

**Natural capital and environmental justice: A socio-spatial
analysis of ecosystem services in England**

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Chapter 4 of this thesis is based on the above peer-reviewed paper. In addition, sections of the paper have been incorporated within the Literature Review in Chapter 2, methodology in Chapter 3 and discussion in Chapter 8.

I am the principal author of this paper undertaking development of the concept, data retrieval, analysis and writing of the manuscript. My co-authors were my PhD Supervisors and I received their advice in the development of the research of the paper. They further provided feedback and edited the drafted manuscripts.

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Abstract

Poorer communities tend to be located within lower quality natural environments, experiencing greater environmental burdens and fewer benefits. These environmental benefits are critical for human health and wellbeing and can be effectively conveyed as ecosystem services derived from natural capital. This thesis presents a multiscale spatial analysis in England which addresses a knowledge gap regarding the social distribution of ecosystem services and natural capital assets in a high-income country context. Understanding how equally natural capital and ecosystem services are distributed is important for informing their equitable management, required by the Convention on Biological Diversity.

Nationally, differences in deprivation are found across a natural capital classification of districts. Higher deprivation is often associated with a lower natural capital, but the pattern is not consistent for all types of natural capital or places. This implies equitable management of ecosystems should be driven at a local level.

To realise this, case study analysis is needed and is carried out for three regions; Leeds, Northampton and the South Pennines. Analysis addresses three ecosystem services, thus accounting for the flows of benefits from natural capital including from assets outside the district boundaries.

The distribution of air pollutant removal is unequal across all regions; service is lower for more deprived areas. Ecosystem services are also lower for more deprived areas in Leeds with respect to surface water runoff reduction, and in Northampton with respect to recreation. Indicating their social distribution is location dependent. Sensitivity tests further show that that social distribution of ecosystem services may depend on how they are quantified.

This thesis provides evidence of some inequalities in the social distribution of ecosystem services, emphasising the need to better account for inequalities within management of natural capital. The mixed results demonstrate a need for further distributional analysis of ecosystem services encompassing more locations and services.

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Glossary

1. Terms

Abiotic	Physical, non-biological (components/processes).
Biodiversity	Variety and variability of living organisms.
Biotic	Living (components/processes).
Bluespace	Visible water, usually considered within an urban context.
Ecosystem	A community of living organisms, their interactions and the non-living components of their environment.
Ecosystem services	The direct and indirect contributions of ecosystems to human health and well-being
Ecosystem service flows	The spatially explicit transfer of the ecosystem service from its source to the area which it benefits.
Environmental Inequality	Differences in environmental conditions (hazards and amenities) across different social, demographic or ethnic groups of the population.
Greenspace	Areas of non-agricultural vegetated land, such as parks, sports fields, woodland, and other undeveloped vegetated land. Where unspecified 'greenspace' refers to public greenspaces, gardens are considered 'private' greenspace.
Health Inequality	The avoidable differences in human health across social, demographic, ethnic or location based groups of the population.
Natural Capital	The stocks and assets of the natural environment
Procedural justice	Concern with the 'fairness' and transparency of processes and meaningful participation in decision making.
(Social) deprivation	The extent an individual or community lack necessities for a 'reasonable' life e.g. financially, employment, living conditions, education and other services.
Social distribution	The distribution (or 'share') of a phenomena across different social groups.

Socio-ecological A linked system of humans and the natural environment comprising of multiple units, actors, processes and institutions.

2. Abbreviations – general terms & datasets

ANGSt	Accessible Natural Greenspace Standard
BAME	Black, Asian and Minority Ethnic (population)
CIR	Colour Infrared Imagery
CN	Curve Number
EIA	Environmental Impact Assessment
EJ	Environmental Justice
ES	Ecosystem Service
GIS	Geographical Information System
GLUD	Generalised Land Use Database
HOST	Hydrology of Soil Types dataset
IHDTM	Integrated Hydrological Digital Terrain Model
IMD	Index of Multiple Deprivation
IPC	Integrated Pollution Control
LAD	Local Authority District
LCM	Land cover Map
LISA	Local Indicators of Spatial Association
LSOA	Lower Super Output Area (small area census unit)
MAUP	Modifiable Areal Unit Problem
MENE	Monitor of Engagement with the Natural Environment (survey)
NDVI	Normalised Difference Vegetation Index
NFM	Natural Flood Management
PES	Payment for Ecosystem Services
PROW	Public Rights of Way
RoFMS	Risk of Flooding from Multiple Sources
SA	Sustainability Appraisal

SBA	(Ecosystem) Service Benefitting Area
SEA	Strategic Environmental Assessment
SPA	(Ecosystem) Service Providing Area
SPU	(Ecosystem) Service Providing Unit
SSSI	Site of Special Scientific Interest
SWRR	Surface Water Runoff Reduction

3. Abbreviations - organisations and projects

3.1 International

CBD	Convention on Biological Diversity
CICES	Common International Classification of Ecosystem Services
EC	European Council
EEA	European Environment Agency
ESPA	Ecosystem Services for Poverty Alleviation
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
MA	Millennium Ecosystem Assessment
PEER	Partnership for European Environmental Research
PRESS	Partnership for European Environmental Research on Ecosystem Services
TEEB	The Economics of Ecosystems and Biodiversity
UN	United Nations
UNECE	UN Economic Commission for Europe
USDA NRCS	United States Natural Resources Conservation Service
US EPA	United States Environmental Protection Agency
WAVES	Wealth Accounting and Valuation of Ecosystem Services
WFD	Water Framework Directive
WHO	World Health Organisation

3.2 National (UK/England)

CABE	Commission for Architecture and the Built Environment
CEH	Centre for Ecology and Hydrology
CSDH	Commission on Social Determinants of Health
COMEAP	Committee on the Medical Effect of Air Pollutants
DCLG	Department for Communities and Local Government
DEFRA	Department for Environment, Food and Rural Affairs
DWI	Drinking Water Inspectorate
EA	Environment Agency
HMG	Her Majesty's Government
iCASP	Yorkshire Integrated Catchment Solutions Programme
LNP	Local Nature Partnership
MENE	Monitor Engagement with the Natural Environment survey
MHCLG	Ministry of Housing, Communities and Local Government
NCC	Natural Capital Committee
NERC	Natural Environment Research Council
NE	Natural England
NIA	Nature Improvement Area
ONS	Office for National Statistics
OS	Ordnance Survey
PHE	Public Health England
SEPA	Scottish Environment Protection Agency
UKNEA	United Kingdom National Ecosystem Assessment
UKNEAFO	United Kingdom National Ecosystem Assessment Follow-On Phase
25YEP	25 Year Environment Plan

Preface

This thesis is the outcome of a Natural Environment Research Council (NERC) Doctoral Training Partnership (DTP) funded research project, as made available through the Panorama (previously the Leeds-York) DTP. Situated within the context of Panorama DTP's 'biosphere' research, this project aligns with the aim of helping "society to benefit from natural resources" and "manage our changing natural environment" whilst using an increasingly interdisciplinary approach to research as determined by NERC's remit (<https://nerc.ukri.org/research/>). This research has been undertaken in collaboration with Natural England.

Chapter 1 Introduction

1.1 Background and research justification

The natural environment meets basic human needs, enhances wellbeing and is inherently connected to people culturally. Its forces threaten humans whilst also offering protection from its natural hazards. Environmental change and degradation has been a longstanding concern (see for example Marsh, 1874), but societal pressures, in particular from population growth and consumption, now increasingly threaten the health of the world's ecosystems with biodiversity "declining faster than any time in human history" (IPBES, 2019 p.2). The latest Global Assessment on Biodiversity and Ecosystem Services depicts a critical situation; "The biosphere, upon which humanity as a whole depends, is being altered to an unparalleled degree across all spatial scales". Furthermore adverse changes are occurring despite international commitments, such as the EU's Biodiversity Strategy which aimed to halt the loss of biodiversity and the degradation of ecosystem services by 2020 (EC, 2011).

Explicit framing of human's dependency on nature as a series of 'Ecosystem Goods and Services' provided by 'Natural Capital' aimed to emphasise the importance of ecosystems to human health and wellbeing. Thus compelling greater action to prevent further degradation to ecosystems (although preserving ecosystem services is not necessarily consistent with conserving biodiversity; Harrison et al., 2014). Specifically, the Millennium Ecosystem Assessment describes Ecosystem Goods and Services (ES) as the "*benefits people obtain from ecosystems*" (MA, 2005). The ES are further grouped by the MA (2005) with respect to the different ways they contribute to human health and wellbeing;

- **Provisioning ES** are those which provide essential goods including water and those for nutrition and shelter.
- **Regulating ES** are those which help to moderate potential hazards
- **Cultural ES** are non-material benefits which may be important to people spiritually or for enhancing wellbeing
- **Supporting ES** are the ecological functions and processes which underpin the other services

Natural Capital is defined as *“the world’s stocks of natural assets which include geology, soil, air, water and all living things”*. It is considered as the capital from which humans obtain the wide range ecosystem services which provide economic, social, cultural and environmental benefits (Natural Capital Coalition, n.d.). Whilst a contested notion, defining the natural environment as a type of ‘capital’ facilitates integration of the value of the natural environment within economic decision-making processes (Costanza et al., 2017). Thus ultimately, the ES and natural capital conceptualisations are intended to promote more sustainable decision making and can now be considered mainstream within environmental policy (Schleyer et al., 2015).

Sustainable decision making is concerned with addressing a range of global challenges with the overarching ambition to create a better future for all. Such vision is defined by the UN’s 17 Sustainable Development Goals to which 193 countries are signatories. Notably, five of these goals explicitly address the need to reduce inequalities, in particular Goal 10 which has a broad remit to *“Reduce inequality within and among countries”*. Addressing inequalities is crucial as more equal societies are demonstrated to be more successful overall socially, economically and ecologically (Wilkinson & Pickett, 2009). Inequalities may exist in relation to income, health, environmental conditions, access to decision making and access to public resources. These factors are inter-related. Thus inequalities in health are socially determined by (amongst other factors) the variable conditions in which people live and work, including the socio-economic, cultural and environmental setting that influences health and wellbeing (Marmot et al., 2010). In consideration of natural capital and ES, poorer and ethnic minority communities tend to be located within lower quality natural environments, experiencing greater environmental burdens and fewer environmental amenities (Agyeman et al., 2016). Moreover the burdens from declining conditions in the natural environment are disproportionately borne by poorer communities, thus exacerbating existing inequalities (Islam & Winkel, 2017).

Concern with these environmental inequalities is the focus of the Environmental Justice (EJ) discourse, which examines issues of fair intra-generational, inter-generational and inter-species distribution in environmental conditions and in meaningful access to, and consideration within, environmental decision making. Environmental Justice has been defined as *“Equal access to a clean environment and equal protection from possible environmental harm irrespective of race, income, class, or other differentiating feature of socio-economic status”* (Cutter, 1995). Whilst a broader definition is given by the United States Environmental Protection Agency; *“Fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with*

respect to the development, implementation & enforcement of environmental laws, regulations and policies. Fair treatment means no group of people should bear a disproportionate share of the negative environmental consequences resulting from industrial, governmental and commercial operations or policies” (US EPA, n.d.).

Thus there are three key components of EJ relating to distributional, procedural and recognition justice. To date, research has centred on distributional concerns, with a wealth of studies demonstrating inequality in the distribution of environmental hazards (Walker, 2009). More recent widening of the scope of EJ research has similarly revealed evidence of inequality in the distribution of environmental benefits. This is of concern given the contribution of greenspace (private or public vegetated areas such as woodland, grassland, street trees) to health and wellbeing, and thus potential exacerbation of health inequalities (Jennings et al., 2016).

Analysis of the inequalities in environmental benefits tend to focus on availability or coverage of greenspace in particular in urban areas (Jennings et al., 2016). These studies assume that those who benefit from these greenspaces are located locally. Whilst this holds true for some benefits derived from greenspaces, the ES framework demonstrates that there are a many benefits which may be derived from those greenspaces, some of which may also be of benefit to people located distant from the greenspace. For example, vegetation can reduce the risk of flooding to communities downstream (Villa et al., 2014). On this basis, the contribution of a range of natural capital beyond urban greenspace is important for the distribution of environmental benefits. Framing analysis of inequalities in environmental benefits in terms of ES can facilitate a clearer, more structured and more representative understanding.

Additionally, it can explicitly account for the spatial dependencies in the delivery of benefits to human health and wellbeing from the natural environment. Despite these advantages, there are few inequality analysis of the distribution of ES (Lakerveld et al., 2015), particularly within high-income countries where there is less direct dependence on local environments for the provisioning ES vital for fulfilling basic human needs. Addressing this knowledge gap through socio-spatial analysis of the distribution of ES can provide a new insight with regards to the inequalities of environmental benefits.

With regards to the ES discourse, the concept is fundamentally based on nature’s contribution to human health, but spatial analysis of ES have focused on the production of services with much less attention given to the beneficiaries of ES (Geijzendorffer et al., 2017). There are few empirical analysis addressing distributional justice issues in the natural capital and ESs literature although recently there have been calls for ES analyses to address the complexity of the socio-ecological system and recognise the

importance of benefit distribution (Daw et al., 2011; Bennett et al., 2015; Schröter et al., 2017). Such calls for equitable management are rooted in international policy, specifically the Convention on Biological Diversity (CBD) which aims for; “the conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources, including by appropriate access to genetic resources“ (CBD, 1992; Article 1).

Spatial analysis is critical to understanding and managing the interface between humans and nature through for example land planning and management of natural capital (Blaschke, 2006). Understanding the spatial connections between where ES are produced, where the beneficiaries are, and how equally the ES are distributed across different sub-groups of the population is therefore essential if we are to manage natural capital in a sustainable and equitable manner. Evaluation of the social distribution of ES thus extends current approaches in ES spatial analysis in line with providing the necessary information for decision making in line with the CBD (1992) and the UN's Sustainable Development Goals (2017).

This research aims to:

Determine the social distribution of natural capital and ecosystem services in England, United Kingdom.

Assessments of the social distribution of ES are rare in high-income countries, including England, which is chosen as an appropriate study area due to the extensive, but as yet largely unconnected, past work on natural capital, and environmental inequality. A well-established EJ literature has demonstrated inequalities in environmental hazards (Lucas et al., 2004) and in greenspaces (CABE, 2010) with respect to socio-economic status, which are of concern in the context of persistent health inequalities (PHE, 2018). From a policy perspective, the UK has embraced a natural capital/ES approach to management of the natural environment (HMG, 2018) and has made international commitments to fairness in planning and management of the natural environment (UNECE Aarhus Convention, 1998; CBD, 1992). Together this demonstrates a need for greater understanding of how specific ES are socially distributed. Whilst the research is situated in England, there is broader relevance to the ES and EJ discourses with respect to the approach taken and implications for sustainable decision making.

This research is focused upon the distributive aspect of EJ, employing multiscale spatial analysis to facilitate analysis nationally and sub-nationally. The spatial extent of case study regions and scales of analysis are chosen to be relevant for spatial planning

and to ensure adequate data are evaluated for robust assessments of inequality. More detailed information with regards to the study areas are provided in subsequent chapters.

1.2 Thesis structure

This section describes the structure of the thesis, and summarises the content of its chapters. Further detail, including the research objectives and justifications for the approach taken, is provided in the relevant chapters.

Chapter 2 presents a review of literature and details the research objectives. The context for the research is provided through summaries of the historic development of the ES and EJ discourses. The extensive array of research in these fields necessitates a subsequent focus upon the research most relevant to this analysis. This includes spatial assessments of ES and of analysis of environmental inequalities.

Chapter 3 provides an in depth account of the research design, developing the conceptual framework for analysis from review of existing ES frameworks and modifying these to account for the distribution of benefits from ES. The structure of the research and the corresponding objectives outlined in Chapter 2 (section 2.5) are driven by the spatially dependent aspects of ES and inequality analysis, which are discussed in detail in this chapter. Once the research structure is established the chapter continues by clarifying the scope of the research and providing an overview of the methodological approach. This includes consideration of the selection of ESs and natural capital for analysis, the approach to mapping ESs and the methods applied for assessing inequality. All these are discussed in the context of relevant literature.

Chapter 4 is the first of four analytical chapters and is based upon published work (Mullin et al., 2017). This is a local authority district level analysis analysing national patterns of natural capital and deprivation, and corresponds to objective 1. It indicates the type and quality of natural environments available at the district level and whether these characteristics relate to levels of deprivation. As a generalisation of environmental benefits it provides a broad insight into the potential distribution of ES (derived from natural capital) nationally. It thus provides the context for more detailed work, and is used to inform the selection of the subsequent regional ES case studies. Given the different scales of analysis, methods adopted in Chapter 4 necessarily differ to the approaches taken in Chapters 5-7.

Chapters 5 - 7 Present results of the analysis of the social distribution of three ecosystem services in three case study regions. Chapter 5 commences with

descriptions of the case study sites which are selected to provide insight across districts with different natural capital deprivation profiles. Analysis of these case studies, Leeds, Northampton and the South Pennines, also generates new knowledge relevant for the local areas.

Results for each ES across the three case studies are each the basis of a separate chapter. Thus Chapter 5 addresses air pollutant removal, Chapter 6 addresses recreation and Chapter 7 addresses surface water runoff reduction. The chapters are structured to first provide ES specific information regarding data and methods, followed by presentation and description of analysis outputs.

Chapter 8 discusses the evidence, generated across Chapters 5-7, on the social distribution of ES. This chapter commences with a review of findings seeking commonalities and differences across ES and case studies and provides further context for interpreting the results, in particular that provided by the district level analysis of natural capital in Chapter 4.

Chapter 9 concludes the thesis, confirming the broader relevance of the research prior to a synthesis and summary of findings with respect to the main research aim. The chapter also addresses the supplementary research questions established at the outset, highlighting insight with regards to new knowledge generated and the wider implications of the research findings. These complement and add the necessary detail to the main research aim. This chapter also highlights the key take home messages for policy and practice, and the main contributions to the development of ES and EJ research in England, and more widely, before discussing further research needs.

Chapter 2 Literature Review

Chapter 2 provides a more detailed introduction to the ecosystem service (ES) and environmental justice (EJ) research fields, including their development historically, the characteristics of empirical research undertaken and the limitations to current work. Section 2.1 addresses ES and section 2.2 addresses EJ, and section 2.3 reviews the current, relatively limited literature which jointly addresses ES and EJ, specifically inequalities in ES. Section 2.4 provides an account of current knowledge of both fields in England, providing the context for this research, of which the aims and objectives are detailed in the final section (2.5).

2.1 Ecosystem Services

2.1.1 A brief history of ecosystem services

Whilst the notion of human's reliance upon the natural environment is not new, the continued decline of natural habitats, biodiversity and resources led to the re-framing of its role as a series of benefits provided to humans which could and should be valued (King, 1966; Odum and Odum, 1972; Schumacher, 1973; Westman, 1977; Braat et al., 1979). Whilst these texts include the terms 'natural capital' and 'ecosystem services', it is Paul R. Ehrlich and Harold A. Mooney who are often attributed with coining the term 'ecosystem services' in their 1981 book "Extinction: The Causes and Consequences of the Disappearance of Species" (Ehrlich & Mooney, 1981).

Alongside these theoretical developments, political support for sustainable development increased towards the end of the 20th Century, moving to the forefront of international agenda at the 1992 UN Rio Earth Summit (Cornell, 2011), where commitments to the Convention on Biodiversity (CBD) were made (CBD, 1992). In this context, during the 1990s, the concept of ecosystem services (ES) became more apparent in the literature, in particular through seminal works by Costanza & Daly (1992), Perrings et al. (1992) and Daily (1997). Concurrently, the field of 'ecological economics', an area of economics concerned with environmental resource use which developed in the 1960s (Pearce, 2002) from earlier concepts such as Arthur Pigou's theory of externalities (Pigou, 1920), became more prominent in the 1980s (Costanza

et al., 2017). Costanza et al. (2017) emphasise that ecosystem services are integral to ecological economics research, which conceptualises the economic system as operating within the bounds of the natural environment (see Braat & de Groot, 2012 for a review of ES development in the distinct fields of economics and ecology).

A detailed discussion of the integration of ES within economic concepts and practice is provided by Gómez-Baggethun et al. (2010). It is the economic perspective of ecosystem services which sets it apart from traditional approaches to conserving the natural environment but which simultaneously attracts critiques (Chaudhary et al., 2015). Thus whilst arguably advancing the agenda of recognition of nature's importance to human health and wellbeing (Costanza et al., 2017), Costanza et al.'s (1997) assessment of the global economic values of ESs and natural capital was also criticised for 'putting a price' on nature and for producing an unrepresentative value. Nonetheless, this can be considered a critical juncture at which ESs became a mainstream concept (Costanza et al., 2017 detail press coverage). Chaudhary et al. (2015) also go beyond the high-income country focus of the majority of ES historical accounts to recognise that, in 1997, another key development occurred in Costa Rica where national policy was the first to initiate a Payments for Ecosystem Services initiative.

ES based research began to rise after this point, but it wasn't until after the publication of UNEP's Millennium Ecosystem Assessment (MA) in 2005, that literature examining ESs proliferated (Seppelt et al., 2011). Drawing on an extensive research base, the MA provided insight and structure to the idea of nature underlying human health and wellbeing, harmonising with the broader sustainability agenda. The ES framework established in the MA (2005), despite developments and critiques (which are reviewed in detail in Chapter 3, section 3.1.1), has endured in current research and practice.

In the years since, academic literature utilising the concept of 'ecosystem services' has consistently increased (Figure 2.1). In 2018 5,336 peer reviewed articles contained the term 'ecosystem services' based on a search of the Web of Science database. As the term has gained popularity this is increasingly likely to include articles that briefly use the term without fully integrating the concept of ES throughout the work. Nonetheless, its use is significant as it demonstrates the popularity and the scope of topics which ESs encompasses. Figure 2.1 also indicates the acceleration of ES based research in 2012 coinciding with the establishment of the journal 'Ecosystem Services' (Braat & de

Groot, 2012). By 2018, this was ranked by Scopus' Citescore system as having the most cited articles in the area of 'Nature and Landscape' Journals.

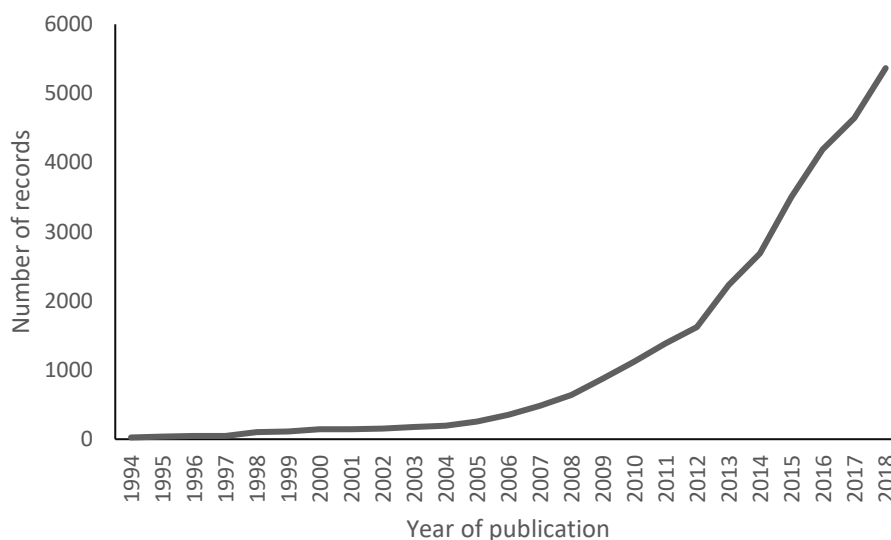


Figure 2.1 Increase in use of the term 'Ecosystem Services' within academic literature since 1994. Source: Web of Science search results, based on the search terms 'Ecosystem' AND 'Service'. Total number of records found = 30,589.

