limit to the number and type of landscape functions which can be combined in such an assessment if the results are to remain useful. Multifunctional approaches to landscape assessment are found in numerous areas of research and range in a continuum from the strongly theoretical (Naveh, 2001; Tress *et al.*, 2004) through to more applied interpretations (Opdam *et al.*, 2003; Wiggering *et al.*, 2006). The multifunctional approach to landscape is a strong theme in green infrastructure planning, being identified as one of the major benefits of such a view of the landscape. Green infrastructure has been identified by some as the ecological framework for environmental, social and economic health (Benedict and McMahon, 2006; Tzoulas *et al.*, 2007). In this respect green infrastructure provides a useful framework in which to develop a landscape strategy driven by large scale economic and social development alongside predicted climate change, but aiming to develop and use a methodology with ecological landscape assessment at its core. Drawing together the results of the three assessments ecological networks, natural greenspace and climate change impacts allows the implications of the landscape drivers and pressures previously identified to be examined. The combined findings of the three assessments also allow a landscape strategy which can respond to such issues as large scale development and predicted climate change to be constructed.

The results of the combined assessment should be viewed as broad indicators of how the landscape could be affected and could respond. Areas of multifunctionality are identified as being preferable to those exhibiting only one function (as determined by the three function approach). This does not preclude the ability of the landscape to function in other ways and it should be borne in mind that the combination of functions examined in this research is by no means exhaustive. The results allow the positive impact a habitat bank could have in the landscape, both currently and in the future, to be determined. The results depicting habitat bank contribution to landscape functionality currently are positive in all cases with areas of multifunctionality seen in all four habitat type landscapes. The woodland and wetland landscapes, however, are revealed to be strongly multifunctional with large areas of the landscape matching their selection criteria. In the woodland landscape in particular such areas appear to have a high level of connectivity. In this respect the habitat banks can be seen to be contributing to a reduction in habitat fragmentation. By considering the situation in the climate change periods 2020 and 2050 it was an aim of this research to examine how habitat banks located optimally now will fair in the future. The lowland heath situation cannot be examined owing to a lack of climate change data. The results for the other habitat types are varied both between habitat and over time. If the 2020 period is considered it appears that grassland is able to retain multifunctionality at the level seen currently (a low level). Woodland and wetland, however, lose all multifunctional areas. By the 2050 time period, grassland and wetland are devoid of multifunctional areas suggesting the habitat banks located based on the requirements of these ecoprofiles are to some extent unsustainable. The woodland landscape reveals an interesting result with an area of multifunctionality being reinstated. This is one of the areas of original multifunctionality and when the advancement of suitable climate space in observed this area can be considered to be the beginning of a wider reinstatement of functionality.

This highlights both an issue and opportunity when using climate change predictions. Whilst the landscape may appear to be of reduced functionality for the ecoprofile initial investigated, it is likely that if the initial ecoprofile has been forced to adjust its spatial niche in the landscape we can assume that species other than those considered may also have been affected in this way. This may allow species which were originally marginal in a particular area to become more numerous or species requiring a larger range shift to use transitional habitat patches. It is important, however, to recognise that all climate space predictions, and indeed the response of species to climate change, become less reliable the further into the future they are projected. Therefore, reassessments of the implications for habitat banks should be carried out frequently. This is not to say that the results seen here are not accurate but that they represent predictive capacity now rather than a definitive result. In addition, habitat banks were added to the landscape at a single time period. If habitat banking was to be adopted as a mitigation mechanism banks would be planned and delivered in a more continuous way.

7.4.1 A multifunctional landscape strategy

Taking the results from the individual assessment approaches and those from the combination of model outputs a landscape strategy can be proposed. This needs to take account of the originally identified drivers and pressures on the landscape, alongside landscape state, impacts on the landscape and finally the responses considered in this research. In the landscape strategy proposed it is necessary to consider both an approach to optimal current location of habitat banks alongside how predictions regarding future climate scenarios may affect and influence the landscape. The strategy is posed as a series of five questions:

i. Where should habitat banks be located?

Locate habitat banks as determined by the ecological network analysis.

- Results from both the ecological network analysis and the natural greenspace assessment show that the specified locations can reduce habitat fragmentation by increasing overall habitat area, increasing habitat patch size and decreasing isolation of habitat patches.
- Increased connectivity can be achieved by stipulating that habitat banks be located such that they become part of larger ecological networks.
- Using the specified locations, habitat banks can have a large positive effect on the size and connectivity of ecological networks and through mosaic banks are able to contribute to more than one type of ecological network, *e.g.* woodland and grassland.

ii. How can habitat banks contribute to a multifunctional landscape?

Habitat banks can increase connectivity levels in the landscape.

- Ecological connectivity gives the opportunity for networks of sites to operate as metapopulations resulting in more robust and resilient populations of species within the landscape. This is of particular importance given the impending and already observed climate change pressures which are likely to affect species persistence.
- Socially, a connected landscape allows an increased number of areas to be accessed providing a greater wealth of experience and contributing to enjoyment and understanding of the landscape. This allows visitor pressure to be spread to ensure individual sites do not become over used

- Habitat banks may also assist species to adapt to climate change by increasing connectivity of habitat or providing transitional habitat patches.

iii. How can habitat banks remain relevant and functional in the future?

The spatial and financial planning of habitat banks at the outset must take into account that landscape changes are inevitable in the future.

- The proposed and future habitat bank locations must be part of a wider ecological network (this criterion was used in the location of habitat banks proposed so far). This provides an insulating effect against loss of connectivity and populations becoming isolated. It also allows the use of habitat banks by species not currently associated spatially with bank locations but which may become so in the future particularly as a result of climate induced range shifts.
- Ideally, habitat banks should ideally be composed of more than one habitat type in order to form mosaic banks with the aim of prolonging their functionality. Assemblages of complementary habitat types provide opportunities for species requiring multiple habitats as part of their life cycle to colonise habitat banks alongside future proofing such sites by allowing them to be linked into a range of ecological networks.
- Should additional funds become available then existing habitat banks should be extended and their quality ensured. This should be carried out with regard to the likely suitable climate space location for species found within both the habitat bank sites and the ecological networks of which the site is a constituent. Such an approach acknowledges the likely need for species to adjust their range as a result of climate change and assists in the retention of connectivity where possible.

iv. How can issues not adequately dealt with by spatial targeting of habitat banks be addressed?

Link the habitat bank approach to existing ecological, biodiversity, greenspace and land use planning strategies.

- Access to natural greenspace could not be entirely achieved in line with ANGSt using habitat banks proposed owing to the scale of development

- The ability of the habitat banks identified to exceed regional BAP habitat creation targets illustrates how this approach could be used to deliver biodiversity and ecological strategies. Habitat banks provide a new funding stream into a sector which has seen significant budget cuts over recent years.

v. Are there any outstanding queries?

The practicality of setting up and running habitat banks remains unclear.

- Whilst this research provides a transparent methodology with respect to the location of habitat banks in order to achieve multifunctional gains in the landscape, bank administration, habitat tradeability and mitigation ratios need to be considered.
- Currently, habitat banks are currently non-statutory and unlegislated for. It has been suggested that successful implementation of biodiversity offsets (a comparable and linked approach) depends crucially on arrangements that provide stakeholders with clearly defined rules and objectives, and are legally, institutionally and financially secure (Treweek *et al.*, 2009).

7.5 Implications and limitations of modelling

The use of a modelling approach within this research means that inevitably there are a number of necessary limitations and compromises. Use of a GIS means that the approach is carried out in a modelled version of the real landscape system. However, this fact is weighed against the ability to consider multiple scenarios and investigate phenomena which would take tens of years to witness in the field. The main limitations of a computer model approach are the accuracy and validity of data. As data ages it becomes a less accurate interpretation of the landscape which it depicts. Data used within this research was the most up to date available, this is particularly the case for landscape habitat data and climate change predictions. However, such datasets will be superseded at some point in the future.

The computing power available to construct and run queries within a GIS represents the other main limiting factor of this research. The use of nine ecoprofiles representing four habitat types was considered to represent the optimum in model run time compared with level of detail of outputs. Given more time and larger computer processor speeds more ecoprofiles might have been constructed. However, the habitats chosen for inclusion in the ecological network analysis represent those identified to be of most relevance and importance according to the sub-regional bodies responsible for BAP monitoring and delivery. Within the climate change assessment the use of a wider range of ecoprofiles to better match those used in the ecological network analysis would allow an increased interpretation of results, particularly for lowland heath. The use of more detailed data showing accessible greenspace would have produced more accurate results. However, at the time such detailed data were not available across the whole study area and it was felt important to develop a methodology that allowed repeatability in different areas and a consistency of data.

Owing to the fact that habitat banking is a relatively new approach to considering investment in the landscape there are still a number of issues with regard to administration, legal issues, responsibilities and impact amelioration in application of mitigation banking theory into practice. More details are still required to determine habitat tradeability, in-kind, out-of-kind, on-site and off-site mitigation and also the ratio of impact to mitigation. Whilst the cost of mitigation was considered in this research this would need to be revisited, particularly in light of the current economic circumstances in the UK. Costs were calculated at the height of the landscape and property price peak in 2008 and therefore may be somewhat inflated compared with the current cost of acquiring land. However, the situation is unlikely to remain the same for a long period and will be sure to continue to fluctuate. With this in mind the values used are considered to be valid but the detailed circumstances should be borne in mind.

There is a complex relationship between the site of a detrimental environmental impact and the location and type of mitigation used in its amelioration. Of particular concern is the location and spatial relationship of these two areas and how this apparently simple question of location can be applied in a real landscape. If a bank site is a considerable distance from the impact site then both ecological and social concerns can be raised. Despite mitigation banking having been carried out in the USA since the 1970s there are still few guidelines regarding distance criteria. It is merely suggested that on-site mitigation is *preferred* and that where off-site mitigation is permitted it should be undertaken "in close proximity and to the extent possible in the same watershed" (US Environmental Protection Agency and Department of the Army, 1993). If applied in a UK context, habitat bank sites should be within the same ecological area, in this research defined by the extent of the JCAs. However, to ensure that theories of environmental justice are adhered to, the communities that will suffer from a net loss of open space as a result of development should have an additional, equal area of open space of similar quality provided in the form of the habitat bank. This would not usually be economically possible and is likely to result in multiple small habitat banks with limited ecological functionality being created, thus removing many of the benefits which can be derived from the pooling of resources to create and manage large strategically placed habitat banks. This factor remains an issue.