Similarly in the policy arena, an ES approach was given international support through the establishment of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) in 2012 (IPBES, n.d.). In particular in Europe, the EU has driven large-scale projects such as PRESS (Partnership for European Environmental Research on Ecosystem Services) (Maes et al., 2011) and The Economics of Ecosystems and Biodiversity study (TEEB, 2010). These collaborations sought to advance the conceptual underpinnings and practical implementation of ESs with emphasis upon application in policy and economics. Nationally, and aligning commitments to the EU's Biodiversity Strategy 2020, numerous European countries have carried out national ecosystem assessments (NEAs) which incorporate assessments of ES, although achieved to varying standards (Schröter et al., 2016). Such work has also been carried beyond the EU, with China's (Ouyang et al., 2016) assessment for the period 2000-2010 being one of the world's largest evaluation of the status and changes in ecosystems. To help unify the growing diversity of ES research

the European Environment Agency (EEA) also developed the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2013).

Whilst science and policy have driven the natural capital and ecosystem service concepts, better accounting for the natural environment in decision-making also requires involvement of businesses. The link to business is vital given their direct dependence and impact on natural capital assets, for example as primary users of some services (e.g. forest timber, commercial fisheries), as beneficiaries of a wider range of ES such as those which protect business assets from natural hazards and as investors in large scale projects (TEEB, 2010). This naturally lends itself to use of the term 'natural capital'¹ in particular given a focus upon valuing a company's natural assets. Chapter 3 section 3.1.1 and section 3.2 provide a more detailed breakdown of account of what natural capital and ES encompass. Internationally, examining how the concept of natural capital can be utilised within business, the Natural Capital Coalition set out its Natural Capital Protocol in 2016. As another example, Wealth Accounting and Valuation of Ecosystem Services (WAVES) is a partnership led by the World Bank to implement Natural Capital Accounting within ministries for planning and finance and central banks globally (WAVES, n.d.).

The rapid growth in popularity of the natural capital and ES concepts has not gone unquestioned. The validity of conceptualising nature as 'capital' and 'services' is considered by some as too anthropocentric and utilitarian and it is argued that ethics not economics should primarily inform justifications for conservation research (Foster, 1997; Potschin and Haines-Young, 2011). Placing a value on an ES (monetary or otherwise) can neglect its social construction and does not fully account for nature's intrinsic values (Wallace, 2007). Whilst respecting these concerns are legitimate, it is believed by many that these are outweighed by the potential of the ES framework to give "political" purchase (Luisetti, 2014, p.685) to environmental concerns, facilitate communication with relevant stakeholders (Guerry et al., 2015) and to open new opportunities for funding (Goldman et al., 2007). Moreover the use of imperfect techniques is preferable to the omission of ESs (Troy and Wilson, 2006; TEEB, 2010), especially in the context of growing environmental challenges.

¹ see Missemer, 2018 for review of the development of the concept of natural capital

The concepts of ecosystem services and natural capital have thus sought to formalise age old ideas of human's reliance in the context of a modern era where decision making is dominated by neoclassical economics and where the traditional conservation focus has not had enough impact to reverse an overall trend of deteriorating natural environments. Although the idea of 'valuing' nature has not gone unchallenged, the concepts have nevertheless been widely adopted in research and policy. From the early research which formulated the concepts, and used them to illustrate the importance of the natural environment to human health and wellbeing, research over the past 20 years has sought to develop, refine and explore the uses of the concepts, and particularly in the last decade, has considered how to implement the concepts in practice. The next section (2.1.2) therefore briefly addresses the links between ES and human health, and the subsequent section (2.1.3) summarises the key aspects of ES research, and in particular highlights the current challenges faced.

2.1.2 The contribution of ecosystem services to human health and wellbeing

ESs are conceived of with respect to their contribution to human health and wellbeing. Human health is defined as "a state of complete physical, mental and social well-being" by the World Health Organisation (WHO, 1948). The world's ecosystems provide the basic resources necessary for life and also the resources that enhance human health and wellbeing (WHO, 2015). The contribution of ESs should however be set in a broader context in recognition of the other 'social determinants of health' which include socioeconomic, cultural and individual factors (Dahlgren and Whitehead, 1991). Barton's (2005) model of the determinants of health and wellbeing in neighbourhoods (Figure 2.2) effectively conveys these factors within seven spheres, from a global to individual level, which combine to create sustainable and healthy settlements. Of these, the global natural environment forms the outer sphere, emphasising its position as a critical dependency for healthy people and communities. The local environment is presented as the next sphere, as the setting within which the built environment is located, where activities take place and from which local resources may be drawn. The inner spheres convey the contributions of the economy, local communities and of individuals to health and wellbeing. Thus Barton's model conveys the multiple

conditions and activities which may moderate or facilitate the impacts of the natural environment on, and otherwise contribute to, an individual's health and wellbeing.

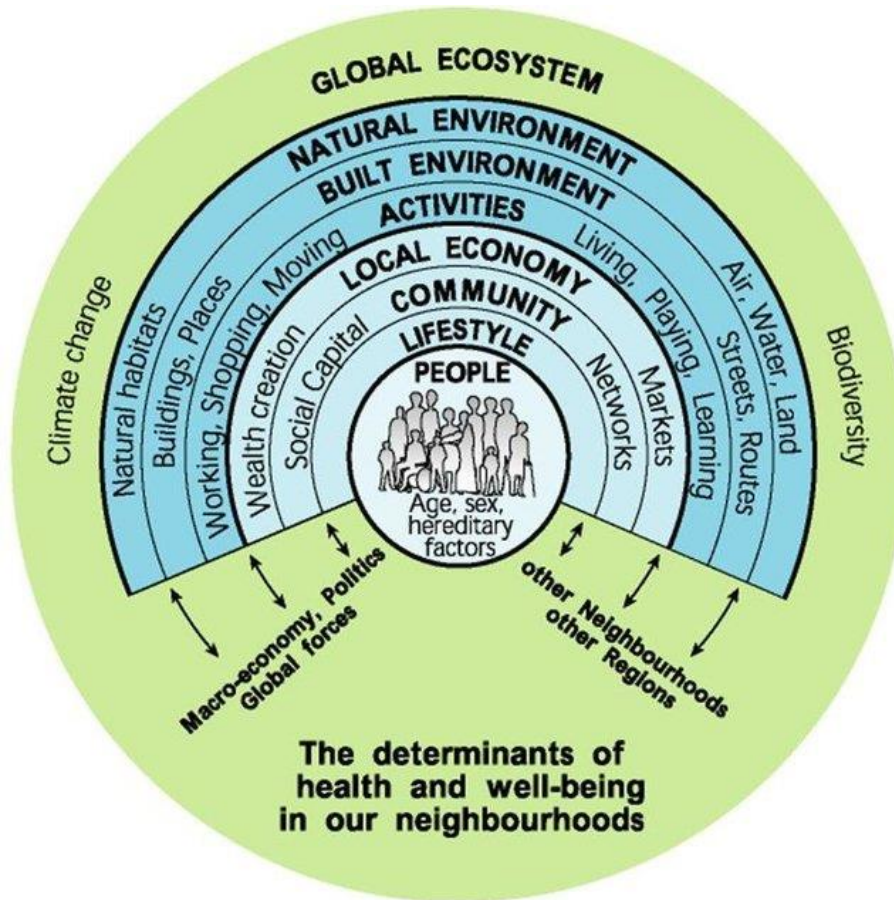


Figure 2.2 Determinants of health and wellbeing in settlements (Barton, 2005)

Evidence of specific linkages between ESs and health are summarised in Table 2.1, and have been previously reviewed by Jackson et al. (2013). The contribution of ESs to human health in high-income countries is mediated by the determinants highlighted by Barton (2005). Therefore, the direct contributions are often dominated by aspects which enhance health and wellbeing as opposed to providing basic resources, to which their contribution is indirect. It is for this reason that the ES-health impacts which are highlighted in Table 2.1. focus upon regulating and cultural ESs rather than provisioning ESs, which are primarily moderated by socioeconomic conditions and activity.

Table 2.1 Ecosystem services and links to health and wellbeing. Impacts of regulation and cultural ES are given with a focus on a high-income country context. Provisioning ES are explicitly linked to basic needs of food, water and shelter and are thus not explored further. * Impact of negative environmental conditions to which ES ameliorate.

Ecosystem Service	Relevant natural capital/setting	Response to environmental setting	Links to health and wellbeing	Health Concern	Source
Recreation	Accessible public greenspace Good quality public greenspace (e.g. facilities available, lower incivilities)	Increase in physical activity Increase contact with nature.	Increase in 5-year survival rate for older people Decreased rates of depression. Lower obesity Reduced symptoms of ADHD Better development of motor skills in children Reduced feelings of stress Increase levels of microbiota Improved social cohesion	Obesity Child development Mental health	Humpel et al. (2002) Takano et al. (2002) Addy et al (2004) Biddle and Ekkekakis (2005) Duncan and Mummery (2005) Giles-Corti et al. (2005) Coen and Ross (2006) Gordon-Larsen et al. (2006) Bird (2007) Nielsen and Hansen (2007) Cooper et al. (2008) Taylor et al.(2009) Coombes et al. (2010)
Other cultural - spiritual/aesthetic/	Presence of natural features (within living/working environment and viewshed) Moderately open grassy spaces interspersed with trees.	Increase in preference for view Increase contact with nature.	Increase feelings of calmness, relaxation Increase in 5-year survival rate for older people Reduction in health complaints Increased resilience e.g. to poverty Reduced stress Increased recovery from operations Increase in self-reported levels of happiness Improved mental health	Mental health	Ulrich (1984) Schroeder and Green (1985) Kaplan & Kaplan (1989) Butterfield and Relf (1990) Randall et al. (1990) Purcell et al (1994) DeVries et al. (2002) Wells & Evans (2003) Bird (2007) White et al. (2013)

Air purification	Increased levels of vegetation; leaf area, canopy cover Dependent on configurations and species	Decrease in pollutants - lower risk of exposure to polluted air	Living close to busy roads associated with 50% increase in risk childhood asthma* Long-term exposure to air pollutants, especially particulates increases mortality.* Air pollution linked to negative effects on pulmonary, cardiovascular and neurological systems* Reduced hospital admissions	Mortality rates Cardiovascular & pulmonary diseases	Pope et al. (2002) Powe and Willis (2004) Nowak et al. (2006) Tiway et al. (2009) COMEAP (2010) Nowak et al. (2014)
Noise regulation	Increase in vegetation cover e.g. trees near roads	Decrease in noise pollution - lower risk of exposure	Traffic sounds penetrating homes and schools impairs verbal and reading development of children* Disruption of sleep patterns, reduced concentration, increased stress. High traffic noise linked to heart disease and higher mean blood pressure. Noise levels >80dBA linked to lower social cohesion and aggression, >120dB to hearing impairment.	Child development Mental health	Regecova and Kellerova (1995) Babisch et al. (2000) Ng (2000) Stansfield et al., 2000) Haines et al. (2001) Ising and Kruppa (2004)
Climate regulation (local)	Presence of vegetation. Trees in urban environments; leaf area and transpiration rates.	Modifies microclimate - less risk of exposure to extreme heat i.e. 'urban heat effect'	Excess summer deaths associated with rapid changes in heat* Higher air temperatures linked to poorer air quality. Heatwaves relate to several illnesses including heatstroke and heat exhaustion.	Mortality rates Heat related illnesses	Chang et al. (2007) Heisler et al. (2007) NHS (2014)
Hazard regulation (flood control)	Presence of wetlands, peatlands, vegetation cover, permeable surfaces	Reduces risk of exposure to flooding which may expose individuals to water, waterborne pollutants, restrict access to healthcare, cause structural damage, restrict access to clean water	Deaths have occurred as a result of flooding, including in the UK* Risk of injury, disease, infection, stress associated with flood events Flooding can exacerbate or provoke mental health issues. Flooding damage to infrastructure can limit access by emergency health services.	Mortality rates Mental health Security	Wheater (2006) Horwitz and Finlayson (2011) Zsomboky et al. (2011) Brown and Murray (2013) Menne and Murray (2013)

2.1.3 Broad trends in ecosystem services research

ES research is highly varied, drawing on a multiplicity of disciplines but dominated by the physical sciences. A Web of Science search for the term “ecosystem services” reveals the vast majority (78%) of publications are related to environment science/studies and ecology, with a further 5% related to various physical sciences and 4.6% assigned to multidisciplinary and economics fields respectively². Despite the essence of ESs being the link between nature and people, and a growing awareness of ESs beyond environmental fields, in particular in policy, there remains a lack of social science research with the ES discipline (Fisher, 2008; Nicholson et al. 2009).

Types of ES studies include but are not limited to conceptual development of frameworks (e.g. De Groot et al., 2002; Jackson et al., 2013; Villa et al., 2014) mapping and assessment (e.g. Baró et al., 2016), valuation (e.g. Troy & Wilson, 2006), linkages with decision making (e.g. Daily et al., 2009), investigation of the links between biodiversity, natural capital, ecosystem functions and the production of ESs (e.g. Balvanera et al., 2006), and measuring the impacts of ESs on health and wellbeing (e.g. Jackson et al., 2013). Studies may be concerned with a single ES (e.g. air pollutant removal; Nowak et al., 2006), ESs produced within specific habitats (e.g. forests; Aznar-Sánchez et al., 2018), identifying trade-offs and synergies between ESs (e.g. Raudsepp-Hearne et al., 2010, Casalegno et al., 2014), ES supply only (e.g. Egoh et al., 2008) or supply and demand (e.g. Nedkov et al., 2012), establishing or evaluating Payment for Ecosystem Services (PES) schemes (e.g. Farley and Costanza, 2010) or developing modelling and assessment tools (e.g. Villa et al., 2014).

Daily (1997) and the MA (2005) laid out the conceptual basis of ESs, providing lists of ES and describing their association with human health and wellbeing. Subsequently much research has evaluated, modified and refined the conceptualisations (e.g. Wallace, 2008; Fisher et al., 2009) but ultimately are adaptations of this earlier work. One consequence has been the production of an array of definitions, conceptual models and approaches to ES assessment which some consider to cause confusion potentially limiting application of the concept in practice (Martínez-Harms and Balvanera, 2012). Conversely, this an important process in a nascent field which joins multiple themes and also demonstrates the adaptability of the concept for differing

² Note that this in broad categorisation, derived from web of science analysis, articles were not individual reviewed to verify their focus.

perspectives (Crossman et al., 2013). Further evaluation of ES conceptualisations, classifications and definitions is given in Chapter 3. Perhaps one of the most influential conceptualisations since the MA (2005) is the cascade model proposed by Haines-Young and Potschin (2011). This describes the flows of ES from habitats and ecosystem functions through to benefits and values for humans. This has formed the basis of multiple assessments (Potschin-Young et al., 2018) but has attracted criticism for underrepresenting non-natural capital inputs and socio-economic feedbacks which co-produce and alter ES and their values (Van Riper and Kyle, 2014; Chaudhary et al., 2015). The lack of socioeconomic perspective is perhaps understandable given the previous observation of low involvement in ES research by social scientists. However, in recent years increasing attention has been given to ensuring representation of these interactions, in particular drawing on the socio-ecological systems literature (Reyers et al., 2013; Andersson et al. 2015; Partelow & Winkler, 2016). Operationalising these complex conceptual models will however require greater multidisciplinary research.

Application of the ES concept is wide-ranging and far-reaching, but several reviews have identified where efforts have concentrated and where key challenges remain. The total number of individual ES usually mapped within a study is five or less (Seppelt et al., 2011) although some simpler approaches facilitate mapping of greater numbers (e.g. Burkhard et al., 2012). The selection of ES assessed tends to reflect the dominance of disciplines operating within the ES discourse, the availability of data and methods and context, but often lacks justification. The most frequently mapped ES include carbon storage/sequestration, food production, recreation, regulation of water flows and provision of water (Egoh et al., 2012; Martinez-Harms and Balvanera, 2012). In urban areas, air pollutant removal (e.g. Escobedo and Nowak, 2009; Jim and Chen, 2009; Gomez-Baggethun et al., 2013) and local cooling from trees and shrubs (Shashua-Bar and Hoffman, 2000) are also commonly mapped (Haase et al., 2014).

Despite being largely overlooked within the MA (2005), urban areas are increasingly examined as a key interface for ES, where the need for ES is considered to be greatest and availability of natural capital lowest (Haase et al., 2014). However, there remains a lack of research examining marine ES (Liquete et al., 2013). Vihervaara et al. (2010) also established a lack of attention to cultural ecosystem service assessments in earlier studies. Whilst this has been more widely addressed in the past decade, the focus on easier to quantify cultural ESs (e.g. recreation, ecotourism, educational values) means that knowledge of some of the less tangible but potentially most important services to humans (e.g. spiritual) remains limited (Milcu et al., 2013). Nevertheless, there is much research taking place in parallel (e.g. psychological) disciplines, which is directly

relevant to ESs but where the ES term is not specifically used (Milcu et al., 2013). These could be better utilised to account for plurality of views in the assessment of all ES (Wallace, 2007; Van Riper et al., 2014).

Most assessments are carried out at the regional or national level; there are much fewer local or site scale studies (Shägner et al., 2013), and assessments at multiple scales is particularly rare. The prevalence of regional/national studies correlates with the more frequent use of coarser resolution secondary data rather than the collection of primary data (Martinez-Harms and Balvanera, 2012) and with the use of proxies as the most popular method for mapping provision of ESs (Shägner et al., 2013). More detailed discussion of methodologies for mapping ESs is given in Chapter 3 section 3.3.1.1. However, a lack of multiscale evaluations is problematic given the cross-scale interactions of ES, the context dependencies of management of natural capital, the diverse scales of social and ecological processes and of ES trade-offs and synergies (Birkhofer et al., 2015; Raudsepp-Hearne & Peterson, 2016; Spake et al., 2019). In response, there have been some notable advances in developing techniques for the re-scaling and multi-scale analysis necessary to encompass a full range of knowledge into ES assessment (e.g. Raudsepp-Hearne & Peterson, 2016; Graham et al., 2019; Spake et al., 2019).

The majority of ES assessments focus upon the supply of ES (service produced by ecosystems) as opposed to their demand (the need or desire for an ES) (Syrbe and Walz, 2012; Bagstad et al., 2013; Burkhard et al., 2014; Haase et al., 2014). Again, this may be partially attributed to the environmental as opposed to social science dominance within ES research, but this imbalance is being addressed as the research base evolves (e.g. García-Nieto et al., 2013; Stürck et al., 2014; Schulp et al., 2014; Parrachini et al., 2014; Baró et al., 2015). Where empirical ES research has better incorporated a social dimension, it is generally with respect to its impact on the production of, and demand for, provisioning ES only (e.g. Hamann, et al., 2016; Dittrich et al., 2017). There remains little empirical understanding about the diversity of stakeholders, their motivations and preferences for a wide range of ecosystem services, or of the distribution of benefits across different social groups considering their demand for a service (Bennett et al., 2015). In addition, more evidence and integration of the mediating effects of human technology and infrastructure upon the production of ES is needed, particularly within urban areas (Birkhofer et al., 2015).

Further complication arises with the divergent understanding of ES demand and how it may be quantified (Wolff et al., 2015), different approaches are detailed in Chapter 3, section 3.3.1.1 and with respect to the individual ES within the analysis Chapters 5-7.

Mapping ES demand is critical for several reasons notably that some (primarily regulating) services (e.g. noise reduction) only exist where there is demand, and where there is low demand the ES may not be considered critical. It is also necessary for land planning and management to understand where current or future demand is unmet (Wolff et al., 2015). Consideration of demand necessitates acknowledgement of the different scales of ES which is essential for understanding how management of natural capital in one location may impact on people's health and wellbeing in another (García-Nieto et al., 2013; Geijzendorffer & Roche, 2014). Ultimately, knowledge of demand and the beneficiaries of ESs is necessary to understand the impacts on health and wellbeing (Bagstad et al., 2013; Villa et al., 2014). Therefore mapping demand in conjunction with supply is important for generating analysis relevant for policy and facilitates sustainable management of ES (for example, it can be used to examine thresholds, Bennett et al., 2015). The lack of attention given to demand is therefore likely a factor leading some to conclude that the ecosystem services discourse has had limited impact upon land management policy and decision making (Vihervaara et al., 2010; Milcu et al., 2013; Guerry et al., 2015).

Other critiques have highlighted a lack of evidence regarding the causal links between biodiversity and ecological structure and functions and between ESs received and the actual impact upon health and wellbeing, which is often assumed rather than demonstrated (Milcu et al., 2013; Bennett et al., 2015; Guerry et al., 2015). Despite considerable research regarding these health links (Balvanera et al., 2006), the measurable changes tend not to be incorporated within wider assessments relevant for management since these tend to be at coarser and larger scales than the evidence relates to (Quijas et al., 2012; Portman, 2013), whilst knowledge of ecological linkages in an urban context remains limited (Kremer et al., 2016).

There are however also good examples of research which is better aligned with challenges in management and planning. Several studies evaluate impacts of different policies or approaches to managing natural capital upon ESs, for example from rewilding, maintenance of Sites of Special Scientific Interest (SSSIs) or large scale conservation schemes (Christie & Rayment, 2012; Hodder et al., 2014; Cerqueira et al., 2015). Others model impacts from land cover change scenarios (Eigenbrod et al., 2011; Bateman, 2013), thus informing policies for sustainable outcomes and future provision of services (Birkhofer et al., 2015). The ES framework is also particularly useful for revealing trade-offs and synergies in outcomes (Raudsepp-Hearne et al., 2010; Howe et al., 2014; Anderson et al., 2015), which can also enable appraisal of particular management interventions (García-Nieto et al., 2013). Despite considerable

ES research being driven by international and national organisations seeking to understand how to implement the natural capital/ES approaches, there is little evidence of how results from ES based policy compares to other approaches (Bennett et al., 2015).