7.6 Providing a methodology for locating habitat banks

The location of habitat banks is critical if such an approach is to address the ecological pressures on the landscape whilst also being successful socially and economically. The scale and agents of the ecological network analysis used in this research are a departure from those seen elsewhere. The sub-regional spatial scale allowed the combination of nationally and locally important species and habitats to be considered. The use of multiple ecoprofiles to represent habitat types resulted in a more realistic approach than could be achieved by considering location of habitat patches alone, yet allowed conclusions to be drawn from the modelling approach of relevance to other species associated with particular habitats. The methodology developed in the ecological network analysis part of this research ensured that the consideration of climate change produced both meaningful and spatially relevant predictions and results. By incorporating a well identified need for communities to have access to natural greenspace the methodology developed was able to take on a multifunctional dimension. Within the natural greenspace assessment the comparison of two existing assessment methods allowed their merits to be assessed and the differences of their approach and results to be examined. This allowed the habitat bank locations proposed to be thoroughly examined whilst also contributing to, and furthering, the methodological debate of greenspace assessment. Through this comparison of approaches critical issues could be identified with the simplistic buffered greenspace approach resulting in its use as an appropriate assessment method for accessible natural greenspace being called into question. The looming presence of climate change and its potential impacts on all types of landscape strategy were thoroughly considered and incorporated into this research. The incorporation of climate change into a landscape strategy is influenced to a great extent by the predictive capacity of climate change models. However, this is no reason to omit such a ubiquitous influence. Whilst only generalised trends as to the impacts climate change may have on habitat bank location were possible in this research they clearly add a dimension to the resulting landscape strategy which ensures future landscape pressures are explicit. Indeed, the clear progression throughout the modelling approach, starting from an ecological baseline and appending additional modules, allows the basis for the resulting strategy to be traced. This indicates the potential for the methodology to transpose to alternative locations and for additional modules to be developed in the future. The overall result of the research approach is a methodology with a multifunctional basis as to how habitat bank locations can be spatially defined and moves forward the currently theoretical concept of habitat banking.

CHAPTER EIGHT | Conclusions

8.0 Introduction

The impacts of change on the structure and function of the landscape are wide ranging. Considering the landscape as a system whose functions must be conserved and retained through landscape planning ensures that the link between cause and effect of change is explicit. This is necessary if a multifunctional system is to be achieved which acknowledges the need for economic and social development whilst recognising the importance of green infrastructure. In landscapes subject to large scale economic development there is a strong need to quantify the existing landscape particularly from the perspective of ecological systems, *i.e.* biodiversity and habitats. This ensures that any detrimental impacts can be identified and allows ways of strengthening the landscape to be established. EIA and SEA Directives place a strong emphasis on balancing the impacts of development with compensation through mitigation. Taking this principle further, habitat banking has been used to gain contributions for the creation and management of important habitats and landscapes which have suffered detrimental impacts through developments. Whilst this approach is working well in many countries it is yet to be realised in the UK.

The implementation of habitat banking requires an appropriate methodology in order that:

i. policy is in place in order to *require* habitat banking,
ii. impacts of development and compensatory payments are equitable,
iii. habitat banks are ecologically sound and perform their required functions, and
iv. the approach works in concert with existing landscape planning and management tools.

This study has developed methods for points three and four, for the first time allowing optimal locations for habitat banks to be proposed which contribute to the retention of a multifunctional landscape. The use of GIS allowed the construction of multiple modules which were brought together allowing a comprehensive examination of the landscape system. Types of changes expected alongside functions which could potentially be provided by habitat banks were identified.

Changes

i. large scale built development
ii. proposed increases in population resulting from new developments
iii. existing landscape fragmentation issues
iv. predicted climate changes

Functions

i. species retention through habitat creation and management
ii. ecological network strengthening and improved landscape connectivity
iii. reduction of the effects of habitat fragmentation
iv. transitional habitat patches and additional stepping stones for species required to shift their range as a result of climate change
v. accessible natural greenspace areas for enjoyment, recreation and education

This approach allowed a spatial targeting methodology to be developed for each function being considered which utilised detailed, subject specific, digital information and modelling techniques. Drawing together the results ensured that the proposed locations of habitat banks were able to balance the required functionality. The proposed banks were then compared to existing biodiversity and greenspace access targets to determine the efficacy of the approach in contributing to other landscape planning and management programmes. The success of the habitat bank approach was clearly articulated as it was determined that 75 – 100% of biodiversity targets could be achieved in addition to the amelioration of the initial detrimental landscape impacts.

8.1 Achieving a multifunctional landscape using habitat banks

The ecological network assessment developed uses ecoprofiles to represent a number of species found within the study landscape which are indicative of particular habitat types of interest. The modelling approach used allows the connectivity of the landscape to be assessed and ecological networks to be defined. This allows areas of high and low functionality to be identified and potential locations for habitat banks to be determined. By considering the degree to which functionality improves with the addition of habitat banks, optimal locations can be suggested which are able to contribute most to reducing fragmentation and increasing habitat connectivity. This is particularly clear where habitat banks proposed are constructed from a number of habitat types. These banks are able to contribute to the ecological networks of a wider range of species and result in qualitative improvements to the overall habitat resource. This approach is transposable having been developed using standard datasets and constructed in a transparent way in order that it would be possible to use it in alternative landscapes. This allows it to be more readily linked to existing habitat creation and management approaches, *e.g.* UK BAP, and identifies habitat banking as having the potential to become a significant delivery vehicle for UK BAP targets, in terms of both hectares and funding. Improvements to the approach are possible, with the most obvious being the inclusion of a greater number of ecoprofiles. An increase in number and range of ecoprofiles would result in a more targeted approach. However, a balance must be achieved between the time and computing power spent researching, constructing and 'running' ecoprofiles and the additional insight achieved which would improve habitat bank location.