Historically both uncertainty analysis and validation of results have been neglected in ES studies (Eigenbrod et al., 2010; Seppelt et al., 2011; Hou et al., 2013; Schulp et al., 2014). Increased evaluation and communication of uncertainties and validation of ES research would better ensure its reliability and applicability for decision-making. ESs deal with 'messy' problems, and some approaches to their assessment, such as land cover based proxies (e.g. Burkhard et al., 2012), have been criticised as overly simplistic and even erroneous (Eigenbrod et al., 2010). Jax et al. (2017) from their evaluation of 27 case studies which aimed to improve the operational function of ES based approaches (the EU-funded OpenNESS project), recommend that the complexity of the problems should be accepted if they are to be pertinent for decision making. Evaluating and communicating uncertainty can help achieve this (Hamel & Bryant, 2017) and is essential for making robust contributions to scientific knowledge. In particular economic values and maps of ES, are risk of being perceived as 'fact', and thus awareness of their reliability is critical (Hauck et al., 2013). There has been some action to address this issue, as shown by incorporation of uncertainty analysis within more recent ES assessments (e.g. Grêt-Regamey et al., 2013; Kandziora et al., 2013; Zhao & Sander, 2018), however currently such testing is still inconsistent.

Overall ES research has rapidly expanded since 2005. Such rapid growth has created a diverse literature, but one which has neglected certain aspects including the need to examine ES demand, and the beneficiaries of ES in addition to supply, closer engagement with social science, more advanced modelling techniques and estimations of uncertainty, and through these better integration with land planning and management. Many of these knowledge gaps are now starting to be addressed, but they continue to be a challenge going forward.

2.2 Environmental Justice

2.2.1 Development of the environmental justice discourse

Early observations of inequality in environmental conditions were reported by A.M. Freeman in 1972 who established that poorer communities and racial minorities had greater exposure to pollution in the United States (Freeman, 1972). However, the issue of environmental justice (EJ) took hold politically in the 1980s through a civil rights

movement (Mohai et al., 2009). A report “Toxic wastes and race in the United States” by the United Church of Christ (1987), which demonstrated that ethnic minority populations were subject to disproportionate environmental risk, may be considered a pivotal moment which increased the profile of environmental inequalities. A growing evidence base, for example Bullard (1983, 1990), Gould (1986), Charlier (1993) and Chakraborty and Armstrong (1997)³, ultimately led to a 1994 US Presidential Executive Order requiring each federal agency to establish an environmental justice strategy (The President, 1994).

Internationally, recognition of environmental justice as a requirement for sustainable development was first formalised in the 1992 Rio declaration (UN, 1992) and subsequently by the UNECE Aarhus Convention (1998) which required signatories to protect “ ***the right of every person of present & future generations to live in an environment adequate to his or her health & well-being***” (Article 1, UNECE, 1998). Three principal components of the convention; access to environmental information, public participation and access to justice in environmental matters were later translated into EU Directives (2003/4/EC, 2003/35/EC) and regulation (N° 1367/2006).

The academic basis of EJ first developed in the US, with a focus upon distributional inequalities. This principally considers whether environmental hazards are disproportionately located proximal to low-income and ethnic minority populations (Brown, 1995). Early debates examined whether inequality is also evidence of discrimination (Been and Gupta, 1997), for example, were polluting facilities actively located in certain communities or did those communities move to areas of lower environmental quality? (Charlier, 1993; Been and Gupta, 1997; Pastor et al., 2001). Thus leading to broader set of EJ concerns constituting procedural justice (Cutter, 1995; Mitchell and Walker, 2007). Procedural justice examines the deeper social, economic and political production of inequalities within complex historical and

3 Reviews of environmental justice research include Mohai et al (2009) and Agyeman et al.

geographical settings (Holifield, 2012). It addresses questions such as how communities are involved in the decision making process, whether there is sufficient access to information for all, and what remedial action would be appropriate when unfair conditions exist (Cutter, 1995; Agyeman and Evans, 2004; Mitchell and Walker, 2007). However, ultimately judgement of whether injustices exist is highly dependent upon the normative construct of what 'fair' is; whether a utilitarian, egalitarian, social contractarian or liberal view is taken.

From the late 1990s, EJ research extended beyond the US (e.g. Cambra et al. 2013; Pearce & Kingham, 2008; Stevens et al. 2008; Viel et al., 2011; Walker et al., 2003; Wong et al., 2016; Chakraborty & Basu, 2019) encompassing a broader set of issues. In Europe, in particular the UK where earlier European research was concentrated (Elvers et al., 2008), EJ has primarily been a top-down rather than grassroots agenda (Agyeman, 2002). There has also been greater focus on injustices incurred by lower socio-economic groups than ethnic minority groups (Mitchell and Walker, 2007) and an introduction of concern for inequalities in environmental benefits as well as harms (Agyeman and Evans, 2004). Interest in inequalities in environmental benefits has now taken hold in many countries (Schüle et al., 2019). This includes distribution analysis of access to public greenspaces (Astell-Burt et al., 2014; Xiao et al., 2017; Hoffiman et al., 2017; Wüstemann et al., 2017), urban tree cover (Van Herzele et al., 2005), urban greenspace cover (Pham et al., 2012), bluespace (Raymond et al., 2016), woodland (Morris et al. 2011), biodiversity (Davis et al. 2012) and tranquil places (Mitchell and Norman, 2012). Overall, Watkins & Gerrish (2018) find in their review that regardless of methodology there was consistent evidence that greenspace coverage is lower in low income areas. The work on urban greenspaces is now extensive, but there remains much unknown regarding distribution of a wide range of benefits in different contexts, in particular beyond urban areas.

Through the development of scope of EJ and inclusion of other dimensions such as the vulnerability of different socio-economic groups (e.g. Moreno-Jiménez et al., 2016), of inter-generational equity and even inter-species justice (Lele et al., 2013), EJ is increasingly regarded as a key element of environmental sustainability (as demonstrated by its inclusion within the UN's Sustainable Development Goals). Moreover, environmental injustice is now better understood to manifest at multiple scales, through for example increasing concern with these complexities of transnational and global (in)justices, especially in the context of climate change (Mohai et al., 2009; Holifield, 2012).

2.2.2 Challenges in assessments of inequalities in environmental hazards and benefits

Despite a strong evidence base supporting a presence of inequality in both environmental hazards and benefits in a range of contexts, location and scales, there are inconsistencies. For example, Wen et al. (2013) examined greenspace distribution in US cities finding that more deprived neighbourhoods tend to be closer to public parks but overall are characterised by less vegetated land. Mears et al. (2019) find that access to greenspace is greater for more deprived areas in Sheffield, but quality of greenspace is lower. McLeod et al. (2000) and Pye et al. (2001) described the varying strength of association between deprivation and environmental indicators and Walker et al. (2003), Briggs et al. (2008) and Huby et al. (2009) illustrated how results changed given different scales of analysis. The lack of current agreement regarding to what extent environmental injustices exist can be attributed to the complexity of the relationship between environmental and social factors and robustness of methodologies.

Historically, studies have been criticised with regards to the quality of environmental data used, a lack of clear causal relationships, the limited understanding of actual impacts, the assumptions made in spatial analysis and the spatial scale and scope of the studies (Bowen, 2002; Maantay, 2002; Wheeler, 2004; Baden et al., 2007; Schule et al., 2017). For example, the majority of environmental justice studies have focussed upon establishing the distribution of sources of pollution and not concentration, exposure and health impact. This is due to challenges such as paucity of comprehensive data sources, such as environmental monitoring data, and adequacy of techniques for modelling exposure beyond proximity (Maantay, 2002; Buzzelli, 2007). Whilst observations of limitations have typically been made with respect to studies examining environmental hazard distribution, they are relevant to assessments of benefits, since such studies have focused upon proximity to greenspaces. However, the distribution of benefits is dependent upon different types of greenspaces in different locations since they provide different ES at different scales.

Walker (2009) more fundamentally criticised the limited scope of the spatial distribution approach, which inhibits analysis of the complex interactions between the environment and health whilst potentially concealing some inequalities. However, if understood to be one of many aspects of justice to be considered, spatial analysis of environmental inequalities, is a valuable exercise; forming the basis of further assessment, enabling analysis across large geographical areas and exploration of many variables, and

therefore can reveal much information with regards to (in)equalities. For example, Curtis (2004) highlighted that in considering disease risk factors, knowledge of proximity forms part of our understanding of exposure to environmental factors. Additionally, spatial analysis can be extended longitudinally to help infer causality i.e. whether a hazardous or beneficial land use is sited within a particular community or a community moves in to that area subsequent to change in environmental conditions (Maantay, 2000; Mohai et al., 2009; Macedo and Haddad, 2015). Longitudinal analysis is therefore desirable (Mitchell and Walker, 2007) but is highly dependent upon data availability.

Most distributional EJ analysis focus upon a single hazard or type of greenspace, thus better acknowledgement of complexities would be enabled through analysis of multiple or cumulative hazards (Krieg and Faber, 2004; Grineski et al., 2015), although some studies have achieved this (e.g. Wheeler, 2004; Pearce et al., 2010). Notably, Pearce et al. (2010) included environmental benefits alongside hazards and found that a social gradient existed with respect to multiple environmental deprivation and that this could be associated with health inequalities. However, only two environmental benefits were included in their analysis. Grineski et al. (2015) further highlighted the nuanced relationship between the distribution of hazards and benefits. For example, higher property values have been revealed in the US for properties located proximal to coasts with associated risks of flooding (Bin & Kruse, 2006; Grineski et al., 2015). In this case it is considered that exposure to the environmental hazards is a choice made in the context of a wider set of environmental benefits such as aesthetically pleasing views and surroundings.

The EJ discourse has thus become more comprehensive in its reach than its original conceptions. Its present conceptualisation usually incorporates procedural and distributional aspects, and hazards and benefits amongst many other considerations. Spatial analysis has revealed much evidence of inequalities in both hazards and benefits, but several challenges exist including the methods used, the effects of spatial scales, evidence of impacts upon health inequalities, the lack of consideration of cumulative effects and of changes over time. Further discussion regarding measuring inequalities is given in Chapter 3, section 3.3.2. The next section focusses upon how EJ is jointly considered with ES within research and policy.

2.3 Intersection of environmental justice and ecosystem services

EJ research is concerned with the environment as a source of benefit, not just an absence of hazard, which aligns with the concept of natural capital and ES supporting health and wellbeing. This therefore suggests there is a clear case for addressing natural capital/ES and EJ within a common framework. In practical terms, social inequality in natural capital/ES distribution has implications for sustainable management of natural capital with respect to the production of ES (Andersson et al., 2007; Ernstson, 2013; Bennett et al., 2015), and trade-offs between welfare and conservation objectives (Daw, 2011). The social distribution of ES is especially pertinent for impacts upon health inequalities, and feedbacks upon natural capital and its management. Furthermore, it is of growing importance to consider how justice is considered within ESs as natural capital/ES concepts are increasingly advocated as a means for informing land planning and management decisions.

Such joint consideration of ES and EJ is encompassed by the ecosystem approach advocated by the Convention on Biological Diversity, signed at the 1992 Rio Earth Summit (CBD, 2004). Consideration of fairness in all valuations of ESs was also an important concern within initial conceptions of ESs as asserted by Daly (1992), Daily (1997) and Costanza & Folke (1997). Despite these early assertions, understanding who benefits from ESs has not been widely addressed within ES literature (Daw et al., 2011; Haase et al., 2014; Bennett et al., 2015). This knowledge gap has received attention recently (Fisher et al. 2013; Barbés-Blázquez et al., 2014; Jones et al., 2016), and particularly on a conceptual basis (e.g. Ernstson, 2013; Lehmann et al., 2018; Latorra et al., 2019). Bennett et al. (2015) determine the distribution of ES socially to be one of the key challenges for future ES research and drawing on Daw et al. (2011) they highlight it as a “prerequisite for effective ecosystem services assessments” (p. 80). More specifically they note a need to address multiple concerns, familiar to EJ discourse, including the need to understand stakeholder diversity, capabilities, rights and plurality of their values, relationships to each other and within the context of governance.

With respect to spatial assessments of ESs, a need to identify winners and losers at different spatial scales and in different regions and spatial mismatches between supply and demand, in particular along rural-urban gradients, is expressed. This information is critical for understanding how policy and management of natural capital in one location effects more distant locations (Bennett et al., 2015). Ernstson (2013) proposed a

framework that seeks to do this based on social-ecological networks. This emphasises how integrated ESs and justice research should be informed by studies at different scales and from different perspectives, illustrating the importance of multi-scale work.

A lack of attention given to distributive justice concerns in ESs research may be partially attributed to more general trends in ES research such as the focus on mapping the supply of ESs at its source, and not on the beneficiaries nor the links between the areas of ES supply and demand (Serna-Chavez et al., 2014). However, more fundamentally, it may be attributed to the goal of maximising ESs for overall wellbeing of a population (no net loss or gain of natural capital/ biodiversity) and the lack of inclusion of justice in many developments of the ES framework, as exemplified by the MA. Lele et al. (2013) observe that the MA only refers to justice through the inclusion of 'freedom of choice' as a component of human health and wellbeing and there is no further reference to intra-generational justice, equity or fairness.

Despite these critiques, there are some areas of research which have focussed upon explicit linkages between ES and EJ. Most notably, inequalities and injustices of ES have been explored in the context of poorer subsistence-based global communities (e.g. MA, 2005; Sikor, 2013; Suich et al. 2015), for example, in the establishment of the Ecosystem Services for Poverty Alleviation programme (ESPA, 2018). There has also been a focus upon equity appraisal of payment for ecosystem services (PES) schemes (McDermott et al. 2013; Martin et al. 2014; Pascual et al., 2014), given that identification of winners and losers is a prerequisite of such schemes. Moreover, social equity within ES studies is commonly addressed with respect to fair representation and participation (e.g. Wilson & Howarth, 2002; Plieninger et al., 2013; Raum, 2018). As discussed above, within EJ literature there has been attention to greenspace access / coverage based on assumed benefits but these studies lack explicit use of the ES framework.

In recent years, studies have sought to address the lack of empirical research regarding the distribution of ES benefits, revealing mixed patterns. Escobedo et al. (2015) and Jenerette et al. (2011) found that regulating service supply (air pollutant removal in Bogotá, Columbia and vegetative cooling in Phoenix, USA respectively) increased with socio-economic status; conversely Escobedo & Nowak (2009) found air pollutant removal in Santiago, Chile was greater in low income areas - although as a function of ambient pollutant concentration - tree cover itself was more extensive in higher income areas. For Florida, USA, Soto et al. (2016) found forest carbon sequestration increased with income, educational attainment and proportion of non-

minority population. Hamann et al. (2016) found greater ecosystem service use in low-income areas in South Africa, emphasising the reliance of poorer communities on their natural surroundings. Whilst a national spatial analysis in China revealed forest based provisioning ESs and biodiversity are greater in poorer areas, but agricultural provisioning ESs and water availability are lower (Eigenbrod et al., 2017).

Distributional analyses have also examined how benefits are distributed across scales. For the Niger Delta, Nigeria, Adekola et al. (2015) showed that locally, in the delta states many people are directly dependent on local ecosystems services, the exception being for oil, where the benefits accrued nationally and internationally (whilst local people suffered the externality costs of oil production). Nahuelhual et al. (2019) sought to infer causality in their evaluation of the distribution of ES in Chile. Water regulation and recreation opportunities were found to be concentrated in larger properties, critical in the context of land ownership being concentrated in the hands of a few and demonstrative of the structural influence on inequalities in ES arising from historically based systems. Conversely, for Portugal, Gomes Lopes et al. (2015) showed how local people received about 45% of all benefits derived from common land ecosystems, with 40% and 15% of benefits flowing to national and global beneficiaries respectively. It is evident that given the focus upon beneficiaries, distributional analysis also necessarily examine demand for ES as well as supply. For example, a disproportionate access to cultural (heritage and recreation) services was found in Finland using analysis explicitly linking the location of ES supply and demand. Although this study by Ala-Hulkko et al. (2015) looks at difference across the whole population, it does not examine whether ES are stratified across social or demographic groups. More generally, it can be observed that an increasing emphasis upon beneficiaries and the spatial dependency of ESs within ES research more readily aligns ESs analysis with assessment of environmental inequalities.

Within the context of EJ focused research, there has been little acknowledgement of the relevance of developments in ES. Despite the definition of specific services, offering an opportunity to better understand the nuances in distributional environmental equality and more explicitly link to health and wellbeing. To date, the few analyses of inequality in ES distribution (Lakerveld et al., 2015), mean that the body of evidence remains too small and heterogeneous to draw general conclusions (e.g. as to who are the main beneficiaries under given contexts). However, the studies do commonly reveal asymmetry in distribution of ES benefits, and also point to the importance of socio-economic factors in ES provision (via management of natural capital), and raise

questions about how to assess equity in the context of ES which are so scale dependent.

The EJ and ESs discourses share key concerns, notably with respect to the importance of connections between society and the environment and how land planning management decisions impact upon human health and wellbeing. Moreover, they are predominantly anthropocentric and interdisciplinary. The spatial distribution of environmental factors, recognition of who the different stakeholders are and participation in decision making are also important features of studies in both fields. As a result of these common factors, they have also faced many similar difficulties in terms of methodology and conceptual underpinnings. Overall there is opportunity to more firmly link ESs and environmental justice. Moreover further empirical work assessing inequalities in natural capital/ES distribution is necessary, addressing a range of contexts (natural capital and ES, social factors, places and landscapes), and including high income countries which tend to be highly urbanised with people less directly dependent upon the supporting environment. It is reasonable to assume that natural capital and ES are socially distributed in these high income countries, but this remains to be tested, which is the focus of this research. Specifically, this research uses England, UK as an example of a high-income country, and thus the next section provides the country's background with respect to ES and EJ research and policy.

2.4 Natural capital, ecosystem services and environmental justice in England, UK.

The UK can be considered a country with a substantial interest in natural capital, ES and EJ. It is the first country within Europe where interest in EJ took hold, one of the first countries to complete a national ecosystem assessment and there are policy drivers for developing the knowledge base of ES distribution across socio-economic and demographic groups. Since the environment is a devolved responsibility of the separate nations within the UK, and data availability and social metrics differ between countries, it is appropriate to examine the nations independently. This section examines knowledge of ES and EJ, and the policy context for each within England, although some research discussed may be part of a wider UK study.

The UKNEA (2011) and follow-on (UKNEA, 2014) have made arguably the most notable contribution to understanding and raising the profile of ESs in the UK (Schroter et al., 2016). This national assessment concluded that the declining extent and quality of multiple habitats over the last 60 years is associated with decreases in the supply of most ESs (UKNEA, 2011). Spatially explicit representation of the distribution of ESs

was however limited in the UKNEA (2011), although carried out for case-studies in the follow-on reports. In the wider literature, ES mapping studies in England have been produced for a range of scales.

Nationally, ES assessments may be focused upon a singular ES and knowledge regarding spatial distributions of single services can highlight particular issues (e.g. Breeze et al., 2011). However, inclusion of a range of services is necessary for fully informed decision making (Martinez-Harms and Balvanera, 2012). Spatial models of multiple ES, and in some studies their change in response historically or to future land cover scenarios, have been effective in highlighting trade-offs between different ESs (Anderson et al., 2009; Eigenbrod et al., 2011; Haines-Young, 2011; Holland et al., 2011; Christie and Rayment, 2012; Firbank et al., 2013; Maskell et al., 2013;). In particular these point to potential conflict between agricultural productivity and other ESs (Eigenbrod et al., 2011; Firbank et al., 2013). Whilst these studies begin to address where the winners and losers of the trade-offs are located, they do not provide insight regarding who the winners and losers are.

Regional and local case studies enable examination of ESs which are most relevant for a particular area or habitat. In England, case studies have been undertaken at the county level in Cornwall (Caselegno et al., 2014), at the river catchment level in Dorset (Newton et al., 2012), for upland areas (Grand-CleMET et al., 2013), lowland areas (Posthumus et al., 2010; Cerqueira et al., 2015) for wetlands (Acreman et al., 2011; McInnes, 2013) and for urban areas (Hölzinger et al., 2014; Speak et al., 2015). Primarily these generate locally relevant knowledge but more widely demonstrate the complexities of ES spatial modelling and the need for prioritisation of some services as they ascertain win-win scenarios are unlikely. Local case studies tend to acknowledge some aspects of procedural and participatory EJ since they facilitate increased stakeholder involvement and typically give greater consideration of the socially constructed values placed on local ESs (Dick et al., 2014).

Research in the UK echoes international recognition of the growing importance of urban ESs. The provision of areas which may provide ESs within the city have been shown to be decreasing (Perry and Nawaz, 2008) and the total value of ESs provided within UK cities is low comparative to other European cities (Larondelle et al., 2014). Urban based studies have focussed on specific ES including flood protection, space for recreation, and climate regulation (Whitford et al., 2001; Speirs, 2003; Donovan et al., 2005; Davies et al., 2011; Hall et al., 2012; DEFRA, 2013). The spatial distributions of multiple ESs are less commonly assessed, although there are some exceptions; Tratolos et al. (2007) determined how different urban forms can influence ecosystem

service provision regardless of housing density. Radford & James' (2013) assessment along an urban-rural transect shows that supply of most ESs increase with decreasing urbanisation, but recreation and pollination exhibit an opposing trend. Although this study only addressed one transect at a single point in time. Hölzinger et al. (2014), in their assessment for the city of Birmingham, contributed to the limited research base which models spatial overlaps between supply of and demand for several services, but do not account for their flows between different areas.

With regards to environmental inequity multiple UK studies have determined a positive relationship between indicators of lower socioeconomic status and several aspects of poorer quality physical environments (see review by Lucas et al., 2004), but this is not apparent for all types of environmental hazard. The most prolific body of work for England has found inequalities with respect to the distribution of air pollutant concentration (Stevenson et al, 1998; Pye et al., 2001; Brainard et al., 2002; Mitchell and Dorling, 2003; Jephcote and Chan, 2012). Although recently, Tonne et al. (2018) revealed low income groups were burdened with higher pollution at their place of residence whilst longer commutes leads to greater individual exposure for high income groups. Inequalities have also been demonstrated with respect to landfill sites (Walker et al, 2005b; Damery et al., 2007), coastal flood risk (Fielding and Burningham, 2005; Walker et al., 2006), IPC sites (Friends of the Earth, 2004; Walker et al., 2005b), noise (Brainard et al., 2004), environmental intrusion (Mitchell & Norman, 2012) water quality (Damery et al., 2008) and a combination of such factors (Walker et al., 2003; Wheeler, 2004; Pearce et al., 2010). However, Walker et al. (2003, 2006) found fluvial flooding was not consistently associated with higher deprivation, leading Wheeler (2004) to conclude that environmental inequalities are dependent upon the measure considered. Numerous studies have sought to better understand the drivers of inequality (Van der Horst and Toke, 2010; McDonald et al., 2010; Cotton and Devine-Wright, 2013). In this respect evaluations of changing patterns of distribution can contribute valuable insight, thus Mitchell and Norman (2012) established that increases in environmental intrusion 1960-2007 were greater in more deprived areas, whilst Mitchell et al. (2015) found air quality improvements 2001-2011 were least for more deprived areas. However, examples of longitudinal analysis are rare given data limitations.

Attention on the social distribution of environmental benefits has focussed on accessible greenspace (CABE, 2010; Natural England, 2015), urban green infrastructure (Ferguson et al., 2018) and more specifically for parks (Barbosa et al. 2007) and woodland (O'Brien & Morris, 2014). Overall it is apparent that that ethnic minorities and people of lower socio-economic status visit greenspace less often, however unequal distributions of greenspaces are not always apparent. Ferguson et al.