The accessible natural greenspace assessment used two approaches to determine the current level of people's access to natural greenspace in the study area. The network analysis approach, whilst determining a low level of access, provided an accurate assessment method, particularly in comparison to the alternative buffered greenspace approach. Impacts of developments on accessible natural greenspace were noteworthy and were both direct (removal of accessible natural greenspace sites) and indirect (increase in the population using sites). Although the location of habitat banks did result in increased accessibility to natural greenspace (an increase of up to 12,000 people) it was not possible to ensure all people had access. The level of accessibility identified in the current landscape made this a very unlikely prospect from the outset. However, the level of access stipulated by the ANGSt as being desirable, whilst clearly being aspirational targets, do provide clear, unambiguous and comparative standards of accessibility and, as such, means that they are a very useful planning tool.

Incorporating habitat banks into the climate space tracking abilities of ecoprofiles allowed the future effects of proposed habitat bank locations to be examined. This ability to test the effectiveness of habitat bank location in the face of additional landscape change was very powerful. Individual habitats were identified as being particularly vulnerable to predicted climate change, *e.g.* natural grassland. Such findings allow habitat creation and management approaches to target particular areas to attempt to remedy this situation. In addition to this role, habitat banks also have a role in improving structural and functional connectivity of habitat, a vital aspect of ensuring landscapes are 'climate proofed'. Without a well connected landscape, species whose range must shift as a result of changes to temperatures or weather patterns cannot make such movements. In this way habitat banks can assist by providing additional transitional habitat patches thus increasing the traversability of the landscape. The results of landscape modelling which incorporates climate change predictions have limitations and are restricted by the predictive and computing power available, accuracy of data, scale of data, and, in this research, number of ecoprofiles it was possible to include. However, the inclusion of climate change as a driver of landscape change, and the ability to examine potential impacts on ecological systems, far outweigh the caveats which must be placed on the results.

Clearly, this approach to locating habitat banks and examining how they can contribute to a multifunctional landscape is advantageous. It allows large scale drivers of change; built development and climate change, to be examined and a response strategy proposed which is compatible with a multi-user landscape. It was determined that habitat banks of all types proposed were to some extent multifunctional, with those constructed of woodland and wetland identified being strongly so. There are, however, issues that cannot be addressed by habitat banks, *e.g.* the very low levels of lowland heath remaining in the landscape and the inherently low levels of natural greenspace accessibility in the study area. Habitat banks can, though, be used as part of a strategy to improve these two situations by strengthening ecological networks and providing additional natural greenspace resources.

8.2 Further work and recommendations

There are a number of areas where this study could be extended

1. Improvements to the modelling methodologies could be achieved with the use of additional datasets, specifically

    i. finer scale land cover data detailing broad habitat types, and
    ii. increased accessible greenspace data such as country parks and local authority defined sites.

2. The determination of additional functions performed by habitat banks, and which could be developed for inclusion in the overall approach, would further its applicability as a multifunctional approach. Such functions include, for example, the ability of habitat banks to attenuate local climate or to incorporate hydrology and water management.
3. The current treatment of climate change predictions results in three distinct time snapshots. Whilst this is a useful approach and allows change over time to be considered, continuous predictions would provide more detail and therefore allow a more detailed response strategy to be developed. It may be possible to achieve annual predictive data, which would better match the time scale of planned built development and landscape planning and management programmes.
4. The approach developed could be applied in an alternative landscape(s). This would allow the following aspects of the approach to be re-examined: drivers of landscape change, appropriate habitat types and ecoprofiles, accessible natural greenspace, and ultimately the balance of functions achievable through the location of habitat banks.

The research has also generated several recommendations which can be made regarding the use of habitat banking as a mitigation approach:

1. A clear policy mandate is required if habitat banking is to be achieved; the CIL is likely to be a good opportunity to develop and implement habitat banking in the UK. This would require local authorities to take on the responsibility for bank administration. However, prior to habitat banking becoming an operational approach a number of issues still need to be addressed: replaceable and non-replaceable habitats, the ratio of impacts to habitat bank credits, and a clear habitat bank administration structure.
2. The spatial targeting of habitat banks using a series of linked GIS modules provides a clear and transparent approach. It allows a variety of drivers to be considered alongside the landscape functions affected by the development and likely to benefit from habitat bank creation. Using green infrastructure as a framework ensures that a balance between landscape functions is

This study has fulfilled its aims of examining and developing a method to locate habitat banks in order to respond practically to landscape change, habitat fragmentation and the degradation of functional landscapes. It has highlighted the importance of considering the many interlinked functions of the landscape in order to develop a balanced and appropriate strategy. The use of GIS has demonstrated the importance of spatial targeting and its use should be seen as an invaluable element in the habitat banking process. The inclusion of climate change predictions has shown the importance of considering future landscape changes when proposing landscape plans. It is important that the findings of this research, including the methodology for spatial targeting of habitat banks, are shared in order to contribute towards taking forward the concept of, and beginning to employ, habitat banking as a landscape scale management approach in the UK.