(2018) found that in Bradford street tree density was highest in areas with a greater proportion of ethnic minority residents and those with lower socio economic status, but an inverse pattern was found with respect to public greenspace. Morse et al. (2011) utilised several indicators of countryside quality for a national assessment and found lower quality is significantly associated with higher deprivation, although only a weak correlation was observed. Church et al. (2014), in a rare example of specifically utilising an ES frame for assessing the social distribution of environmental benefits, found that for cultural ESs there is some evidence that the least deprived have better access to natural culturally important landscapes, but evidence was inconclusive. In summary, evidence for an unequal social distribution of greenspace in England exists but it is equivocal. Furthermore, it is not possible to draw conclusions on the distribution of a fuller range of natural capital in a range of contexts, or the multiple services they provide. Thus, for the UK there is a general lack of knowledge on the social distribution of natural capital and particularly the ecosystem services that flow from it.

Inequalities in the distribution of ES are relevant to a range of UK policies and guidance. UK Government inquiries report environmental inequalities as a material factor in explaining UK health inequality (Acheson, 1998; Marmot, 2010). In the UK, the health 'gap' has continually widened and lower socioeconomic groups currently experience disproportionately higher mortality rates from heart and respiratory diseases and lung cancer and are more likely to die prematurely from cardiovascular disease. There are also inequalities reported for mental health and childhood obesity (PHE, 2018). Past assumptions that overall improvements to social determinants of health would reduce inequalities (Acheson, 1998) have proven to be invalid (Exworthy et al., 2003), hence there is now support for a 'proportionate universalism' approach, whereby the action to improve public health should be proportionate to the requirements of the vulnerability of population and the inequalities present (CSDH, 2008). Commissioned by the Department of Health, the 2010 Marmot review advised that reducing environmental inequalities is necessary for reducing these health inequalities (Marmot, 2010).

Addressing environmental inequalities was introduced as a key element in the UK Government Sustainable Development Strategy (HMG, 2005). Principles of environmental justice were also embedded within the Government's Department for Environment and Rural Affairs' 'The Third Sector Strategy' for England and Wales (DEFRA, 2008) and the Environment Agency's social policy (EA, 2005). HM Treasury's Green Book guides best practice in assessment of equality of environmental and health impacts in the policy, project and programme appraisal process (HM Treasury, 2018).

Building on requirements for the Equality Act (2010) (which does not include consideration of equity with regards to socioeconomic status), The Department for Transport and the National Institute for Health and Care Excellence include guidance on appraisal of impacts on environmental inequality. Public Health England (2014) also aims to improve health equity, but this was largely reasoned through associated increases in physical activity and a wider range of ESs were not considered.

With respect to natural capital and ES, policies relating to the management of the environment in England have increasingly looked to use the concept of natural capital and ES frameworks. In 2011, the Natural Environment White Paper (DEFRA, 2011) established the Natural Capital Committee (NCC) whose 'State of Natural Capital' reports have proposed frameworks for integrating natural capital within accounting (NCC, 2013; 2015). Consequently, the Office for National Statistics (ONS) is currently integrating natural capital within its environmental accounting (ONS, 2015a). Most recently the 25 Year Environment Plan (25YEP) for England is built upon the concept of natural capital (HMG, 2018), with an aim to have 'Net Gain' in natural capital. The 25YEP also gives some explicit consideration to environmental inequalities stating "we want to ensure an equal distribution of environmental benefits, resources and opportunities", and specifically addressing the need to connect people from minority ethnic and low-income groups to nature and the sustainable management of land. However, in line with research of inequalities in environmental benefits it remains largely focussed upon cultural ES and people's time spent in nature, with little attention given to the potential inequalities in multiple ES.

The UK can thus be characterised as a country with substantial interest in EJ, but with analyses that neglect natural capital and ecosystem services, and conversely substantial interest in natural capital and the benefits to people from ecosystem services, but with little consideration of how those benefits are socially distributed.

2.5 Research aims and objectives

The ES and EJ concepts both examine aspects of the human-nature relationship and are ultimately concerned with impacts on human health and wellbeing; established in the latter half of the 20th Century they have rapidly developed to comprise a large literature base. Despite significant progress, there remains challenges in both fields, with several challenges common to both. This includes for example, debate surrounding their theoretical basis, the balance between biophysical and social science

analysis approaches, model oversimplification, data availability, sensitivity to spatial scales, uncertainties of results, and a lack of temporal analysis.

It is also evident that within the ES discourse there is a lack of empirical analysis establishing the distributions of ES across different socio-economic and demographic groups, most notably in high income countries where linkages between people and their natural environment are less direct. This neglects the importance of equitable distribution conveyed within early developments of the ES concept. Concurrently, distributional concerns within the EJ discourse have traditionally focused upon differential exposure to environmental hazards, although there is now significant attention awarded to the importance of inequalities in environmental benefits. Yet analysis tends to be based upon proximity to various types of greenspace, normally within an urban context and with inconsistent conclusions. Few spatial assessments have recognised the distribution of a range of specific benefits, and their spatial complexities i.e. that benefits may be generated by natural capital both nearby and distant, and by different types of natural capital, as depicted by the natural capital/ES frameworks. Ultimately, this overlooks opportunities for the distributional aspect of justice to be better incorporated within spatial planning and environmental management processes.

This is the focus of this research, which aims to provide insight into the following question;

Is there inequality in the social distribution of ecosystem services in England?

A spatial analysis is used to generate new knowledge to address this question. Spatial analysis is fundamental to both assessment of ES and of distributional EJ assessments. England provides an example of a high-income country which at policy level is adopting a natural capital/ES approach to natural environment management, with awareness of the need to address environmental inequalities in the context of notable health inequalities, yet a lack of comprehensive understanding of inequalities in environmental benefits. Based on existing studies of urban greenspace distribution, it is hypothesised that some inequalities will be observed.

The analysis seeks to achieve the following objectives;

Objective 1: To assess the social distribution of natural capital across England.

Objective 2: To assess the social distribution of multiple ecosystem services for case study regions in England.

Objective 3: To evaluate the robustness of results to ecosystem service model assumptions and uncertainties.

Fulfilling these objectives comprises a multi-scale approach providing insight from national and regional analysis, which is to the author's knowledge unique in the context of the research aim. A multi-scale approach aligns with recommendations within both EJ and ES literature, although to achieve the national insight, it is necessary to focus solely on natural capital as ES are overly complex to assess nationally (ES are addressed via regional case studies). Moreover, insight nationally and regionally will go beyond urban areas, which have been the focus of greenspace inequalities. This is important since there are indications that some ES (e.g. recreation) reduce along urban to rural transects. Multiple ES are sought to be assessed for the second objective since there may be trade-offs between some ES, particularly at different scales. Assessment of the robustness of results is critical given the critique that many ES assessments do not account for uncertainties, whilst environmental inequality has been shown to be sensitive to measurement approach.

In carrying out these objectives, answers to these more specific questions are sought;

- *Which ES are appropriate to be assessed from the perspective of equitable management of natural capital in England?*
- *How can the different scales at which ESs are delivered from natural capital to beneficiaries be accounted for?*
- *Are inequalities in natural capital and ES present at different scales?*
- *Are inequalities consistent across different ES?*
- *Is there a rural-urban gradient in inequalities?*
- *Are findings robust to uncertainties and model assumptions?*
- *Can opportunities for synergistic social and ecological outcomes be identified?*
- *Are there opportunities for closer integration of distributive justice concerns and ES assessments within land planning policy and management?*

This chapter has introduced the concepts of ES and EJ in greater detail, reviewing their historic basis, current research challenges and how the two discourses have been examined jointly within the current literature. Specifically, knowledge of these two fields

and their relevance to policy in England has been reviewed to provide insight into the country level context relevant for this research. The main research question and accompanying aims have been presented, which seek to address some of the weaknesses within the existing literature base.

The next chapter (3), sets out the theoretical frameworks of EJ and ES and how these may be related, it provides a review of methodologies and the approach that is taken to accomplish the research objectives.

Chapter 3 Research design

Chapter 2 discussed the concepts of natural capital, ecosystem services (ESs), and distributional justice, reviewing the historical basis, current knowledge and relevance of these discourses. This set out the context and significance of this research. The focus of Chapter 3 is how the natural capital and ES concepts are operationalised for analysis; this entails presenting the research design and methodological approach.

There is abundant literature which addresses the conceptualisation, classification and quantification of ESs and natural capital. Critiques of ES research often refer to confusion caused by the multiplicity of terms, and varied understanding of ESs and methods (Boyd and Banzhaf, 2007; Wallace, 2008; Fisher et al., 2009; Seppelt et al., 2011; La Notte et al., 2017). However, as Crossman et al. (2013) argue, much of the ambiguity of findings may be avoided by clarification at the outset of the approach taken and terminology adopted. This chapter therefore appraises the range of approaches within ES research and clarifies the conceptual basis, adopted terminology and modelling approach taken in this research, thus clearly defining the scope of the research.

The chapter is organised as follows: The research framework is established in the first section (3.1). This includes the evaluation of some of the most common conceptualisations of natural capital and ES leading to the presentation of the conceptual foundation of this research in section 3.1.2. This also includes clarification of the terminology adopted. Given the research aim, the analysis is inherently spatial hence the overall research design, which takes a spatial analysis approach, is presented in section 3.1.3.

Subsequently, more detailed accounts are provided regarding the selection of natural capital and ES for analysis (section 3.2), potential methods applied for mapping these (section 3.3.1) and associated challenges and limitations. The discussion of methods for mapping natural capital and ES presented in this chapter is an overview account of the approaches taken in the study. Bespoke methods, specific to natural capital overall and to individual ESs, are related in their corresponding chapters which follow (Chapters 4-7). The final section (3.3.2) specifies methods for assessing inequalities. These methods are common for all natural capital and ES studied.

3.1 Conceptual framework

3.1.1. Review of ecosystem services and natural capital frameworks

As defined in the introductory chapter (Chapter 1, section 1.1), natural capital is “the stock of renewable and non-renewable resources (e.g. plants, animals, air, water, soils, minerals) that combine to yield a flow of benefits to people” (Natural Capital Coalition, n.d.). Whilst ESs are commonly defined as “the benefits people obtain from ecosystems” (MA, 2005). How these relate, are produced and become contributions to human health and wellbeing are established by conceptual models. As a highly multi-disciplinary subject, there are multiple perspectives incorporated within the natural capital, and ES literature (La Notte, 2017) and there is no consensus on an optimal conceptual model. The diversity of models (e.g. Daily, 1997; Costanza, 1997; MA, 2005; Boyd and Banzhaf, 2007; Fisher et al., 2009; TEEB, 2010; Haines-Young & Potschin, 2013; see figures 3.1-3.3) has created inconsistencies which limit broader conclusions being drawn from across different studies (La Notte et al., 2017). La Notte et al. (2017) further argue that a more consistent approach would ensure rigor in ES analysis. However as simplifications of reality, limitations of conceptual models are unavoidable (Boyd and Banzhaf 2007; Fisher and Turner 2008; Wallace 2008; Potschin and Haines-Young, 2011; Gomez-Baggethun and Barton, 2013) and the relative advantages and disadvantages of different models should be considered. Ultimately, the multiplicity of conceptual models allows flexibility meaning that the chosen ES model can be suited to the purpose of the research.

Whilst natural capital is often understood to be the foundation of ESs, ES conceptual models may not explicitly reference natural capital and natural capital conceptual models may not explicitly reference ESs. Generally, less attention in the scholarly literature has been awarded to explicit framing of natural capital than for ES. However, it has been addressed more widely in policy and practice, in particular as it is promoted as a concept to be utilised within business (e.g. Bonner et al., 2012; NCC, 2015; Petersen et al., 2015). ‘Natural capital benefits’ is one example of an alternative term to ES used in natural capital models (e.g. Mace et al., 2015; NCC, 2015), whilst the term ES isn’t used it’s evident that the concepts are consistent. The use of different terms may be considered more appropriate given that natural capital assets may also include abiotic (non-biological) based assets such as geology and wind. Confusingly some reports include ESs alongside natural assets under natural capital as an umbrella term (e.g. Bonner et al., 2012). This conflicts with the definition of natural capital as ‘physical

natural assets' not services and could lead to double counting when valuing that natural capital.

Others convey the concept of natural capital through an economics lens; as the foundation upon which the economy and key components of the socio-ecological system are built upon. This has been demonstrated by Herman Daly's 'Triangle' (Daly, 1973 and subsequently adapted by Meadows, 1998; Figure 3.1). It is a broader view of natural capital which doesn't specify benefits as ESs and more closely ties in with the common usage of 'capital' as an economic term. The simplicity of the model emphasises the reliance of humans upon natural capital but does not consider the explicit pathways between natural capital and the benefits to human wellbeing. More detailed models which examine these pathways illustrate that benefits do not simply arise from natural capital but are co-produced by it together with other forms of capital e.g. human capital, financial capital (Palomo et al., 2016; Costanza et al., 2017). These are considered more representative conceptualisations of natural capital since they reflect the connectivity of socio-ecological systems.

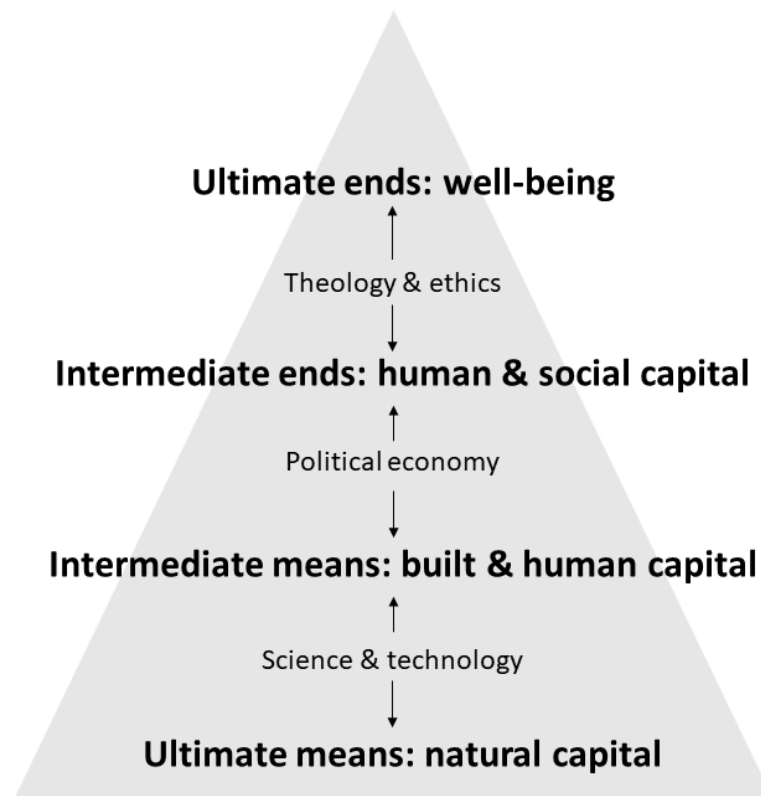


Figure 3.1. Modified 'Daly's triangle' from Meadows (1998). Natural capital is the foundation for human health and wellbeing.

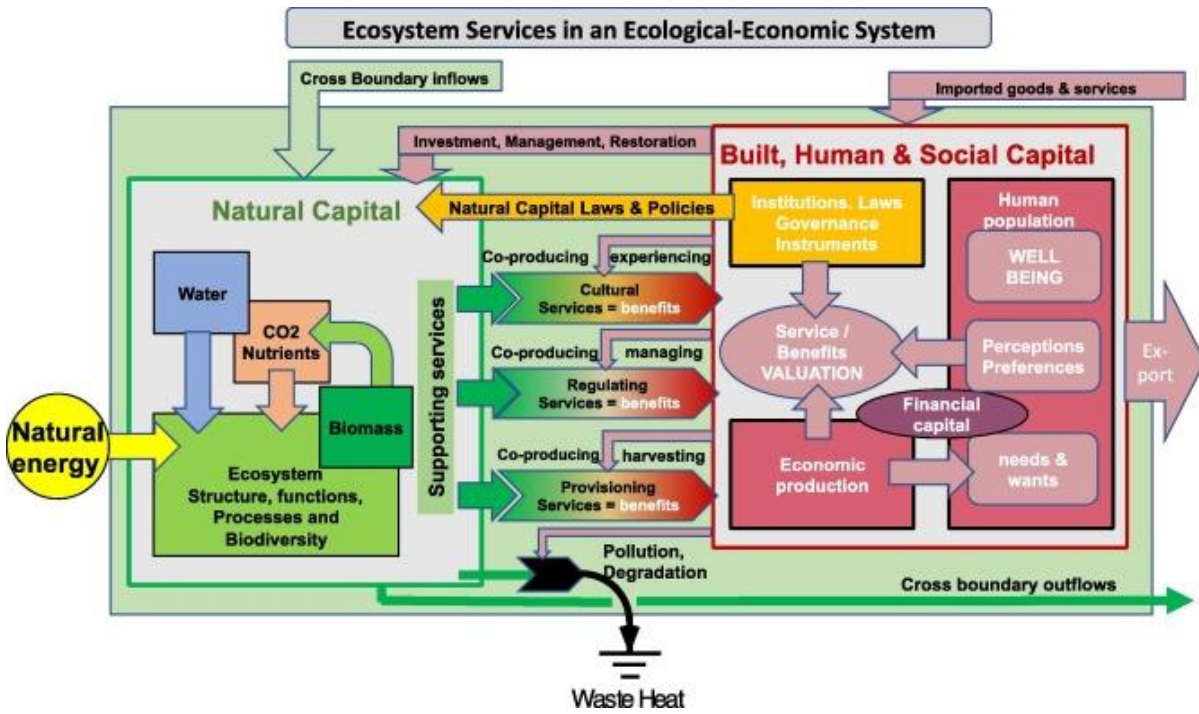


Figure 3.2. Costanza et al.'s (2017) representation of the complex linkages between natural capital, ES and social and built capital with human health and wellbeing. Interactions are conceptualised as a series of energy flows. Benefits from ES are co-produced by ecological and social inputs and interactions.

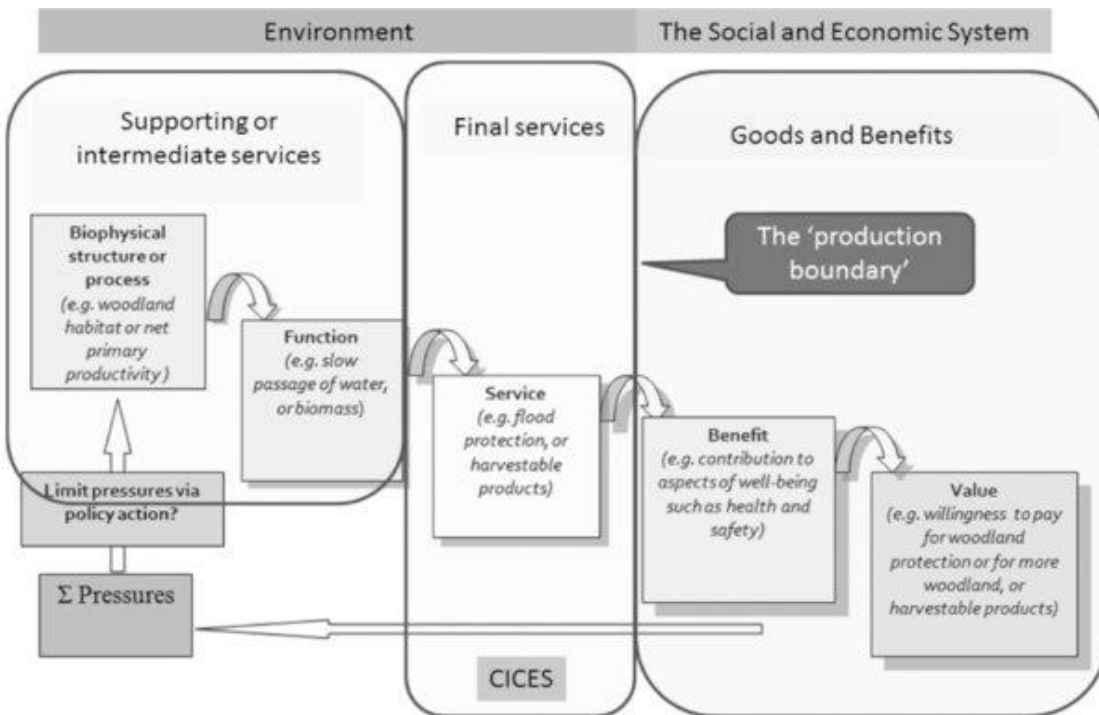


Figure 3.3 ES cascade model (Potschin-Young et al., 2018; adapted from Potschin and Haines-Young, 2011).

Overall, despite some differences in the complexity of conceptual models, natural capital is consistently considered a critical component of the socio-ecological system upon which human wellbeing relies. In this study, the focus is upon natural capital as assets from which ESs are generated, whilst for benefits to be generated other forms of input are required. It is also acknowledged that there are wider benefits beyond this (e.g. energy created from wind power) but these aspects are beyond the focus of this research.

As observed above, conceptual models of ESs may or may not explicitly refer to natural capital. The accelerated growth in ES research has brought about many conceptualisations and classifications of ES (e.g. Boyd and Banzhaf, 2007; Wallace, 2007; Fisher et al., 2009; Haines-Young and Potschin, 2012; Landers and Nahlik, 2013). Several of the most common ES classifications (i.e. lists of individual ES such as those given by TEEB and CICES) are based on Haines & Potschin's (2011) cascade framework (La Notte et al., 2017). This depicts a linear 'production' chain where ecosystem structures and processes generate goods and services which are converted into benefits for humans and can be valued if desired (Figure 3.3). This conceptualisation is useful for showing different stages in the production of benefits from ESs and for depicting the reliance of humans upon ecosystems (La Notte et al., 2017). A key criticism of the cascade is that it largely negates the socio-economic feedbacks important for the generation of ESs. Adaptations which better incorporate these feedbacks have been proposed by De Groot et al. (2010) for TEEB and Spangenberg et al. (2014). Nevertheless, it is still argued that the cascade neglects the co-production of ESs by socio-economic and bio-physical processes (Villa et al., 2014). Further concerns with the model is that it implies all ESs generated are converted into benefits for humans, not accounting for the depletion of goods and services before they reach the beneficiaries (Villa et al., 2014). Whilst Costanza et al. (2017) point out that it actually overcomplicates the concept of ES. The cascade separately defines ES and benefits but Costanza et al.'s (2017) stance is that there should be no distinction since ES are defined as the benefits obtained from ecosystem functions and processes.

Alternative models aim to convey the non-linear nature of the socio-ecological system, providing greater detail regarding the components of the social system which are important for the generation and moderation of ESs and the complexity of interactions (e.g. Felipe-Luci et al., 2015; Costanza et al., 2017). This includes for example, the importance of the preferences and desires of the human population in addition to the more formal roles of governance, economic production, and management in the co-production of ESs (Figure 3.2; Costanza et al., 2017). Ernston (2013) takes this a step

further by proposing a framework based on a socio-ecological network. For city-wide and larger analysis, the network comprises of nodes representing the natural capital and its management (its 'protective capacity') whilst the network connections between nodes represent the spatial flows of socio-ecological processes. The framework aims to demonstrate the social production of ESs whilst emphasising the spatial links between locations and how changes in one area affect the wider production of ESs.