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Appendices

APPENDIX ONE | Key Characteristics of JCAs

87 East Anglian Chalk (The Countryside Agency, 1999a)

- Distinctive, open, variable topography of the Chalk, a continuation of the Chilterns.
- Large-scale rolling downland, mainly arable, with distinctive beech belts along roads and in hilltop clumps and ash-dominated woodland.
- Long straight roads, open grass tracks, isolated 19th century white or yellow brick farmhouses and distinctive nucleated villages, generally within valleys.
- Few large towns (Baldock, Royston and influence of Cambridge) on major transport routes and enlarged commuter villages which still retain their rural character.
- Generally muted colour range with distinctive white soils and building materials but relatively lively landform.
- Manicured character of stud landscape around Newmarket, with domesticated smaller-scale settled landscape to the east with rows of pine.
- Significant linear ancient or Roman earthworks: Devil's Dyke, Fleam Dyke and Icknield Way.

88 Bedfordshire and Cambridgeshire Claylands (The Countryside Agency, 1999b)

- Gently undulating topography and plateau areas, divided by broad shallow valleys.
- Predominantly an open and intensive arable landscape. Fields bounded by either open ditches or sparse closely trimmed hedges both containing variable number and quality of hedgerow trees.
- River corridors of Great Ouse and Ivel compose cohesive sub-areas characterised by flood plain grassland, riverine willows and larger hedges.
- Woodland cover variable. Clusters of ancient deciduous woods on higher plateau area to north-west between Salcey and Grafham Water. Smaller plantations and secondary woodland within river valleys.
- Settlement pattern clusters around major road and rail corridors (A1 and M1) many with raw built edges. Smaller, dispersed settlements elsewhere. Village edge grasslands an important feature.
- Generally a diversity of building materials, including brick, thatch and stone. Limestone villages on the upper Great Ouse.
- Man-made reservoir at Grafham Water. Restored gravel working lakes adjacent to river Ouse, and water-bodies in Marston Vale resulting from clay extraction.
- Brickfields of Marston Vale and Peterborough form a major industrial landscape. Mixed extraction, dereliction and landfill.
- Medieval earthworks including deserted villages the major feature of visible archaeology.

89 Northamptonshire Vales (The Countryside Agency, 1999b)

- Gentle clay ridges and valleys with little woodland and strong patterns of Tudor and parliamentary enclosure.
- Distinctive river valleys of Soar, Welland and Nene with flat floodplains and gravel terraces.
- Large towns of Leicester and Northampton dominate much of the landscape.
- Frequent small towns and large villages, often characterised by red brick buildings.
- Prominent parks and country houses.
- Frequent imposing, spired churches.
- Attractive stone buildings in older village centres and eastern towns and villages.
- Great diversity of landscape and settlement pattern with many sub units, eg Nene Valley and Welland Valley.

90 Bedfordshire Greensand Ridge (The Countryside Agency, 1999a)

- Narrow escarpment formed of Lower Greensand, with distinct scarp slope to north-west and dip slope to south-east.
- Mixed land use on north-west facing scarp slope, including a high proportion of woods (both deciduous and coniferous), heath and pasture. Medium-sized arable and wooded landscape on dip slope.
- Panoramic views to north across claylands.
- Number of historic parklands and estates, including Woburn, Haynes, Shuttleworth, Sandy Lodge and Southill give the impression of a well-tended landscape.
- Settlement pattern includes estate villages and hamlets in folds of ridge. Local materials include ironstone, brick, thatch and render.
- Integrity of area breached by river Ivel valley.
- Existing and redundant sand quarries especially around Leighton Buzzard.

91 Yardley – Whittlewood Ridge (The Countryside Agency, 1999b)

- Broad plateau with shallow soils elevated above adjacent vales.
- A strong historic landscape character, largely due to the continued presence of extensive areas of ancient woodland.
- Mixed land uses of pasture, arable and woodland.
- Generally medium-sized fields with full hedges and hedgerow trees, mainly oak.
- Low density of settlement and consequently few local roads; cut through by major north-south canal, rail and road routes.

92 Rockingham Forest (The Countryside Agency, 1999b)

- Undulating landform rising to prominent scarp along edge of Welland Valley in Rockingham Forest.
- Large woodlands on higher ground enclose the landscape.
- High historic and nature-conservation interest in woodlands.
- Remnants of unimproved grassland throughout, with limestone heaths and fragments of acid bogs in the Soke of Peterborough.

- Foreground views are occupied by large arable fields with low hedges.
- Large mature landscape parks and country houses.
- Dry stone walls around villages, becoming more common in open countryside in Soke of Peterborough.
- Nucleated villages often in sheltered streamside locations.
- Distinctive buildings constructed in local stone: ironstone in west, limestone in east.
- Undisturbed, deeply rural quality despite nearby towns and adjoining trunk roads.
- Prominent, disused ironstone quarries (gullets) and abandoned second world war airfields.
- A sharp transition between the countryside and the main towns of Kettering, Corby and Peterborough (lying just outside the area) which have developed rapidly in recent years.

93 High Leicestershire (The Countryside Agency, 1999b)

- Broad rolling ridges and varied, often steep-sided valleys.
- Well-treed character from hedgerows, hedgerow trees, copses, spinneys and small woodlands, the last often sited on ridges.
- Mixed farming, but with arable mainly on the ridge tops and the wide valley bottoms.
- Sparse settlement of small villages with little modern development.
- Ironstone and limestone churches and vernacular buildings but also abundant brick.
- Frequent and very prominent ridge and furrow and many deserted settlements.
- Green lanes, quiet country and a remote, rural, often empty character.

94 Leicestershire Vales (The Countryside Agency, 1999b)

- Gentle clay ridges and valleys with little woodland and strong patterns of Tudor and parliamentary enclosure.
- Distinctive river valleys of Soar, Welland and Nene with flat floodplains and gravel terraces.
- Large towns of Leicester and Northampton dominate much of the landscape.
- Frequent small towns and large villages, often characterised by red brick buildings.
- Prominent parks and country houses.
- Frequent imposing, spired churches.
- Attractive stone buildings in older village centres and eastern towns and villages.
- Great diversity of landscape and settlement pattern with many sub units, eg Nene Valley and Welland Valley.

95 Northamptonshire Uplands (The Countryside Agency, 1999b)

- Rounded, undulating hills with many long, low ridgelines.
- Abundant and prominent ridge and furrow with frequent deserted and shrunken settlements.
- Sparse settlement of nucleated villages on hilltops or valley heads.

- Mixed farming: open arable contrasts with pasture enclosed by good hedges with frequent hedgerow trees.
- Wide views from the edges and across the ridgetops.
- Straight, wide, enclosure roads, often following ridges.
- Little woodland, but prominent coverts on higher ground.
- Ironstone and limestone older buildings with a transition across the area. Brick buildings in some villages.
- Great variety of landform with distinctive local features like Hemplow Hills.
- Large and nationally-important historic parks.

96 Dunsmore and Feldon (The Countryside Agency, 1999b)

- Farmland with large geometric fields divided by straight hedges with many hedgerow trees.
- Generally well-wooded appearance but also extensive open arable farmland.
- Heathland character still evident in woodland clearings and roadsides.
- Plateau landscape of open, flat, rather empty character, with long views.
- Plateau fringes more enclosed, with rolling landform and woodland more dominant.
- Large ancient woodlands of high nature-conservation value in the west.
- Strong urban influence in some areas.

107 Cotswolds (The Countryside Agency, 1999b)

- Defined by its underlying geology: a dramatic scarp rising above adjacent lowlands with steep combes, scarp foot villages and beech woodlands.
- Rolling, open, high wold plateaux moulded by physical and human influences, with arable and large blocks of woodland, divided up by small, narrow valleys.
- Incised landscapes with deep wide valleys.
- Flat, open dip slope landscape with extensive arable farmland.
- Prominent outliers within the lowlands.
- Honey-coloured Cotswold stone in walls, houses and churches.
- Attractive stone villages with a unity of design and materials.

108 Upper Thames Clay Vales (The Countryside Agency, 1999c)

- Broad belt of open, gently undulating lowland farmland on Upper Jurassic clays containing a variety of contrasting landscapes. Includes the enclosed pastures of the claylands and the wet valley bottoms and the more settled open arable lands of the gravel.
- The valley bottoms, with open floodplain landscapes displaying gravel workings and flooded pits, a regular and well-ordered field pattern, willow pollards and reedbeds along the water courses.
- The Vales in Oxfordshire are dominated by 18th century enclosure landscapes of small woods and hawthorn/blackthorn hedges. Former and current gravel workings along the Thames floodplain also include open water features. The distinctive character of Otmoor with its patchwork pattern of small fields defined by healthy hedgerows of elm add interest and variety to this area.
- In Buckinghamshire, the Vale is a predominantly pastoral landscape including regular fields within a well-defined network of trimmed hedgerows often with oak/ash hedgerow trees and some small blocks of woodland.

- Brick-built buildings within the Vales reflect the widespread use of the local clay as a building material.

109 Midvale Ridge (The Countryside Agency, 1999c)

- Low irregular wooded limestone ridge giving way to a series of isolated steep-sided tabular hills in the east which rise from the surrounding clay vales.
- Large geometrically spaced fields divided by regular pattern of hedgerows and trees supporting both arable and pastoral farming.
- Villages, typically built of local limestone, perched high up on spurs, hilltops and along ridges giving extensive views across the open, gently undulating, clay vales to the north and south.
- Visible archaeology dating from early Roman settlement of the area found on prominent areas of higher ground.
- Spring-line settlements associated with blocks of ancient woodland along the ridge.
- Contrast between the moderately elevated limestone hills and ridges and the surrounding low-lying clay vales.

110 Chilterns (The Countryside Agency, 1999c)

- Chalk hills and plateau with a prominent escarpment in many places, and extensive dip slope with numerous dry valleys.
- Remnants of chalk downland on the escarpment and valley sides. Extensive areas of downland invaded by scrub.
- The most extensive areas of beech woodland in the country on the plateau, and 'hanging' woodlands in the valleys.
- Enclosed and intimate landscapes of the valleys contrasting with the more open plateau top and extensive views from the scarp to the clay vale below.
- Small fields and dense network of ancient hedges, often on steep ground. The agricultural landscape often dominated by hedges, trees and small woodlands.
- Many surviving areas of semi-open common land on the plateau.
- Scattered villages and farmsteads, some of medieval origin, displaying consistent use of traditional building materials including flint, brick, and clay tiles.
- Network of ancient green lanes and tracks including the Ridgeway which links numerous archaeological sites and settlements.
- Frequent grand country houses and designed landscapes occupying prominent positions on sloping valley sides.