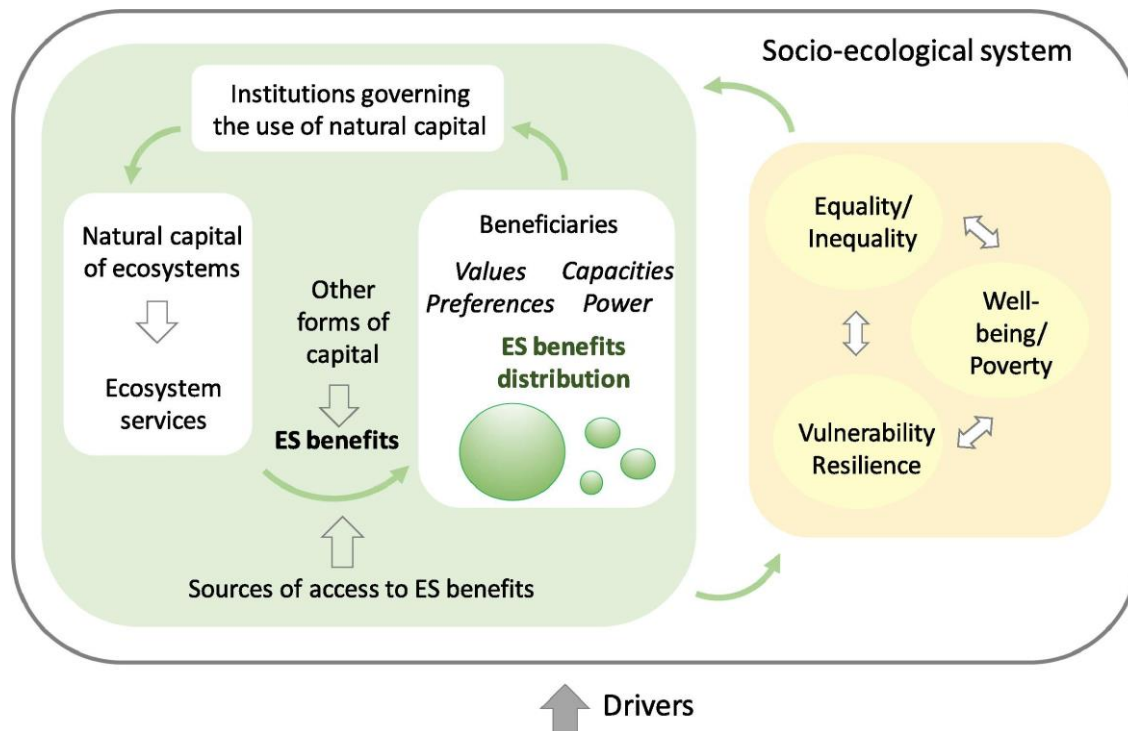


Figure 3.4 Laterra et al's (2019) ES framework; an ES framework (left) which recognises the distribution of benefits and its connection with a socio-ecological system (right) which emphasises the impact of benefit distribution.

Until recently these frameworks placed greater emphasis upon role of social processes within the production of ES, giving no explicit reference to the importance of a fair or equal distribution in ES. Although this may be implicitly assumed with consideration of the perceptions and needs of people, governance, policies and the management of natural capital. The frameworks tend to convey benefits to humans as a whole, perhaps encouraging consideration of 'net benefits'. Within the ES discourse there has thus been growing recognition of the need to better identify ES beneficiaries (Bagstad et al., 2013; Villa et al., 2014) and who these beneficiaries are (Bennett et al., 2015). Despite this, until very recently (Mullin et al., 2018; Laterra et al., 2019) the social

distribution of ES benefits was an omission within the ES frameworks. However, in their 2019 paper reviewing inequalities in ES in Latin America,

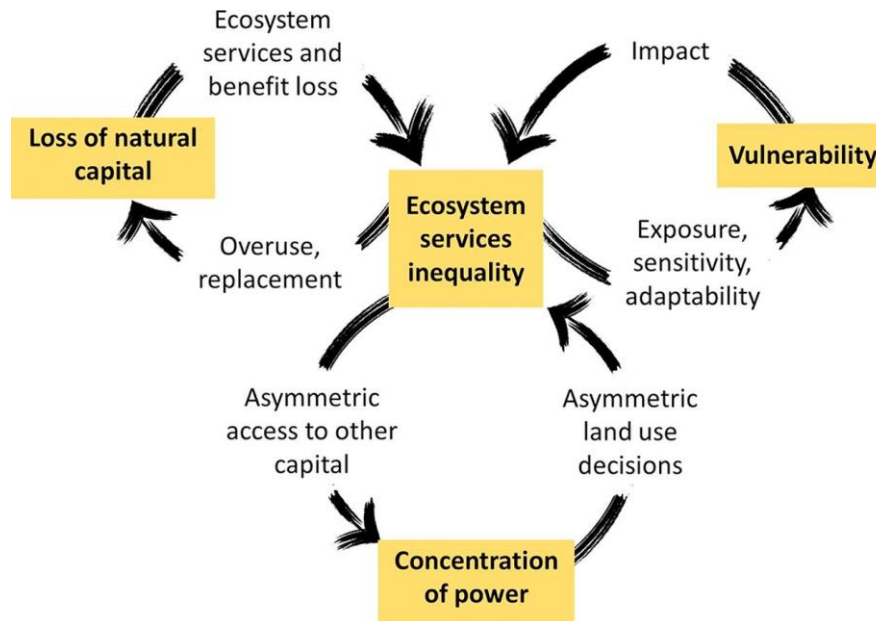


Figure 3.5 Laterra et al.'s (2019) “ecosystem services inequality trap”, which emphasises the socio-ecological feedbacks from inequality in ES, which reinforce such inequalities.

Laterra et al. present an ES framework which emphasises the need to understand the social distribution of ES (Laterra et al., 2019; Figure 3.4). In addition they demonstrate the impact of an unequal distribution as three cycles which it perpetuates and which ultimately lead to increasing inequalities (Figure 3.5). These includes negative impacts upon natural capital, upon social vulnerabilities and upon access to decision making (thus linking with participatory and procedural justice) (Laterra et al., 2019). Explicit recognition of the socially disaggregated benefits of ESs can be considered a first step in addressing these feedbacks and so to adopt an equitable approach to the management of natural capital.

3.1.2 Incorporating benefit distribution within an ecosystem service framework

Figure 3.6 illustrates a conceptual framework which emphasises the importance of disaggregating benefits from natural capital and ESs by social factors. Through quantifying ESs in a disaggregated manner, the (in)equality in their distribution can be determined. This represents the conceptual basis for this research. Whilst it simplifies

some of the components of the socio-ecological system as presented by Costanza et al. (2017) its purpose is to convey how inequality assessments align with the general natural capital and ES frameworks. The framework understands humans as both co-producers and beneficiaries of ES. Quantification of ES's net contribution to wellbeing may be important to consider in some contexts but quantifying how it differs for different social groups enables land use planning and management of ecosystems in a way which proactively addresses issues of equity.

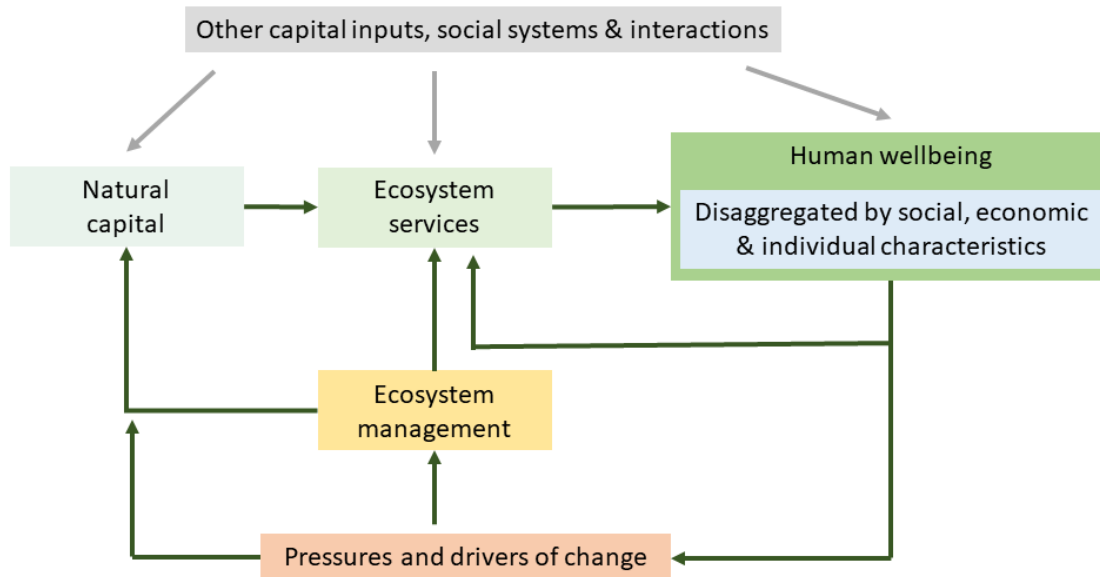


Figure 3.6. Conceptual framework integrating social distribution analysis within ES and natural capital concepts.

Critical to assessing social distributions, analysis needs to be spatially explicit and the next section (3.1.3) addresses the importance of this within modelling and analysis of ESs. It further addresses how this leads to the structure of the research as a multiscale analysis.

3.1.3 Spatial scale as a concept central to research design

There are multiple reasons why spatial scale is integral within ES and EJ research and, as highlighted in Chapter 1, the approach in this research is based upon spatial analysis. This section explains in greater detail how ES models are conceptualised spatially and hence how spatial scale is fundamental to the design and structure of this study.

The conceptual models explained in the previous section depict the connections and processes between social, economic and ecological processes. However, they do not

depict the spatial dependency of these processes which is crucial for assessing the distribution of an ES and thus considering the equity of that distribution (TEEB, 2010; Syrbe and Walz, 2012; Ernstson, 2013,). Villamagna et al. (2013), Bagstad et al. (2013) and Villa et al. (2014) present frameworks which focus upon spatially explicit non-linear pathways from natural capital as the source of ESs to the beneficiaries of these ESs.

Derived from these models, Figure 3.7 illustrates the spatially explicit conceptualisation of ESs which is used to map ESs in this research. Since the concepts and terminology used to convey the spatially explicit components of ES models, including ES supply, demand and flows are not used in a consistent manner (Schröter, 2014), the definitions applied are also clarified in Table 3.1. Figure 3.7 illustrates the spatially distinct 'service providing areas' or SPAs (where ES supply is generated) and 'service benefitting areas' or SBAs (where the ESs are used). ES demand reflects the needs or desires of people located within the SBAs. ES flows are the spatial and temporal connections between the SPAs and the SBAs (Bagstad et al., 2013; Villamagna et al., 2013). These may be driven by direct or indirect social, economic or ecological processes. Bagstad et al. (2013) emphasise the non-linearity of ES flows demonstrating that the ES initially supplied may be depleted prior to reaching some beneficiaries through ES sinks or rivalry between beneficiaries. Additionally, Villamagna et al. (2013) emphasise that the need or desire for a particular ES may exceed that which is supplied, or that the reverse may be true. This is important when we consider how beneficial an ES may be, i.e. if there is no demand for a service in a particular area then is the ES supplied there beneficial? ES supply relative to ES demand is conveyed by the term 'net ES'.

Table 3.1. Definitions of terms commonly used in ESs literature * *Italic font denotes alternative definition not adopted in this study*

Term	Definition*
Ecosystem goods and services	"The benefits people obtain from ecosystems" (MA, 2005). "The direct and indirect contributions of ecosystems to human well-being" (TEEB, 2010). "Ecosystem services are the specific results of those processes that either directly sustain or enhance human life (as does natural protection from the sun's harmful ultraviolet [UVI rays) or maintain the quality of ecosystem goods (as water purification maintains the quality of streamflow)" (Brown et al., 2007).
Ecosystem processes	Changes or reactions occurring in ecosystems; either physical, chemical or biological (MA, 2005). Operations, reactions and interactions between and within different abiotic and biotic elements of ecosystems. Regardless of whether humans benefit (Costanza et al., 2017). Quantified in terms of rates. (Wallace, 2007).
Ecosystem structures	Biophysical architecture of ecosystems; species composition making up the architecture may vary (TEEB, 2010).
Ecosystem functions	"A subset of the interactions between ecosystem structure and processes that underpin the capacity of an ecosystem to provide goods and services. The building blocks of ecosystem functions are the interactions between structure and processes, which may be physical (e.g. infiltration of water, sediment movement), chemical (e.g. reduction, oxidation) or biological (e.g. photosynthesis and denitrification)" (TEEB, 2010) OR <i>Synonymous with ecosystem processes (Wallace, 2007; Brown et al., 2007).</i>
Ecosystem capacity	"Long-term potential of ecosystems to provide services appreciated by humans in a sustainable way" (Schröter et al., 2014).
Intermediate ecosystem services	Biological, chemical, and physical interactions between ecosystem components lead to final ecosystem services, these are not end-products (Boyd and Banzhaf, 2007).
Final ecosystem services	Direct contributions to human well-being (Fisher et al., 2009).
Ecosystem service supply	Refers to the capacity of a particular area to provide a specific bundle of ecosystem goods and services within a given time period (Burkhard et al., 2012). Level of supply is dependent on different sets of landscape properties that influence the level of service supply (Willemen et al., 2012) and the ecosystem structures and processes (Bastian et al., 2013) Potential provision of ecosystem benefits, irrespective of whether humans actually use or value the function at that point in time (Tallis et al., 2011; Villa et al., 2014).

(continued)

Term	Definition
Ecosystem service provisioning areas (SPA)	Commensurate with ecosystem service supply. Source or supply regions (Villa et al., 2014).
Ecosystem service benefitting areas (SBA)	The area which can take advantage of an ecosystem service. Ecosystem service benefitting areas may be spatially disparate to the providing areas. (Syrbe and Walz, 2012). Commensurate with ecosystem service demand. Use or demand regions (Villa et al., 2014).
Ecosystem service providing units	Spatial units that are the source of ecosystem service (Syrbe and Walz, 2012). Includes the total collection of organisms and their traits required to deliver a given ecosystem service at the level needed by service beneficiaries (Vandewalle et al. 2009).
Ecosystem service trade-offs	The way in which one ecosystem service responds to a change in another ecosystem service (Millennium Ecosystem Assessment, 2005).
Ecosystem benefits	"Ecosystem service benefit as the outcome of the set of processes that join a beneficiary group with specified source ecosystem(s) through a clearly identified spatio-temporal flow" (Villa et al., 2014).
Ecosystem service flows	How an ecosystem service supply is transported to its beneficiaries, defined in space and time by physical or information processes. (Villa et al., 2014). "the transmission of a service from ecosystems to people." (Bagstad et al., 2013) OR " <i>The actual use of ES...a conceptual ideas that focuses on a point in time and space of the last contribution of the ecosystem to human well-being</i> " (Schröter et al., 2014).
Sink regions	Areas where the ecosystem service is reduced partially or completely prior to reaching the beneficiaries. (Villa et al., 2014).
Rival ecosystem service	A user or beneficiary will reduce the amount of ecosystem service benefit available to other users or beneficiaries (Villa et al., 2014). Note difference with 'exclusivity' whereby use of an ecosystem service excludes the use by another potential beneficiary, provisioning services are exclusive (Brown, 2007).
Non-rival ecosystem service	The ecosystem service benefit received by one user does not 'appreciably' impact upon the benefit received by another. (Villa et al., 2014)

Figure 3.7 also demonstrates how ES flows operate at different scales, depending on the specific physical and social processes involved, meaning that SPAs and SBAs are specific to an ES (Hein et al., 2006; Crossman et al., 2013; Larondelle & Lauf, 2016). For example, an area of woodland defined as a SPA, may produce several benefits to different areas (the SBAs). One benefit may be the improved air quality which directly benefits people present in the local area. At the same time the woodland may be a desirable place to visit for cultural reasons and the SBA in this case is potentially larger and/or more distant. At the largest scale, the woodland can provide global benefits by sequestering carbon and thereby contributing to climate regulation. Additionally, unique to each ES, there is a minimum supply unit for ES service supply. For example a single tree may have a cooling effect but the larger spatial unit of a woodland is required to provide recreational opportunity (Andersson et al., 2015; Raudsepp-Hearne & Peterson, 2016). Following this, a spatially explicit approach to mapping ES, including accounting for different SBAs and SPAs, is taken throughout the research design, in particular with respect to the selection of ES (section 3.2.2) and to the methodology applied (section 3.3.1.2).

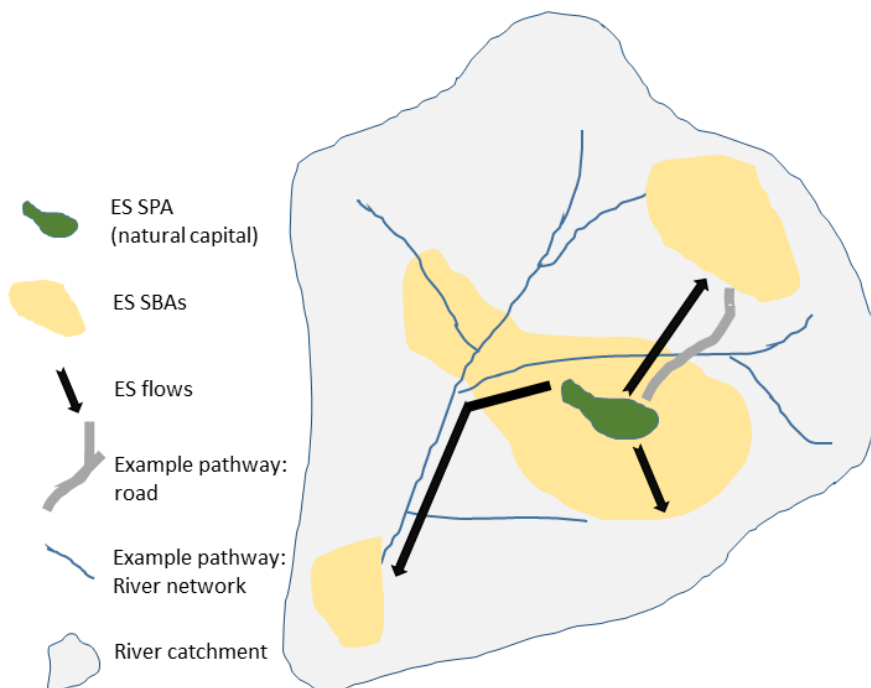


Figure 3.7 Spatial connections between natural capital which supplies ES (SPAs) and the location of the beneficiaries (SBAs). The ES flows may occur via natural or anthropogenic means (e.g. river or road). They may operate at different scales and SBAs may or may not be spatially discrete from SPAs.

Similarly, spatially explicit processes and scale of analysis are important considerations in EJ distributional analysis. Identification of the most pertinent scale for analysing a particular environmental hazard is important (Birkmann, 2007). However data availability may be a constraint to assessment at the desired scale and scales of social and environmental data and processes do not necessarily align (this is also a constraint for mapping ES). This is dependent on the environmental and social metrics selected. For example some social data such as household income is available at household level (e.g. 'Understanding Society Longitudinal Study'; University of Essex, 2018), others such as Census data are aggregated to small areas (although these may be then assigned to household locations). Available data on environmental hazards is often modelled across coarser resolutions (e.g. 1km background air pollutant concentration, DEFRA) or estimated based on proximity to a hazardous sites (e.g. landfill sites). Thus a key challenge for EJ distributional analysis is deciding on an appropriate scale to use for comparing social and environmental data. This should also consider the sensitivity of the magnitude of (in)equality to the scale used (Baden et al., 2007); a well-recognised methodological complexity inherent to distributional analysis (Baden et al., 2007; discussed further in section 3.3.2.2).

To account for the potential impact of scale upon results, assessment at multiple scales can be undertaken. A single scale analysis uncovers patterns typical of only the selected scale (Wilbanks, 2006). Whilst multi-scale studies can lead to a more in-depth understanding of phenomena because socio-ecological processes, pressures and structures operate at multiple, nested scales and often transcend these. Moreover, the size of the study area is usually inversely related to the scale of analysis; analysis across larger areas usually involves generalisation of information. This necessary compromise is due to the higher complexity, data and time requirements of finer scale studies which are not feasible for much larger areas (Norton et al., 2016). Multi-scale analysis combines the benefits of both large area and detailed (but small area) analysis.

The overarching aim of this study is to better understand how benefits from the environment are socially distributed in England. Recognising the advantages of a multiscale analysis this research is structured so that analysis with national coverage, which is necessarily coarse, is complemented by more detailed case study analysis (Figure 3.8). A national analysis provides a full picture, but reliably mapping ES nationally whilst accounting for the complexities in ES supply, demand and flows and the data required to achieve this is not viable (Norton et al., 2016). Therefore the national analysis is coarser and assesses the co-occurrence of natural capital and deprivation in English districts (objective 1). This is a descriptive approach addressing

the social and environmental conditions that the conceptual framework (Figure 3.6) is based upon. The results provide an important context for policy and may help to identify priorities and linkages between regions. Additionally, they can be used to inform the selection of case study regions.

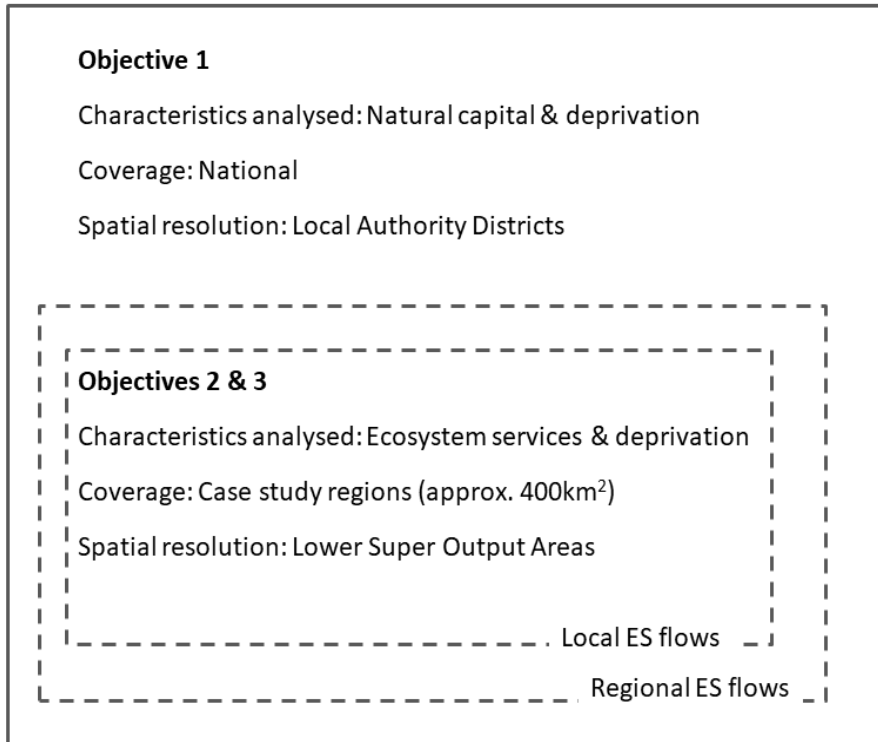


Figure 3.8 Multi-scale structure of research objectives. Case study analysis is situated within the wider context of a national analysis. More in-depth, spatially resolved analysis carried out within case study regions which also accounts for the benefits gained from natural capital extending beyond the case study boundaries.

Analysis within case study regions can utilise higher resolution data and more accurately reflect the spatially explicit ecological and/or social processes which generate ES and affect their distribution. The case study analysis itself will also incorporate multiple scales to account for different ES flows and utilise different input data dependent on the minimum service supply units. The case studies have a narrower focus (as it is not feasible to study all ESs) but provide a more detailed, reliable and locally relevant insight into who benefits from ESs by incorporating the key elements of ES supply, flow and demand as described by Figure 3.7. This information has greater relevance for potential planning and management interventions. This section of the analysis addresses objectives 2 (and 3).

Collectively, the national and local case study analyses help to inform whether or not environmental inequalities are more pervasive at a particular scale or location or with

regards to a particular type of natural capital or ES. In turn this can inform what the focus should be of further distributional assessments, and to what extent and at what level should policy and delivery seek to redress any inequalities.

The conceptual model is used to inform the different components of ES which need to be quantified. In this case methods need to be established to quantify and map ES supply (incorporating spatial flows) and ES demand for each of the selected ES within the case study regions. As highlighted, the scales of analysis underlie the decisions regarding the methodologies applied. Considering the conceptual models and research structure established, the subsequent sections in this chapter expand upon the methodological approach.

3.2 Selection of natural capital and ecosystem services

To ensure an appropriate selection of natural capital and ESs are included within the study, the full range of natural capital and ES must first be recognised. As discussed previously there are numerous conceptual models within ES research, in addition (and often arising from these) there are numerous ES classification systems. Classification systems facilitate modelling, assessment and discussion of ES (Costanza et al., 2017). Selection of natural capital (for objective 1: a national analysis) and ES (for objectives 2 & 3: case study analysis) should be realised from a clear and comprehensive classification. Therefore this section first reviews key classification systems and outlines the classifications used to select the natural capital and ES, before reasoning the ES chosen for analysis.

3.2.1 Natural capital

Natural capital tends to be described as a series of categories, as opposed to classifications of ES which are normally extensive lists of specific ES. More detailed definitions of natural capital tend to be associated with the study purpose, scale and location. The UK's Natural Capital Committee (NCC) (2015) lists 10 categories of natural capital assets; species, ecological communities, soils, freshwater, land, atmosphere, minerals, sub-soil assets, coasts, oceans. This incorporates both biotic (living) and abiotic (non-living) natural capital however the focus of this research is on biotic natural capital since it is this that generates ESs.

For biotic assets, the NCC (2015) framework references the MA global habitats classification, however the UK Broad Habitats (Table 3.2; Jackson, 2000) are better suited to the English context of this research. The UK Broad Habitats were established as part of the UK Biodiversity Action Plan to act as a framework for monitoring the

ecological condition of land and water of the UK. They were designed to be comprehensive and exclusive (i.e. a particular location only fits into one class) with simple definitions (Jackson, 2000). The broad habitats facilitate quantification of the extent of different aspects of several natural capital categories; the land, freshwater, coast, soil and ecological communities. Therefore, these constitute the natural capital included for objective 1's analysis (Chapter 4). The broad habitats were designated to broader groups in the UK NEA assessment (2011; see Table 3.2) and it is this classification which the indicators of natural capital are based upon (see Chapter 4, section 4.2.1). In addition to these, Chapter 4 section 4.2.1 outlines indicators related to species and soil categories. Detailed information of how the spatial distribution of this natural capital is quantified is also provided in section 4.2.2. Inclusion of this wide-range of natural capital provides a comprehensive overview of its distribution in England. It is focused on the living natural capital assets which provide ESs and ultimately benefit human health and wellbeing, however the natural capital selected also need to be quantifiable and within English boundaries therefore oceans are excluded.

Table 3.2 Types of natural capital defined by NCC's (2015) categories which includes abiotic assets and by UK broad habitats following UK NEA (2011). Italics indicates those not incorporate within this study.

Natural capital categories	UK broad habitats
Species	Woodlands
Ecological communities	Urban
Soils	Coastal margins
Freshwater	Semi-natural grassland
Land	Enclosed farmland
<i>Atmosphere</i>	Mountain, moorlands and heath
<i>Sub-soil assets</i>	Freshwaters
<i>Oceans</i>	<i>Marine</i>
<i>Coasts</i>	
<i>Minerals</i>	

3.2.2 Ecosystem services

ES classification systems are usually driven by an underlying conceptualisation (La Notte et al., 2017), such as those discussed in section 2. Quantification of multiple ES is important to ensure equality or fairness in outcomes (Tallis et al., 2008). However, for practical reasons, not all ES can be included within this study, rather the most

relevant and quantifiable ES must be selected. To make this selection it is first necessary to consider all potential ES and this is achieved through review of existing ES classification system.

The major ES classification systems include Daily (1997), MA (2005), TEEB (2010), UKNEA (2010), CICES (Haines-Young & Potschin, 2013). The MA (2005) is perhaps the most well-known classification and TEEB (2010) presents an update on this, whilst CICES was developed primarily for accounting purposes. All classifications group ES into provisioning, regulating, cultural and sometimes supporting services. In general provisioning services are 'biomass', or 'goods' such as food and timber. Regulating services tend to be based on interactions within ecosystems and describe processes which improve environmental quality such as improving soil quality. Cultural services describe any socially important uses or perceptions of ecosystems by humans and supporting services are ecological functions (La Notte et al., 2017).

The ES classifications list many of the same individual services within these categories, with just semantic differences (e.g. air purification compared to air quality regulation). However, there are some differences in content, especially notable between CICES and the other schemes (which principally align). One critique of the MA classification, is that it has limited scope, for example by focussing upon terrestrial ecosystems (Beaumont et al., 2007; Liqueste et al., 2013). This range of ES defined was broadened by CICES through inclusion of abiotic factors, however this does not adhere to the definition of ESs as being produced by ecological processes (Fisher et al., 2009). CICES also provides a more comprehensive classification by presenting a hierarchical system and employs more technical descriptions of each ES (CICES, 2017). Although many of these are less suited to the presentation and reporting of results compared to the simpler class descriptions of other classifications.

The most distinct differences in ES classifications occur with respect to the cultural and supporting services. In the cultural services group, TEEB and CICES include multiple ES such as 'education', 'recreation' and 'spiritual' (Table 3.3) but often others such as heritage are used. The UKNEA (2010) which is based on MA (2005) aggregates these to a single service of 'environmental settings' and Daily (1997) omits several of these individual ES. Including the multiple, individual cultural ES is beneficial since they will have different ES flows, supply and demand and trade-offs can be identified.

Supporting services introduced by the MA, are modified and included as a 'habitat' category by subsequent classifications such as TEEB, but are not recognised as a separate set of services by CICES (and included within the regulating category). A lack of consistency in how to incorporate supporting services is potentially due to the limited understanding of the complex relationship between biodiversity and other ES (Naidoo

et al., 2008; Anderson et al., 2009; TEEB, 2010) and also of ecological thresholds and resilience (TEEB, 2010). Supporting services are not included by some to avoid double counting and also because they may be considered 'invaluable'.

No single classification is optimal. Overall, the CICES and TEEB classes are clearly defined and comprehensive for each service category, although the ES included within each do differ. Principally, the TEEB classification is referred to for consideration of potential services for inclusion within this analysis (see the list of ES in Table 3.3). I select a subset of ES for distributional analysis in case study regions for objective 2 since some ES are more important than others, not all are possible to map across larger areas (e.g. city-wide), and methodological and practical constraints require a focus upon a selection of ES.

It is desirable that the selected ES:

- Have direct ES flows and therefore its SPAs and SBAs can be clearly identified
- Are important for the general population's health and wellbeing in England (i.e. are not solely beneficial to a population sub-group, e.g. farmers)
- Have ES flows which operate at different scales (and therefore different SPAs)
- Can be mapped across regions using nationally available existing datasets
- Incorporate different categories of ES

Supporting services and provisional services are not selected for further analysis in the case study regions. Regarding supporting services, by definition they underpin the provision of other services and their inclusion can potentially result in 'double-counting'; they do not directly provide benefits to human health and wellbeing (La Notte et al., 2017). Provisioning services are critical for life, supplying water, food and shelter. However, in high-income countries, the ES flows from source to beneficiary are indirect, highly diverse and based around complex socio-economic interactions. The welfare of an individual is reliant upon, for example, the type of food outlets available, ability to purchase nutritious food (Patteron et al., 2012) and the pricing of clean water (McDonald et al., 2010). The health and wellbeing of the general public is not generally dependent on the production of food in the local area rather the local benefits from these provisioning ES are primarily to business/land owners and employees.

Table 3.3. ES classification based on TEEB (2010) and with equivalent class from CICES (Haines-Young & Potschin, 2013).

I - international, N - national, R - regional, L – local indicate most dominant scales upon which the ES flows are important. Lowercase italics (*i, n, r, l*) represent scales upon which the ES flows have some importance. Scales assigned primarily based on Geijzendorffer and Roche (2014) and are relevant for high-income countries.

Ecosystem services	CICES class examples	Dominant scales
PROVISIONING		
Food	Biomass (nutrition), wild plants, algae and their outputs	N, I, <i>r, l</i>
Fresh water	Surface water for drinking Surface water for non-drinking purposes	R, I, <i>n</i>
Raw materials (e.g. wood)	Fibres and other materials from plants, algae and animals for direct use or processing	R, N, I, <i>l</i>
Fodder & fertilizer	Materials from plants, algae and animals for agricultural use	L, R, N, I
Genetic resources	Genetic materials from all biota	N, I
Medicinal resources	Fibres and other materials from plants, algae and animals for direct use or processing	N, I
Ornamental resources	Fibres and other materials from plants, algae and animals for direct use or processing	N, I
REGULATING		
Air pollutant removal	Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals and ecosystems	L
Climate regulation	Atmospheric composition and climate regulation	L, I
Noise regulation	Mediation of smell/noise/visual impacts	L
Flood regulation	Mediation of liquid flows	L, R
Drainage & natural irrigation (drought prevention)	Hydrological cycle and water flow maintenance	R, I, <i>n</i>

Ecosystem services	CICES class examples	Dominant scales
Water purification	Mediation of waste, toxics and other nuisances by biota/ecosystems	R, I
Erosion prevention	Mass stabilisation and control of erosion rates	L, R
Soil detoxification & soil fertility maintenance	Decomposition and fixing processes	R, I
Pollination	Pollination and seed dispersal	R, I
Pest & disease control	Pest control & disease control	L, R, n
CULTURAL		
Appreciated scenery	Aesthetic	L, R, n, i
Tourism	Physical use of land-/seascapes in different environmental settings	N, I
Recreation	Physical use of land-/seascapes in different environmental settings	L, R
Use in science & education	Scientific, entertainment, educational	L, R, n, i
Inspiration for art etc.	Symbolic	L, R, N, I
Cultural heritage		L, R, N
Spiritual & religious use	Sacred or religious	L, R, N, I
HABITAT/SUPPORTING		
Maintenance of genetic diversity	Lifecycle maintenance, habitat and gene pool protection	L, R, N, I
Habitats for species		

Cultural services are directly important for human health and wellbeing, however, many of these services such as “spiritual” or “inspiration for art” are less tangible. Their social construction and basis on individual preference and emotional response mean that these services are difficult to quantify across large areas. Rather small area qualitative analysis are best suited to these ES (Gomez-Baggethun and Barton, 2013). The most commonly mapped cultural service is recreation (Milcu et al., 2013). Although individually and culturally shaped preferences are still important for recreation, the supply of this service is fundamentally based on availability of accessible spaces (Nicholls, 2001). Given these physical requirements, it is possible to map recreation flows, supply and demand across large areas (e.g. Ala-Hulkko et al, 2016).

Regarding regulating services, some have more direct relevance to people’s health and wellbeing, whilst others are more indirectly beneficial because they underpin provisioning services. This includes pollination, maintenance of soil fertility, erosion prevention, water purification and natural irrigation and pest & disease control. These are primarily required for raw material, food and drinking water supply. Although it is acknowledged that erosion control is directly important for health and wellbeing of inhabitants of some coastal land, and water purification is directly important for the health and wellbeing of inland water recreational users. Climate regulation has benefits extending globally, however once more the spatial distribution of benefits is a result of indirect feedbacks, for example from the reduction of damage from extreme weather events. Due to the indirect ES flows of these services, as discussed for provisioning services, they are not included in this study.

With respect to the other regulating services included in Table 3.3; air pollutant removal, climate regulation (microclimate), flood regulation, noise regulation; they have direct benefits to human health and wellbeing. Considering the selection criteria, the dominant scales of these ES, as listed in Table 3.3, are considered. Examining ES which are pertinent at different scales enables consideration of the role of different areas of natural capital and ES flows, which in turn produce different spatial and potentially social distributions, and best demonstrate potential trade-offs between services.

Locally provided ES include air pollutant removal, climate regulation (microclimate only) and noise regulation. Local to regional scale services include flood regulation and recreation, disturbance prevention, regulation of water flows and recreation. Since flood regulation and recreation have different ES flows and represent both regulating and cultural ES they are both included in the analysis. Of the locally provided ES, air pollutant removal is of the greatest and most widespread concern for the health and

wellbeing of people in England. Selection of three services (air pollutant removal, flood regulation, recreation) is considered to be adequate to explore different ES categories and a range of scales and is achievable for multiple case study regions. The following sub-sections present a more detailed discussion regarding the relevance of these ES for health and wellbeing in England.

3.2.2.1 Flood prevention

Weather related hazards are the most common type of natural hazard in England; flooding specifically is classed as a major risk (HMG, 2015). Approximately 8% of the English population (12% land area) has greater than 0.1% annual chance of flooding from rivers and the sea. Impacts from recent flood events include the flooding of 20,000 properties in northern England in 2015/16, 11,000 properties in southern England in 2013/14 and 55,300 properties were flooded across many areas of the country in summer 2007. Government estimates the total cost of these events as roughly £5.8 billion (HMG, 2016).

Beyond the high financial costs, flooding impacts local communities and residents' quality of life, health and wellbeing. For example by limiting access to basic goods and services, including healthcare. Physical illnesses and deterioration of mental health may also result, adding strain to healthcare resources. Displacement and extended recovery processes can bring further strain in personal relationships (Menne & Murray, 2013). The Pitt Review commissioned by the UK Government to learn from 2007 floods found that 67% of the flood victims questioned felt repercussions for their or their partner's emotional health. Personal accounts also reported greater difficulty in managing existing illnesses and leading a healthy lifestyle (Pitt, 2007).

The implication of these potential effects from flood events is that a reduction of flood hazard as may be provided by natural environments, is an ES of particular importance to health and wellbeing in England. Consideration of the role of ecosystems is also necessary for managing flood risk in a more holistic manner as required by the 2007 European Floods Directive (Council Directive, 2007/60/EC).

Amounts of surface water runoff are a key factor in flood hazard (Whitford et al., 2001). Surface water runoff is determined by interactions between land cover, soil permeability, antecedent ground conditions, geology, topography and the water channel network (Duku et al., 2015). Reducing and slowing the flow of surface water can contribute to lesser and delayed flood peaks (although this is dependent upon the spatial and temporal interaction of hydrological processes) and reduce the pressure on constructed drainage systems (Whitford et al., 2001). Permeable natural land cover

and soils contribute to a flood regulation ES, mediating flood hazard from rivers and surface water flooding. The relevance of this ES is echoed by the recent direction of UK flood management policy towards 'natural flood management' in addition to engineering solutions (Dadson et al., 2017) as illustrated by the Environment Agency's recent project "Working with Natural Processes" (Environment Agency, 2017b).

The flows of flood regulation ES are direct based on hydrological processes which operate at the catchment scale. The beneficiaries of this ES are those who are located downstream or in the immediate vicinity where runoff is reduced (in the case of surface water flooding) (Syrbe & Walz, 2010). Several studies have demonstrated how surface water runoff reduction by ecosystems may be mapped (e.g. Whitford et al., 2001; Tratalos et al., 2007; Zeng et al., 2017). The term surface water runoff reduction (SWRR) is preferred and used hereafter since potential methods demonstrated explicitly quantify this, with the assumption that flood hazard will consequently be reduced. Considering the aims and scope of the research, quantification of SWRR as opposed to intensive full hydrological modelling is deemed appropriate for fulfilling the study objectives.

3.2.2.2 Air pollutant removal

Outdoor air pollution is a major public health concern globally. Lelieveld et al. (2015) project global premature deaths from outdoor air pollution to reach 6.6 million annually by 2050. Airborne particulates which can penetrate the respiratory system are believed to pose the greatest risk to health, with an estimated 391,000 premature deaths from long-term exposure in the EU in 2015 (EEA, 2018). Additionally, based on concentrations in 2015, 76,000 and 16,400 premature deaths from NO₂ and O₃ exposure respectively are estimated annually in the EU (EEA, 2018). For England, it is estimated that premature deaths attributed to exposure to PM_{2.5} are 29,000 annually (at 2008 pollutant concentrations) (COMEAP, 2010) and to NO₂ are between 28,000 and 36,000 annually (COMEAP, 2018). Air pollutants not only reduce length of life but also have lifelong health impacts with high vulnerabilities for children and elderly. Issues in childhood include harmful effects to the heart, immune system and respiratory problems (Arden Pope, 2000; RCPCH, 2016). Together the health costs burden of air pollution in England is estimated at £20 billion annually (RCPCH, 2016).

Given the severity of the issues arising from poor air quality, the Air Quality Directive (2008/50/EC) was implemented in the EU to define air quality standards and requires the monitoring of sulphur dioxide, nitrogen dioxide, particulates, lead, benzene and carbon monoxide against these standards. The Air Quality Standards Regulations 2010

implement these standards in UK law. Whilst overall emissions of air pollutants decreased from 2000 to 2016, exceedances of standards are common. In 2014 daily limits of concentrations were exceeded with respect to PM₁₀ in 19 EU countries, O₃ in 17 EU countries (more than 25 times) and NO_x annual limits were exceeded in 17 EU countries (EEA, 2018).

Although air quality standard exceedances tend to be of greatest concern in urban areas, it is not solely an urban concern. Pollutant concentrations higher than legal limits have been recorded in rural areas, for example, 15% rural areas in the EU recorded exceedances of the PM₁₀ daily limit in 2011 (twice that recorded in 2009) (EEA, 2018). There is also no known 'safe' level of air pollutants - standards are set on the basis of health risks being 'acceptable' – and even in areas where levels are lower, there may be a degree of health risk, especially for the most vulnerable (WHO, 2006).

Approaches to tackling poor air quality necessarily focus on reduction of emissions from major sources such as road transport (e.g. via emission technology and clean air zones, DEFRA, 2019). However, vegetation plays a role in improving air quality through either absorbing gaseous pollutants or through the deposition of particles onto the vegetation itself, which constitutes the air pollutant removal ES.

Given the potential role of vegetation in helping to address the major issues faced nationally and internationally regarding air quality and which have direct consequences to health, the air pollutant removal ES is included for analysis in case study regions. The decision to assess the social distribution of this ES has further relevance given the known issues regarding inequalities in exposure to poor air quality (e.g. Mitchell et al., 2015). Furthermore, ES flows are local, contrasting with the catchment scale of SWRR, thus providing a more comprehensive insight into the social distribution of ESs.

3.2.2.3 Recreation

When referred to as an ES, recreation incorporates any leisure activity in outdoor natural environments such as woodland, rights of ways and alongside rivers and coastlines. Recreational activities range from sedentary to physically vigorous, and examples include appreciating scenery, fishing, playing with children or adventure sports (Natural England, 2018b).

In England, the annual Monitoring Engagement with the Natural Environment (MENE) survey has recorded details of and changes over time in outdoor recreational visits since 2009 (Natural England, 2018b). This emphasises the broad and increasing participation, and thus the importance of this ES to people's wellbeing. The last annual report for the period 2017/18 found 62% of the adult population visit the natural

environment at least once a week, whilst 93% consider close to home greenspace is important. However, notable inequalities have also been revealed, with BAME populations and lower socio-economic groups less likely to undertake recreation in the natural environment. Specifically, for 2017/18 51% of respondents in the 10% most deprived areas visited the natural environment at least once a week, compared to a higher 72% for the 10% least deprived (Natural England, 2018b).

The benefit of outdoor recreation for human health and wellbeing has long been recognised. For example, in 1833 a UK Government Select Committee on Public Walks sought to safeguard open spaces for public walks near towns for “the health and comfort of the inhabitants” (HC, 1833). To this day, the most common activity recorded by MENE is walking (Natural England, 2015). There are numerous mechanisms by which outdoor recreation facilitates improvements in health and wellbeing. The most direct physical link is that exposure to natural environments helps to develop a healthy internal biome, promoting healthy immune systems and reducing inflammatory-based diseases (e.g. asthma) (Rook, 2013; Sandifer et al., 2015). Others have found lower mortality in the greenest areas (van den Berg et al., 2015) and lower rates of obesity for those with greater exposure to natural environments (Coombes et al., 2010), although the strength of the relationship is dependent on other socio-demographic factors.

There is also some evidence of positive impacts on mental health; studies have found reduced stress, anxiety, fatigue and improved self-esteem and mood through recreation in natural environments, particularly in green environments which are perceived to be high quality (Weimann et al., 2015; Houlden et al., 2018) compared to outdoor recreation in non-green surroundings (Barton and Petty, 2010; Bowler et al., 2010). There are however some inconsistencies in the evidence; for example Buchecker et al. (2015) found only a marginal, but significant, impact of regular outdoor recreation in a natural environment with self-reported well-being and psychological resilience. Other studies have noted that mental-health benefits are often short-term (Alcock et al., 2014; Hartig et al., 2014) and Gascon et al. (2015) found no significant relationship for children. Considering broader aspects of wellbeing, recreation in parks and other local community settings is also considered important for social cohesion (Jennings & Bamkole, 2019).

Typical of developing countries, in England, mental health and non-communicable diseases are now the main causes of years of life lost or worsening quality of life (PHE, 2018). Given the potential links to health outcomes recreation ES plays an important part of a holistic solution to increased prevention of such illnesses. Recognising its

importance, guidelines regarding the minimum access to recreation have been produced for England (Handley et al., 2003). Internationally, the UN Convention on the Rights of the Child asserts the rights of children to participate in recreational activities.

Recreation is selected for analysis within case study regions given the context of its importance (increasingly so) for human health and wellbeing, but also since mapping access to recreation has been widely achieved, providing a set of methods to draw upon (e.g. Wood, 1961; Rossi et al., 2015; Liu et al., 2017). These studies observe that recreation usually takes place close to home, but that travel across short distances often occurs. Therefore, analysis of recreation ES considers a contrasting scale of ES flows to both SWRR and air pollutant removal.