APPENDIX TWO | BAP Species

recorded in the MKSM study area in habitats of interest

Species Name	Common Name	Species Group	Habitat association
Bufo calamita	Natterjack Toad	amphibian	Wetland habitats
Burhinus oedicnemus	Stone-curlew	bird	Unimproved grassland
Caprimulgus europaeus	European Nightjar	bird	Lowland heath
Carduelis cannabina	Common Linnet	bird	Lowland heath
Emberiza schoeniclus	Reed Bunting	bird	Wetland habitats
Lanius collurio	Red-backed Shrike	bird	Lowland heath
Lullula arborea	Wood Lark	bird	Lowland heath
Muscicapa striata	Spotted Flycatcher	bird	Broadleaf and mixed woodland
Boloria euphrosyne	Pearl-bordered Fritillary	insect - butterfly	Broadleaf and mixed woodland
Hesperia comma	Silver-spotted Skipper	insect - butterfly	Unimproved grassland
Lysandra bellargus	Adonis Blue	insect - butterfly	Unimproved grassland
Melitaea athalia	Heath Fritillary	insect - butterfly	Lowland heath
Catocala promissa	Light Crimson Underwing	insect - moth	Broadleaf and mixed woodland
Catocala sponsa	Dark Crimson Underwing	insect - moth	Broadleaf and mixed woodland
Dicycla oo	Heart Moth	insect - moth	Broadleaf and mixed woodland
Heliophobus reticulata	Bordered Gothic	insect - moth	Unimproved grassland
Hemaris tityus	Narrow-bordered Bee Hawk-moth	insect - moth	Unimproved grassland
Jodia croceago	Orange Upperwing	insect - moth	Broadleaf and mixed woodland
Mythimna turca	Double-line	insect - moth	Broadleaf and mixed woodland
Pechipogo strigilata	Common Fan-foot	insect - moth	Broadleaf and mixed woodland
Polia bombycina	Pale Shining Brown	insect - moth	Unimproved grassland
Rheumaptera hastata	Argent & Sable	insect - moth	Broadleaf and mixed woodland
Trisateles emortualis	Olive Crescent	insect - moth	Broadleaf and mixed woodland
Xestia rhomboidea	Square-spotted Clay	insect - moth	Broadleaf and mixed woodland
Asilus crabroniformis	Hornet robberfly	insect - true fly (Diptera)	Unimproved grassland

Dorycera graminum	Picture-winged fly	insect - true fly (Diptera)	Unimproved grassland
Arvicola terrestris	European Water Vole	terrestrial mammal	Wetland habitats
Barbastella barbastellus	Western Barbastelle	terrestrial mammal	Broadleaf and mixed woodland
Muscardinus avellanarius	Hazel Dormouse	terrestrial mammal	Broadleaf and mixed woodland
Pipistrellus pipistrellus sensu lato	Pipistrelle bat	terrestrial mammal	Wetland habitats