To summarise, the natural capital assets considered for assessment of their social distribution nationally (objective 1) are as wide-ranging as possible. For assessing the social distribution of ES within case study regions (objectives 2 & 3) the ES included for analysis require a more selective approach. This considers factors such as who the immediate beneficiaries of the service are, the feasibility of mapping the spatial (and social) distribution of the service across large areas, their importance for the health and wellbeing of the general population in England and that different spatial scales of ES flows may affect the distribution of services. The ES to be included are air purification, recreation and SWRR. The approach to mapping these ES are discussed in the next section whilst detailed methods are provided in the respective chapters (Chapters 5, 6 & 7) since reliable mapping of ES requires a tailored approach. Similarly, the specific methods for mapping natural capital are detailed in Chapter 4.

3.3 Methodology

This section addresses the main approach to mapping ES and assessing their social distribution within the case study regions. The methods for determining the spatial distribution of natural capital are detailed in Chapter 4. The spatial model presented in Figure 3.7 guides the approach to mapping ES. Given a review of existing methodologies, I also discuss the tests that could best reveal the sensitivity of results to modelling assumptions and potential uncertainties. Once the spatial distribution of natural capital and ES are established, the final stage in analysis for each objective is to determine how they are socially distributed. Since this analysis is carried out in the same way for all ES the common details of the methods are presented here, avoiding repetition in the individual analysis chapters

For this research, the analysis will be quantitative, utilising secondary data and relatively simple spatial analysis techniques. These characteristics are largely driven by the large geographical areas of interest as well as the practical considerations such as data availability and resources. However, as highlighted, this approach also enables a range of variables to be explored and the use of secondary data enables methodologies to be adopted for other areas.

3.3.1 Modelling ecosystem services

3.3.1.1 A review of ecosystem service modelling approaches

Methods for quantifying and mapping ESs and natural capital may broadly be either monetary or non-monetary and biophysical or socio-cultural (Harrison et al., 2018). Monetary based valuations of natural capital and ESs are important for furthering “conservation efforts in the real world” (Luisetti et al., 2014, p.685). As public goods and services, they are usually market externalities, thus economic valuation enables their incorporation within economic based decision making and national accounting.

ESs are an inherently utilitarian conception, hence monetisation of ES is widely accepted. However, the limitations of methods applied for economic valuation have been widely recognised, and include the inability to separate market and policy influences from individual preferences (Pascual et al., 2010), the lack of ability to calculate societal values in addition to individual values and the lack of time and support for participants to carefully consider complex issues (Whittington et al., 1992; Pascual et al., 2010). In general, monetary values are derived for specific geographies, beyond which it is presumptuous to apply them (the value transfer problem); natural capital and ES provide multiple benefits, which may vary in importance to different

people in different places and for different reasons. Monetary values may also confuse non-experts and stakeholders who have limited knowledge of how these are calculated (Costanza et al., 2017)

Given these limitations and that to better understand the social distribution of ES, knowledge of relative ES supply and demand is of greater importance than monetary values, monetary valuation is not required hence non-monetary methods are adopted. Advantages of non-monetary approaches include an ability to more easily incorporate a wider variety of ES enabling a more comprehensive analysis recognising trade-offs, synergies and cumulative effects (Burkhard et al., 2012). Less data and resources may also be required (Burkhard et al., 2014). There are multiple methods for mapping ESs in non-monetary terms, and often assessments focus upon supply (Bagstad et al., 2013), but the majority of approaches can be applied to model demand too, and the merits and limitations of potential methods are relevant for both supply and demand (see Martínez-Harms and Balvanera, 2012; Wolff et al., 2015 and Harrison et al., 2018 for reviews of approaches mapping ES supply, demand and both respectively).

In their review of 43 methods available for quantifying ES, Harrison et al. (2018) list 7 main types of broadly biophysical method and 8 main types of broadly socio-cultural method, but note that often methods are difficult to classify or are a hybrid. The socio-cultural methods are of particular importance for accounting for procedural and participatory aspects of justice. However, the aim here is to establish whether inequalities exist over relatively large case studies (i.e. relevant to local planning) and results from participatory techniques are not suitable for extrapolation across large areas. Furthermore, biophysical methods are appropriate for the selected ES.

Biophysical methods include process-based modelling (e.g. hydrological modelling), the use of direct measurement (e.g. of air quality), simple proxies (e.g. matrix of values for all ES assigned to land cover classes based on expert opinion), proxies based on causal relationships (e.g. values for carbon storage based on multiple environmental datasets and known relationships between them), and simple GIS mapping (e.g. mapping recreation sites) (Martínez-Harms and Balvanera, 2012; Harrison et al., 2018). A combination of methods may also be applied (Harrison et al., 2018). Factors which commonly determine the selected method include the study purpose, size of study area, desired accuracy and spatial scales, ESs included, whether novel or established techniques are desirable and practical constraints such as time, costs, data availability, and expertise drive the selection of methods (Harrison et al., 2018).

Proxy-based methods are the most common approach. Of these, the use of look-up tables is the simplest, whereby an index is usually assigned to each land cover type

(Maes et al., 2011; Burkhard et al., 2012). The indexes can indicate ES supply or demand and may be derived from other studies, expert opinion or stakeholder consultations (Martínez-Harms and Balvanera, 2012). The key advantage of this approach is its simplicity and it is often a solution when practical constraints are limiting. It is especially useful for mapping ES across large areas since data requirements and processing times are lower than other methods (Naidoo et al., 2008, Burkhard et al., 2012). Furthermore, it's an approach which can be easily understood by stakeholders, making it useful for raising awareness of potential impacts of changes in land cover (Vihervaara et al., 2010; Burkhard et al., 2012). On the other hand, several authors have doubted the accuracy of results from application of this method (Kienast et al., 2009, Eigenbrod et al., 2010, Lautenbach et al., 2011, Geijzendorffer and Roche, 2013 and Hou et al., 2013). ES are modelled as homogenous across large areas with the same land cover type, producing a uniformity error; indices based on stakeholder or expert opinion are highly subjective and inappropriate transfer from other study areas can lead to sampling and regionalisation errors (Troy and Wilson, 2006; Plummer, 2009; Martínez-Harms and Balvanera, 2012).

Alternatively, proxies developed from known causal relationships retain the advantage of being straightforward to apply across large areas, but are more complex and may have more specific dataset requirements. The accuracy will vary depending upon the ES, reliability of the causal model and datasets and applicability to the study area. A review by Martínez-Harms and Balvanera (2012) found that casual models were the most commonly applied method, although this contrasts with Shägner et al. (2013) who found that the simpler proxy estimations were more frequently used. In their critique of proxy based approaches Eigenbrod et al. (2010a) did note some increased accuracy when causal models are used.

Extrapolation of primary data across larger areas can also be subject to uniformity error, thus not fully representing the distribution of ES (Martínez-Harms & Balvanera, 2012). As a validated approach, it is advocated by Shägner et al. (2013) as a means of improving ES quantification, however increased complexity and its reliance upon primary data leads to greater resource requirements (Seppelt et al., 2011). Biophysical models also tend to be discipline specific applying knowledge of processes and functions, examples include soil erosion and hydrology models. Although these better represent complex processes, they are available for a limited number of ESs and tend to require intensive resources and discipline specific expertise (Rieb et al., 2017).

There are several tools developed specifically for modelling ES (see Sharps et al, 2017 and the 'Tool Assessor' at <https://ecosystemsknowledge.net/tool> for a comparison of

tools). These employ a variety of the techniques discussed above depending on study purpose and which ES are addressed. Several also explicitly incorporate both ES supply and demand (e.g. ARIES, Villa et al., 2014; Eco-Serv, Winn et al., 2015; Invest, Sharp et al., 2018). Furthermore some are based upon more novel approaches to ES quantification which are emerging. For example, the ARIES project (www.ariesonline.org) combines agent-based modelling with Bayesian Belief Networks. This is especially effective at modelling ES flows (including their complexities such as the concepts of sinks and rivalry) (Villa et al., 2014) and in situations where data is insufficient (Vigerstol and Aukema, 2011). Its implementation is currently limited by the requirement for a high level of technical skill and involvement with its creators (Sharps et al., 2017). In general the limitation to the use ES toolkits is whether they are suited to the particular study including the ES to be assessed, scale, size of study and data availability (Harrison et al., 2018).

Regardless of methods used, ES assessments have been criticised for a lack of accuracy, validation testing and sensitivity testing (Plummer, 2009; Shägner et al., 2014). Several studies have attempted to quantify potential errors resulting from particular methods (Eigenbrod et al. 2010a, b; Lautenbach et al., 2011; Martínez-Harms and Balvanera, 2012; Geijzendorffer and Roche, 2013; Hou et al., 2013; Shägner et al., 2013; Van der Biest, 2015). There is a general consensus that simple proxy based methods based on land cover tend to produce the poorest estimates of ES supply. Although the reliability of simple proxies was shown to be dependent upon the ES (Van der Biest et al., 2015) and these studies have not examined the accuracy of proxies for several ESs. The proxies tested by Eigenbrod et al. (2010) corresponded to only three ESs and were applied to a coarse (10 km² resolution) dataset and therefore it can perhaps be expected that there was only weak correlation with extrapolated primary data. Some argue that modelling ES using potentially unreliable methods may lead to detrimental land management decisions or inappropriate application of findings (Eigenbrod et al., 2010; Seppelt et al. 2011). Others, responding to the urgent need to account for ES within decision making, prefer the use of potentially less reliable methods to a 'do nothing scenario', but emphasise that the limitations are clearly communicated and the potential inaccuracies are carefully considered (TEEB, 2010). Therefore overall, where possible approaches which better account for the complexity of production of ESs than the use of simple land cover proxies should be taken.

The quality and credibility of ES modelling should also be better addressed by applying validation and/or sensitivity analysis as standard practice (Martinez-Harms and Balvanera, 2012; Hamel and Bryant, 2017). This is demonstrated by some of the

available ES modelling tools; ARIES shows the spatial variations in errors of ES estimations (Villa et al., 2014), INVEST provides optional uncertainty analysis through for example the use of Monte Carlo simulations (Sharp et al., 2018). However, there remains a lack of studies which incorporate uncertainty analysis; Seppelt et al. (2011) found one third of studies reviewed (153 in total) included basic quantitative analysis of uncertainty. Hamel and Bryant (2017) attribute this to the emerging nature of the field which is interdisciplinary and requires consideration of complex socio-ecological interactions and the perception that uncertainty analysis is inherently complex and time-consuming amongst other factors. Indeed, Hamel and Bryant (2017) propose that relatively simple approaches to uncertainty analysis are still effective and compel researchers to evaluate model assumptions and the robustness of results. One approach they advocate involves recognition and ranking of potential sources of uncertainties and focussing only those which are likely to have the most notable impact. For these, assessing the effects on results can be achieved by rerunning models for upper and lower bounds.

3.3.1.2 Study methods overview

A spatial analysis approach is taken in this study to assess how natural capital is socially distributed nationally (objective 1) and how air pollutant removal, recreation and SWRR are socially distributed in case study regions (objectives 2 & 3). Analysis of spatial data is undertaken in ESRI ArcGIS 10.3, QGIS 3.0, Geospatial Modelling Environment (Hawthorne, 2001) and GeoDa 1.10 (Anselin, 2003). This section summarises the approaches to modelling each of the selected ES for this analysis, with more detailed accounts given in the respective ES chapters (5-7).

Multiple case study regions are used to address objective 2 as this provides insight into a range of natural and social environments. They should also be large enough to statistically assess the social distribution of the selected ES and to have relevance for local planning. Objective 3 addresses the concerns raised in the previous section regarding the need to be aware of how robust outputs are to uncertainties and how underlying model assumptions may impact on the results. This will require sensitivity analysis which entails re-running ES models multiple times using alternative inputs. Since the social data and ES estimates are linked by their location, models which best convey the spatial variability of ES are desirable. Furthermore, following the conceptual basis for modelling, it is essential that the ES supply, demand and ES flows are fully accounted for in the ES models applied. Since the scale of ES flows for recreation and SWRR are deemed as local to regional, the potential size of areas across which ES supply may need to be modelled could be extended considerably. Since multiple and

potentially large case study regions will be assessed, the use of secondary data which is readily available nationally is preferable. This also demonstrates replicability of the analysis in other areas.

Given the scope of the analysis, the methods applied will need to consider practical constraints such as time and data availability. Ideally 'off the shelf' models provided by ES toolsets would be applied, however review of available tools revealed that this would not be suitable. Reasons include, that some do not include the selected ES, extensive or primary data requirements, a lack of applicability in these areas, or an approach which was considered inappropriate. For example, INVEST models recreation on the basis on photos uploaded to an online sharing platform; this however is not suitable for assessing all types of recreation space, such as a local sports field. Eco-serv addresses all of the ES selected but social data is in-built in many of the models and this would prevent independent comparison of ES estimates to social data.

Therefore methods used for modelling ES are developed specifically for this analysis, utilising other studies such as Whitford et al. (2001), Holt et al. (2015), and Tratalos et al. (2007) who mapped multiple ES for city-wide case studies in the UK. The outputs of the methods need to indicate where ES supply and demand is higher and lower, however, absolute values are not produced. Methods will be tailored for each ES and ideally go beyond the use of simple land cover proxies, given their limitations highlighted above. However, considering the requirements observed above, practical constraints such as the time taken to implement each method preclude the use of complex bio-physical models. In general, for air pollutant removal and SWRR the input data for modelling supply consists of land cover classifications tailored to match the key drivers of the ES supply and the minimum service providing unit. In addition, knowledge of causal relationships is applied which considers interactions of the land cover with other environmental factors such as soil type (SWRR) and air pollutant concentrations (air pollutant removal). For recreation a simple mapping approach is used, which combines existing datasets with mapping from aerial imagery to identify recreation supply. With respect to ES demand modelling approaches are largely based on the concepts of demand reviewed by Wolff et al. (2015). For example, demand for regulating services is based on the need to reduce a particular hazard. For the selected ES, this information is readily available from secondary datasets.

To fully account for ES supply, ES flows must be considered, as described in section 3.1.3. For this analysis, the case study regions define the area where the beneficiaries are located (i.e. the SBA). It is for this area (i.e. within the case study boundaries) that ES demand is modelled. Knowledge of the spatial ES flows of each service is used to define the potential SPA, for which ES supply is computed. Once supply and demand

maps are generated, their values are aggregated to administrative units. The aggregated values (x) are normalised to rescale values from 0-10, whereby 0 indicates the lowest supply or lowest demand and 10 indicates the highest supply or highest demand (Equation 3.1). It should be noted that the values are relative, for example a value of '5' assigned to supply does not mean the needs are met if a value of '5' is also assigned to demand.

$$x_{new} = 10 \left(\frac{x - x_{min}}{x_{max} - x_{min}} \right) \dots \dots \dots \text{Equation 3.1}$$

The use of normalised values facilitates their comparison and enables demand to be subtracted from supply to generate a single net ES index. A net ES index of '0' therefore indicates the area(s) with the greatest negative difference between supply and demand (i.e. higher demand and lower supply). Net ES index of '10' indicates the area(s) with the greatest positive difference supply and demand (i.e. lower demand and higher supply).

Sensitivity tests are undertaken for all ES and address issues of uncertainty and in underlying assumptions. The tests involve re-calculating the social distribution of the ES given variations applied to supply and demand models. Sources of uncertainty include the spatial, thematic and temporal accuracy of datasets. For air pollutant removal and SWRR there is uncertainty regarding the accuracy of values assigned to land cover classes to indicate their contribution to ES supply. For recreation there is uncertainty in the identification of which recreation spaces should be included. In addition there are assumptions that the selected methods are a reasonable representation of a complex set of processes regarding ES supply, demand and ES flows. The sensitivity tests for ES supply are bespoke to the service but are based upon the estimations of ES applied to the data. This is considered the greater potential source of error rather than uncertainties inherent in the data itself. Accounts of the specific variations applied are given in the individual ES chapters (5-7).

Table 3.4. Summary of methods for modelling ES supply and demand. Methods are tailored to each ES and explained in detail in Chapters 5-7.

ES	Supply	Demand	Flow
Air purification (Chapter 5)	Proxy based on pollutant removal rates of vegetation types (from classified 5m resolution CIR imagery) weighted by proximity to roads as a proxy for air pollutant concentration.	Distance from roads	Local (SPAs = SBAs).
Recreation (Chapter 6)	Polygon vectors defining boundaries of accessible recreational spaces.	Demand is set as equal for all on the basis of supply calculated is a minimum standard.	Euclidean distances (approximated travel distances adapted to size of recreational spaces up to 10km buffer).
Surface water runoff reduction (Chapter 7)	Runoff calculated using SCS CN method applied to combined land cover/ soil data. The reduction in runoff by natural land cover is calculated by comparison to a hypothetical scenario where all land cover is assigned impermeable.	Flood risk from multiple sources.	River catchment.

For demand, the greatest potential impact upon results is likely due to its conceptualisation. For the baseline scenarios, the demand is not based on per capita needs as the services are deemed as non-rival and non-exclusive (see Schröter et al., 2012). However, from a utilitarian perspective the number of people who need/potentially benefit from the ES is important, specifically for making economically based decisions (e.g. Burkhard et al., 2012; Holzinger et al., 2013). Therefore sensitivity to models of ES demand is tested for all ES by adding a weighting based on population density.

3.3.2 Determining environmental equality

Once ES supply, demand and net ES index are quantified, their values are compared to socio-economic data to determine their social distribution. The following section describes the data and methods used to achieve this. Section 3.3.2.2 discusses the need in the first instance for establishing a common scale for the environmental and

social data. The final section in this chapter (3.3.2.2) examines the methods used to evaluate inequality and highlights those applied in this analysis.

3.3.2.1 Socio-economic data

In the UK, environmental inequalities have largely been shown with respect to socio-economic status and/or multiple deprivation (Briggs et al., 2008), and comparisons of the mapped ES to deprivation form the basis of the inequality assessment carried out in this study. Measures of deprivation summarise relative disadvantage across a range of material and non-material factors. People may be deprived of income, adequate housing, employment opportunities and environmental amenities (Dorling, 1996). Deprivation cannot be directly measured, hence small area input variables are used to construct composite indices. Several deprivation indices and measures of socio-economic status have commonly been used in England, these are summarised in Table 3.5 (also see Mitchell et al., 2015; Fairburn et al., 2016).

Those based on data from the national census (Townsend deprivation index, Jarman Underprivileged score, NS-SEC, Breadline Britain Index) have the advantage of being nationally available for several nested geographies and comparable over time (Norman, 2010). The disadvantages of census based measures include that the data is only updated every 10 years and that they may omit other available relevant indicators in particular those which address rural deprivation (Higgs & White, 2000).

The IMD has been developed since 2000 and is the key deprivation index for UK central and Local Government, for example, in local needs assessments and for allocation of funding for economic and social programmes (Fairburn et al., 2016). Fairburn et al. (2016) give a detailed account of its development and application within in UK based environmental justice research. The index is constructed from indicators covering seven domains, weighted prior to aggregation, including income, employment, education, skills and training, health and disability, crime, barriers to housing and services, and living environment (DCLG, 2015). The data used to generate the domains are derived from various administrative sources and comprise of 37 individual indicators, for example, claimants for jobseekers allowance, recorded crime and school exam results (DCLG, 2015). This study uses the IMD to assess inequality since this is a comprehensive, regularly updated and commonly used measure of deprivation. IMD 2015 was the most current release available at the time of analysis and is primarily constructed from 2012-2013 data (Smith et al., 2015). For each area, the IMD provides a relative ranking whereby the lowest ranks indicate the most deprived areas and the highest ranks, the least deprived.

Table 3.5 Common metrics of deprivation in England

Measure of deprivation	Data source	Variables included
Townsend index (Townsend, 1987)	Census	% Households with no car % Overcrowded households % Non-owner occupied houses % Unemployed (of economically active residents)
Jarman Underprivileged score (Jarman, 1983)	Census	% Unemployment % overcrowding % lone pensioners % single parents % born in commonwealth % children under 5 % low social class % one year migrants
ONS National Statistics Socio-economic Classification (NS-SEC) (based on Goldthorpe, 2007)	Census	Occupation based, includes the following classes: 1. Higher managerial, administrative and professional 2. Lower managerial, administrative and professional occupations 3. Intermediate occupations 4. Small employers and own account workers 5. Lower supervisory and technical occupations 6. Semi-routine occupations 7. Routine occupations 8. Never worked and long-term unemployed
Breadline Britain Index (Gordon, 1997)	Breadline survey/Living standards survey & census	Weightings derived from survey data and applied to the following census variables: % Overcrowded households % Houses rented from local government % Lone-parent households % Households with an unemployed reference person % Households with no car % Privately rented households % Households with a member with a limiting long-term illness % Households no central heating % Households with a reference person is a low social class (NS-SEC 6, 7 or 8)
Index of Multiple Deprivation (DCLG, 2015)	Multiple	37 separate indicators organised into 7 weighted domains: Income (22.5%) Employment (22.5%) Health & disability (13.5%) Education, skills & training (13.5%) Barriers to housing & services (9.3) Crime (9.3%) Living Environment (9.3%)

One limitation to the use of the IMD 2015 is the inclusion of the living environment variable which incorporates air quality (and thus will not be completely independent for assessment of the social distribution of the air pollutant removal ES). However, this domain is one of the three with the lowest weighting (9.3%) and thus should not substantially change the results. Moreover, the various deprivation indices are known to be correlated (Norman, 2010).

3.3.2.2 Matching scales of environmental and socio-economic data

Spatial analysis concerned with environmental equity require socio-economic and environmental data to be spatially linked. Social and environmental data may be spatially represented in multiple different ways according to the differences in the phenomena they represent, how the data has been collected and its likely purpose. The data may be of different quality in terms of accuracy, coverage, sample size, spatial resolution or temporal currency, it may also be of different type (e.g. continuous or discrete) and formats (e.g. raster surface or vector). All of these factors influence how an analysis can be carried out including the scales at which data can be compared, which in turn may influence the degree to which (in)equalities are revealed (Mitchell & Walker, 2003; Baden et al., 2007).

Deprivation indexes and other socio-economic data are usually aggregated datasets. Data corresponding to individuals or households are aggregated to a coarser spatial unit in order to preserve the respondents' anonymity (ONS, n.d.). The smallest geographical boundaries for which IMD data is available are Lower Super Output Areas (LSOAs). LSOAs are delimited in a way which aims for consistency in the population they contain (a mean of 1500), but are variable in physical size (ONS, n.d.). IMD2015 indices aggregated to Local Authority District (LAD) boundaries, which are much larger areas relevant to local government, are also available. In 2011, there were 32844 LSOAs in England and 326 LADs.