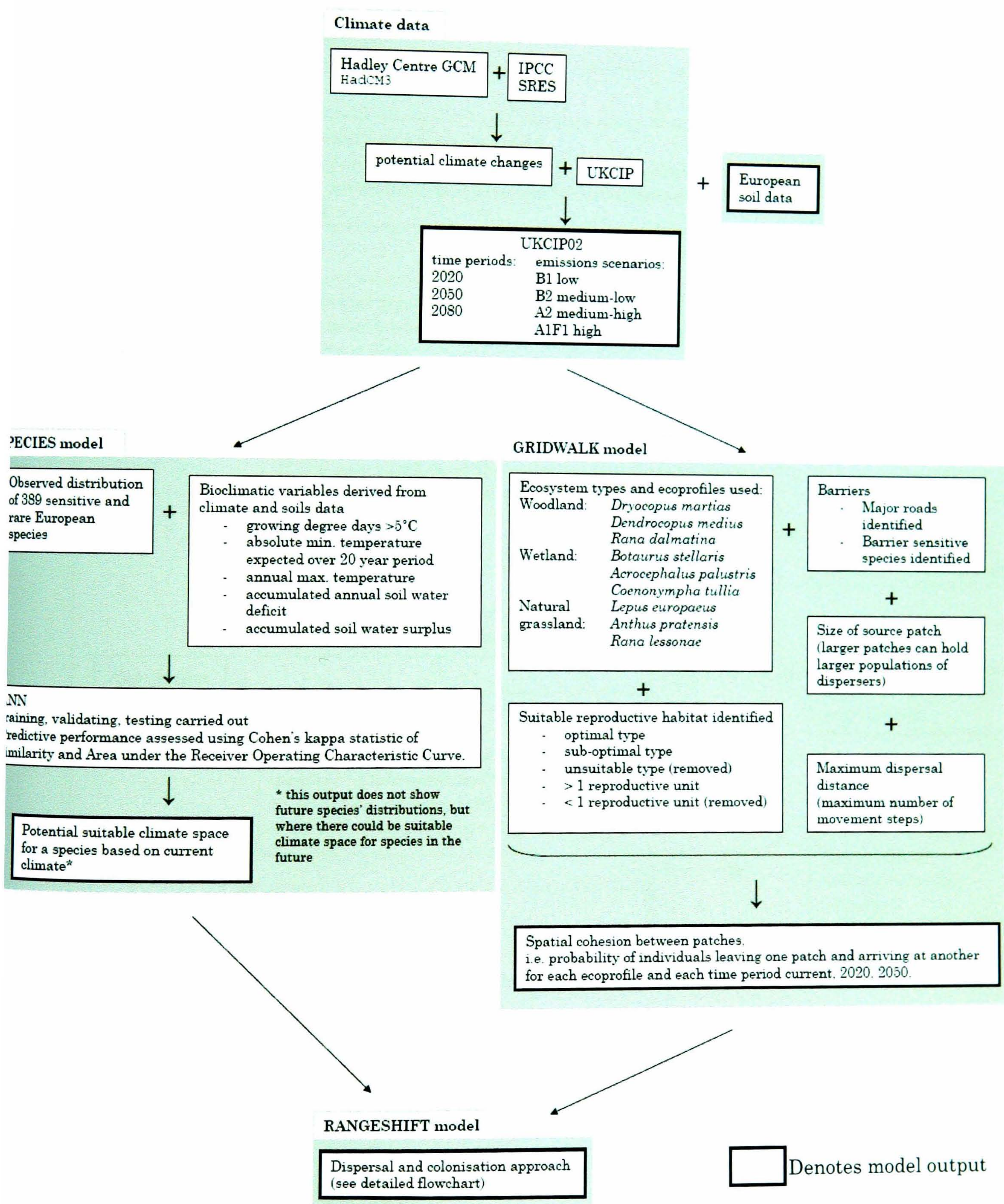
APPENDIX THREE | GIS datasets

Habitats suitable for each ecoprofile

Ecoprofile	Habitat	Min. polygon area (Ha)	Dataset reference
Wetland habitat - A	Coastal and floodplain grazing marsh	0.5	(English Nature, 2002a; English Nature, 2002d; English Nature, 2003; Ordnance Survey, 2006)
	Fens	0.5	
	Reedbeds	0.5	
	Marsh, reedbeds and swamp	0.5	
Wetland habitat - B	Coastal and floodplain grazing marsh	1.0	(English Nature, 2002a; English Nature, 2002d; English Nature, 2003; Ordnance Survey, 2006)
	Fens	1.0	
	Reedbeds	1.0	
	Marsh, reedbeds and swamp	1.0	
	Marsh, reedbeds, swamp and rough grassland	1.0	
	Marsh reedbeds, swamp, rough grassland and scrub	1.0	
Lowland heath – A	Lowland heath	1.0	(English Nature, 2001d; English Nature, 2002b; Ordnance Survey, 2006)
	Purple moor grass and rush pasture	1.0	
	Heath	1.0	
	Heath, rough grassland and scrub	1.0	
	Heath and scrub	1.0	
Lowland heath – B	Lowland heath	4.5	(English Nature, 2001d; English Nature, 2002b)
	Purple moor grass and rush pasture	4.5	
Unimproved grassland – A	Lowland calcareous grassland	2.0	(English Nature, 2001b; English Nature, 2002c; English Nature, 2004a; English Nature, 2004b)
	Lowland dry acid grassland	2.0	
	Lowland meadow	2.0	
	Upland hay meadow	2.0	
Unimproved grassland – B	Lowland calcareous grassland	0.8	(English Nature, 2001b; English Nature, 2002c; English Nature, 2004a; English Nature, 2004b)
	Lowland dry acid grassland	0.8	
	Lowland meadow	0.8	
	Upland hay meadow	0.8	
Broadleaf & mixed woodland – A	Lowland beech and yew woodland	1.0	(English Nature, 2001a; English Nature, 2001c; English Nature, 2002e; English Nature, 2004c; Forestry Commission, 2002; Natural England, 2008; Ordnance Survey, 2006)
	Lowland mixed deciduous woodland	1.0	
	Upland oak woodland	1.0	
	Wet woodland	1.0	
	Ancient woodland	1.0	
	Broadleaf woodland	1.0	
Broadleaf & mixed woodland – B	Lowland beech and yew woodland	20.0	(English Nature, 2001a; English Nature, 2001c; English Nature, 2002e; English Nature, 2004c; Forestry Commission, 2002; Natural England, 2008; Ordnance Survey, 2006)
	Lowland mixed deciduous woodland	20.0	
	Upland oak woodland	20.0	
	Wet woodland	20.0	
	Ancient woodland	20.0	
	Broadleaf woodland	20.0	

Broadleaf & mixed woodland – C	Lowland beech and yew woodland	0.8	
	Lowland mixed deciduous woodland	0.8	
	Upland oak woodland	0.8	
	Wet woodland	0.8	(English Nature, 2001a; English Nature, 2001c; English Nature, 2002e; English Nature, 2004c; Forestry Commission, 2002; Natural England, 2008; Ordnance Survey, 2006)
	Ancient woodland	0.8	
	Broadleaf woodland	0.8	
	Mixed woodland	0.8	
	Shrub	0.8	
	Scattered broadleaf woodland	0.8	
	Scattered broadleaf woodland and scrub	0.8	
	Broadleaf woodland and scrub	0.8	
	Scrub	0.8	

APPENDIX **FOUR** | Climate modelling approach



The flowchart depicts the activities which take place in a one year time step in the RANGESHIFT model.