Some indicators of environmental conditions are similarly aggregated, for example, the percentage of LSOA area covered by greenspace (e.g. Generalised Land Use Database, 2005). However, it is usually presented in the manner most appropriate for the features and characteristics it describes. For modelling ES supply and demand a range of input spatial datasets are combined and the scale of mapping is relative to the service considered, consistent with the ES SPUs. For example, air pollutant removal supply and demand are mapped as continuous raster surfaces at a fine scale on the basis that the minimum SPU is a single tree. Conversely, recreation supply is mapped as a series of discrete polygons a minimum of 300 m² in size. However, for almost all

datasets, environmental data are represented at a finer scale than the IMD (LSOAs). Thus the scale of the inequality assessment is guided by the administrative units for which IMD data is available. Specifically, LADs are used for the national analysis and the use of this coarser spatial unit is discussed in greater detail in Chapter 4. For analysis at a finer scale in the case study regions, LSOAs are used.

Therefore, once mapped, it is necessary to aggregate natural capital indicators to LADs (objective 1) and ES supply and demand to LSOAs (objectives 2 & 3). Results will therefore be subject to the effects of data aggregation, including the Modifiable Areal Unit Problem (MAUP) and ecological fallacy (Gehlke and Biehl, 1934; Openshaw, 1984). MAUP describes two effects; a scale effect whereby the size of aggregated areas impacts upon the results and the zone effect where the placement of the aggregated area boundaries impacts upon the results. As a consequence of MAUP, correlation has been shown to increase with larger areal units (Gehlke and Biehl, 1934; Fotheringham et al., 2000). Baden et al. (2007) find that the magnitude of correlation changes with aggregation, but they note that the direction of the relationship does not and that correlation is also dependent upon the chosen size of study area. Fotheringham and Wong (1991) also observed that the impact of aggregation varies depending on variables. It is thus expected that the boundary effects will also differ for each ES modelled in this analysis. Whilst a boundary effect cannot be eliminated, it is mitigated in this analysis through the consideration of ES flows and therefore supply beyond the LSOA boundaries which better accounts for the different scales upon which social and biophysical processes operate. Ecological fallacy refers to the inference that a statistical outcome, for example the correlation between variables or the level of deprivation for a group, also applies to at an individual level, and vice-versa (Robinson, 1950). Careful interpretation of results can mitigate against this concern.

To aggregate ES supply and ES demand to LSOAs, an area weighted mean (Equation 3.2) is computed. This better reflects the dominance of values within an area, with lower sensitivity to low coverage of extreme values. Recreation ES is determined by areal coverage only and therefore average coverage per km² is computed.

$$\mu_{aw} = \frac{\sum Cx}{\sum C} \dots \dots \dots \text{Equation 3.2}$$

μ_{aw} = area weighted mean,

C = area within polygon covered by x,

x = continuous value (ES)

Aggregation of ES supply and demand prior to merging into a single index enables their comparison to IMD ranks in addition to net ES index. This can reveal more about the factors determining the social distribution of the ESs. The net ES index is calculated at the aggregated level.

It should also be noted that reliable results also require social and environmental data to correspond as closely as possible temporally (Haining, 2003). Datasets used in this analysis are not all available for the same years, therefore the most recent versions of the data at the time of analysis are used.

3.3.2.3 Measuring inequality

Numerous statistical measures have been applied to assess degrees of inequality in relation to multiple concerns including income, health and the environment (e.g. Harper et al., 2013). Table 3.6 summarises those commonly employed in EJ studies to determine how (un)equal the distribution of environmental benefits or hazards are across different socio-economic groups. There is no 'correct' way of measuring inequality and results can depend on the statistical technique used. To avoid this and to develop a strong understanding patterns of any inequalities, most EJ studies employ a range of techniques (e.g. Chakraborty et al., 2011; Boyce et al., 2016; Pope & Boone, 2016). For this analysis, associations between deprivation and ES are examined in several ways, combining descriptive statistics with inequality indices.

Descriptive statistics

Descriptive statistics provide overviews of distributions in a manner which is widely understood. For this analysis, boxplots are used to visualise the distributions of ES across population-weighted deprivation deciles. Boxplots are effective since they convey a large amount of information, including; changes in ES across deprivation deciles (including non-linear), central tendency, the dispersion of values, and the presence of any outliers across deprivation deciles. In this case the 'boxes' show the median, upper and lower quartiles of LSOA ES values, with outliers indicating values beyond 1.5 x interquartile range. Population-weighted deprivation deciles are generated by ordering LSOAs by IMD ranks and assigning these to 10 groups (deciles) with approximately equal populations. Population weighting produces deciles which represent roughly 10% of the total population rather than 10% of the number of LSOAs. Boxplots are generated with respect to ES supply, demand and net ES index.

Correlations

Regression and correlation are common ways of assessing inequalities in environmental justice literature (Miao et al., 2015). Regression models are not used in this instance since they requires five strong assumptions (rarely tested for in the literature) and infer causality (Bowen, 2002) which is not the intention of this research. For quantifying statistically significant association between deprivation and ES, Spearman Rank correlation coefficients are calculated. Spearman Ranks are used here since, despite being a statistically weaker test, it is non-parametric and therefore appropriate for applying to ranked deprivation data.

Concentration curve & concentration index

In addition to correlation, relative inequality is assessed using concentration curves and the related concentration index, which are developments of the Lorenz Curve and Gini index (Gini, 1912). Initially applied to assess health inequalities (Wagstaff et al., 1991), Walker et al. (2005a) and Su et al. (2009) first demonstrated the use of the concentration index to assess inequalities in the distribution of environmental hazards. Whilst the Gini index is one of the most commonly used metrics for evaluating inequalities (Maguire & Sheriff, 2011), their adaptation as concentration curves/indexes enable inequality assessment in an additional dimension, in this case deprivation.

Concentration curves (Kakwani, 1997; Wagstaff et al., 1991) plot the cumulative distribution of ES across population-weighted deprivation quintiles referenced against a hypothetical perfectly equal distribution. The population weighted deprivation quintiles from most to least deprived are plotted along the x-axis and cumulative percentage of net ES index is plotted on the y-axis. The closer the concentration curves lie to the line of perfect equality (a 45° line, showing each quintile to have a 20% share of ES), the more equal the distribution. If the concentration curve falls below the line of equality, the corresponding deciles have less than an equal proportion of ES (Figure 3.9).

Table 3.6 Summary of statistical techniques used to evaluate inequalities of environmental outcomes within the environmental justice discourse. This is not an exhaustive list but includes the most commonly applied measures. Composed from reviews by Maquire & Sheriff (2011), Harper et al. (2013), and Miao et al. (2015). Example of studies which have applied the techniques are also given.

Type of statistical measure	Examples of measures	Purpose
Visual presentations	Histograms, boxplots, scatterplots	Visual comparison of changes in environmental phenomena across social groups (and vice-versa). e.g. Jenerette et al. (2011), Mitchell & Norman (2012), Escobedo et al. (2015)
Summary statistics	Mean, median, standard deviation	Descriptive statistics which provide initial insights into differences in environmental phenomena across social groups (and vice-versa) e.g. Lakerveld et al. (2015).
Difference tests	t-test, Wilcoxon-Mann-Whitney test, ANOVA	Tests if differences between means/medians are statistically significant. e.g. Barbosa et al. (2007), Abercrombie et al. (2008)
Ratios	Ratio of medians, ration of 90 th percentiles	Identifies differences between highest and lowest socio-economic groups. e.g. Boyce et al. (2016)
Correlation coefficients	Pearson's or spearman rank correlation coefficient	Tests for statistically significant associations between two variables, indicator strength of linear relationship and direction. e.g. Chakraborty et al. (2014)
Regression models	Logit regression, multiple regression, Probit regression	Examines relationships between two or more variables. Inferring the influence of one or more independent variable on a dependent variable. Varied set of models to suit different data types e.g. Pope & Boone (2016), Soto et al. (2016)
Visual ranking tools	Lorenz curve, concentration curve	Graphs show cumulative 'exposure' to an environmental condition (y-axis) against the cumulative percent of population ranked by 'exposure' (Lorenz) or socio-economic variable (concentration). Visualises how equally a phenomena is shared across the population using a hypothetical perfectly equal scenario for comparison. e.g. Su et al. (2009)
Inequality indices	Gini index, concentration index, Atkinson index, Kolm-Pollak index, Thell index	Summary statistics of the degree of inequality experienced. Gini and concentration indexes are numerical summaries of how actual distributions differ from a 'perfectly equal' scenario as shown in their corresponding graphs. The Atkinson index can be modified to become more sensitive to different parts of the distribution (e.g. changes for low socio-economic groups). Kolm-Pollak indicates absolute differences. e.g. Gomes-Lopes et al., 2015; Boyce et al. (2016)

The concentration index ($-1 \leq C \leq 1$), is a summary index which quantifies the degree of inequality. It is represented by double the area between the concentration curve and line of equality (Figure 3.9; Kakwani et al., 1997). $C = 0$ represents an equal distribution (line of equality), positive values of C indicate the concentration curve lies below the line of equality and negative above the line of equality.

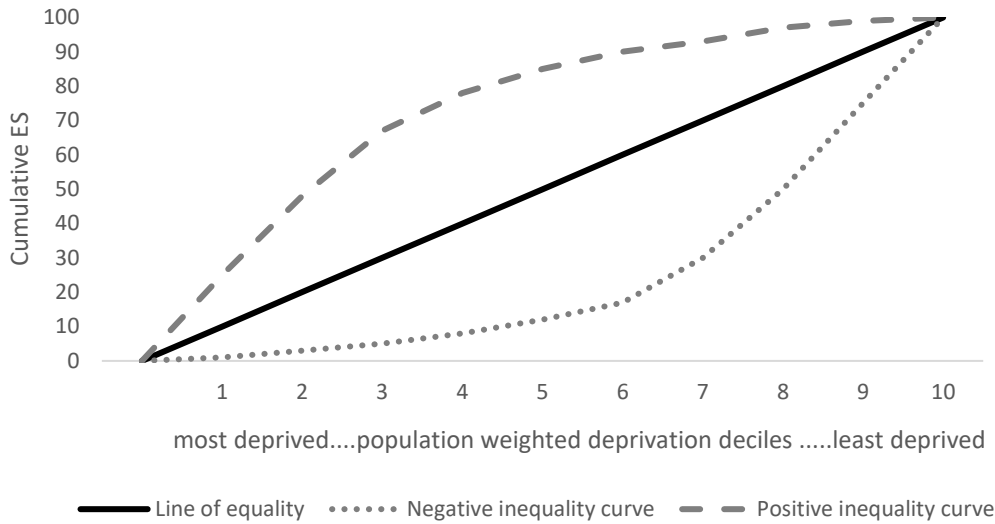


Figure 3.9 Illustrative example of concentration curves. Deciles represent 10% of total population, with decile 1 most deprived following IMD2015. Negative inequality indicates a lower share of ESs for more deprived populations.

However, since ES are beneficial, careful interpretation of the concentration index is required. Typically it has been used to evaluate inequality of environmental hazards, in which case a negative value would indicate a greater burden of hazard for more deprived populations, which would normally be considered undesirable (e.g. Su et al., 2009). For this analysis, a positive value is undesirable, since this represents more deprived areas receiving a lower share of ES.

Since the values of the concentration curve are provided at intervals, the concentration index is approximated by calculating the area of trapezoids underneath the curve, and using this to calculate the ratio $a : b$ (eq. 3.3).

$$C_1 = 1 - 2 \sum_{k=1}^n (X_k - X_{k-1})(Y_k + Y_{k-1}) \dots \dots \dots \text{Equation 3.3}$$

Where:

X_k = cumulative proportion of population for $k = 0, \dots, n$, with $X_0 = 0, X_n = 1$. Note since each quintile has an approximately equal population, X_k will increase by increments of 0.2.

Y_k = cumulative proportion of Net ES for $k = 0, \dots, n$, with $Y_0 = 0, Y_n = 1$

Overall, these measures were selected to examine the distribution of ES across the whole population (i.e. including 'middle income' groups), which is considered important since we know very little regarding the distribution of ES. For example, if ratios were used, the differences between the most and least deprived areas would be assessed, but may overlook important differences between the most, least and 'middle' deprived areas. This limitation was illustrated by Brunt et al. (2017), who found that air pollution was actually highest for both the least and most deprived areas; using only ratios would not uncover this pattern.

The statistical techniques described thus far do not account for potential spatial autocorrelation (Fotheringham et al., 2000; Cushing et al., 2015). That is, observations adjacent in space are more likely to be similar, as described by Tobler's first law of Geography "everything is related to everything else, but near things are more related than distant things" (Tobler, 1970, 236). A spatially explicit approach will uncover the spatial variation in any relationship between ES and deprivation. The method I adopt is the Local Indicators of Spatial Association (LISA) (Anselin, 1995). This is a bivariate technique, extended from the computation of Moran's I which can identify hotspots, coldspots and where there is no significant relationship between variables. This can also help inform whether particular areas have a greater influence on the global statistics (Anselin, 1995).

The measures of inequality are computed in 'r' and SPSS for descriptive statistics and correlations, Microsoft Excel is used to generate the concentration curve and index, finally GeoDa is used to generate maps of local associations.

3.4 Research design and methodology: A summary

This chapter has introduced the conceptual basis for the research, illustrating the need for disaggregating the benefits gained from natural capital and ESs, which can enable assessment of inequalities. This conveys how knowledge of the social distribution can become part of the feedback mechanisms which drive change in natural capital management and therefore changes the distribution of that natural capital and the ESs it generates. Additionally this chapter has outlined the spatially explicit models required to map the distribution of ESs. This entails consideration of where the supply of ESs is generated relative to where the beneficiaries of the services are. The complexity of the spatial interactions limits the potential for mapping multiple ESs nationally, and therefore a multiscale approach to the research design is taken. This combines a coarser assessment of associations between natural capital and deprivation nationally (objective 1 – Chapter 4) with detailed analysis of ESs' social distribution (objective 2 – Chapters 5-7) carried out for case study regions.

The conceptual basis for this research has informed the research methodology, which the subsequent sections of this chapter have been dedicated to. This has provided an overview of the methods relevant to multiple aspects of the analysis including details of which natural capital and ESs are assessed, a summary of the approach to modelling ES which is spatially explicit, the approach to sensitivity testing corresponding to objective 3 and an account of how inequality is assessed. The environmental data and the specific methods used to map natural capital and ES differ for objectives 1 and 2, and for each ES within objective 2, they are therefore addressed in their corresponding chapters. The next chapter presents the methods and results from analysis of the national distribution of natural capital. This forms a comprehensive overview which provides context for and informs the selection of case studies for fulfilling objective 2.

Chapter 4 A national analysis of deprivation and natural capital

Distributive analysis of natural capital provides an essential first step in understanding how that natural capital may be equitably managed within an ecosystems approach, including informing development of spatial strategies that address social, economic and ecological challenges. The aim of this chapter is to present analysis which addresses the first research objective; examining the social distribution of natural capital assets which underlie the provision of ES, at a coarse scale but with national coverage. As outlined in Chapter 3, this analysis is part of a multi-scale approach which provides a national perspective but also provides the context for more detailed case study analysis presented in subsequent chapters which examine the distribution of flows of ES from natural capital. This chapter corresponds to a peer-reviewed paper in *Landscape and Urban Planning* (Mullin et al., 2018).

Section 4.1.1 provides a brief overview of the status of natural capital across England, and building on Chapter 3 section 3.3.1, section 4.1.2 describes the mapping of indicators of natural capital. The Index of Multiple Deprivation is used to indicate socio-economic conditions - the choice of this indicator was discussed in greater detail in Chapter 3 section 3.3.2.1. Section 4.1.3 further explains the use of IMD at the coarser district scale, as different IMD measures are available at this scale. Section 4.2 details the computation of natural capital indicators from the input datasets and the spatial clustering methods which are used to aggregate these to district level. Section 4.3 presents the results with section 4.3.3 illustrating how natural capital varies by deprivation, and the final sections 4.4, 4.5 and 4.6 consider implications of this analysis, including a review of methods and implications for environmental equity and planning and land management nationally. These discussion are extended in Chapter 8, following the regional case studies presented in Chapters 5-7.

4.1 Study area and data

4.1.1 England's natural environment

Key features of England's landscape are summarised in Table 4.1. State of the environment reporting (UKNEA, 2011; ONS, 2015b) reveals a mixed picture with indicators variously revealing improving status (e.g. surface water abstraction, surface water quality, greenhouse gas emission, use of construction materials), little change (use of non-construction materials, sea- and wetland birds), or continued decline (forest and farmland birds). A general trend for an increase in cultural and regulating ESs and some decreases in provisioning services has been observed from 1993-2012 across 9 UK monitoring sites (Dick et al., 2016). Rising consumption, demographic change and climate change are the principal forces placing natural capital at further risk (NCC, 2013).

Table 4.1 Characteristics of England's landscape

Type	Extent*	Observations
Urban	9% of land cover	81.2 % of the population live in urban areas
Agricultural	70% of land cover	Most is privately owned
Woodland	9% of land cover	One of the lowest afforestation rates in Europe
Wetland	4% of land cover	Almost half are protected
Upland	5% of land cover	
Rivers and streams	136,000 km	
Canals	2,600 km	
Lakes and reservoirs	5700 (number)	In addition to an extensive coastline

* England covers approximately 130,000 km². Percentages are approximate. Source: UKNEA (2011), DCLG (2013)

4.1.2 Natural capital datasets

As defined in Chapter 1, natural capital are considered as the natural assets which produce benefits or are of value to people or the stock of non-renewable and renewable natural resources. In Chapter 3 section 3.2.1, it is specified that given the purposes of our study, the assets of interest are only those which are ecosystem based, giving rise to ES. Specifically, this includes species, ecological communities, soils, freshwaters, land, natural processes and function (following Mace et al., 2015; NCC, 2015).

Natural capital indicators have largely been developed for monetary valuation (e.g. , Costanza et al., 1997; ONS, 2015a) or for assessing its criticality (e.g. De Groot et al., 2003; Ekins et al., 2003; Mace et al., 2015). Discussion in Chapter 3 highlighted that for the purposes of quantifying how natural capital is distributed socially, we require spatially disaggregated, objective and relative measures of natural capital but a monetary value is not required. Nevertheless, existing approaches applied to determine monetary values, or the risks and thresholds of natural capital, are utilised to inform the selection of indicators used in this analysis (as listed in Table 4.2).

The first steps of 'Principals for Natural Capital Accounting' established in the UK by the ONS (ONS, 2017) require assessment of natural capital stock extent and condition. For creation of a risk register, Mace et al. (2015) add a requirement for knowledge of 'spatial configuration', whilst De Groot et al. (2003) consider the condition of the assets including naturalness, biodiversity, uniqueness, fragility, value for supporting life and renewability as 'critical' factors. On this basis, indicators of both the extent and quality (condition) of natural capital were sought; these are understood to be important for the supply of multiple ESs. Spatial configuration, natural processes and functions are not incorporated as these are specific to individual ESs and the flows of goods and services from the asset to the beneficiaries (Andersson et al., 2015). As highlighted previously, these cannot be reliably modelled for the full range of services nationally. However, inclusion of the spatial extent of built-up areas is used to provide context, and this also provides some indication of the number of potential beneficiaries of or actors in the production of local services.

'UK broad habitats' are used to categorise different types of natural capital, an approach consistent with Mace et al. (2015) and the UKNEA (2011), see also Chapter 3 section 3.2.1. Data on the extent of each broad habitat are available from a single source (Land Cover Map of Great Britain, LCM 2007). Data for freshwater extents is obtained using Ordnance Survey vector data since rivers are the dominant freshwater features in many areas and these are not well identified by the LCM which has a 25m spatial resolution. 'Urban' and 'suburban' land cover are included because, whilst these do not describe natural capital directly, they provide an important context, aiding the interpretation of results, and are strongly associated with private residential garden area (Generalised Land Use Database, 2006) and also population density, a key factor in ES demand.

Table 4.2 Natural capital indicators. ¹ Extents of natural capital follow UK Broad Habitats (Jackson, 2000)²As defined by MA (2005) and UKNEA (2011). ³LCM 2007 – Land Cover Map of Great Britain 2007.

	Natural Capital	Natural Capital Indicator ¹	Relevant ecosystem service(s) ²	Data source(s)
Indicators of quantity	Land	Broadleaved and coniferous woodland (% total area covered)	Fuel, fibre, climate regulation, hazard regulation, air, soil & water purification, noise regulation, aesthetic and education.	LCM 2007 ³ (www.ceh.ac.uk)
		Wetland and coastal land (% total area covered)	Hazard regulation, water purification, climate regulation, recreation, aesthetic and education.	LCM 2007 (www.ceh.ac.uk)
		Low density built-up land (% total area covered by suburban land cover class)	Recreation, noise regulation, hazard regulation.	LCM 2007 (www.ceh.ac.uk)
		High density built-up land (% total area covered by urban land cover class)	-	LCM 2007 (www.ceh.ac.uk)
		Semi- natural grassland (% total area covered)	Soil & water purification, recreation, aesthetic, hazard regulation.	LCM 2007 (www.ceh.ac.uk)
		Agricultural land (% total area covered by enclosed farmland broad habitat)	Food, fuel, hazard regulation, aesthetic.	LCM 2007 (www.ceh.ac.uk)
	Mountain (% total area covered)	Climate regulation, food, water purification, hazard regulation, aesthetic.	LCM 2007 (www.ceh.ac.uk)	
Freshwater	Freshwater (length of river or lake shoreline per km ²)	Freshwater, food, recreation, aesthetic.	OS OpenRivers, OS Meridian 2015 (www.ordnancesurvey.co.uk)	

	Natural Capital	Natural Capital Indicator ¹	Relevant ecosystem service(s) ²	Data source(s)
Indicators of quality	Land	Land with protected status (% total area covered)	Wild species diversity, pollination, educational.	Boundary datasets, all data regularly updated & downloaded 2015 - 'Local Nature Reserves', 'National Nature Reserves', 'Ramsar Sites', 'Sites of Special Scientific Interest', 'Special Areas of Conservation', 'Special Protection Areas' (www.gis.naturalengland.org.uk) 'RSPB UK reserves 2015' (www.rspb.org.uk)
		Publically accessible spaces (% total area covered)	Recreation, education, cultural heritage.	Boundary datasets: 'Country parks', 'CROW 4, 16, 15, access', 'Doorstep greens', 'Local Nature Reserves', 'National Nature Reserves', 'Millennium Greens', 'Registered parks and gardens'. All datasets downloaded 2015. (Natural England; gis.naturalengland.org.uk). Accessible woodland dataset (The Woodland Trust, BG v.11, 2014 enquiries @woodlandtrust.org.uk)
		Agricultural land quality (% agricultural land modelled as good to excellent – grades 1 & 2)	Food	Agricultural land classification 2013 (gis.naturalengland.org.uk)
	Land, soil	Density of carbon in topsoil (area weighted mean of carbon density tha^{-1})	Climate regulation.	Soil carbon natural capital map 2007 (catalogue.ceh.ac.uk)
	Ecological communities and species	Ecological status (area weighted mean of ecological status)	Wild species diversity, pollination.	Ecological status grid 2014 - a modelled biodiversity index adjusted for recorder effort and environmental zone (catalogue.ceh.ac.uk)
	Freshwater	Water quality (% total length of waterbodies with good or high overall status)	Freshwater, recreation, food, aesthetic.	Water Framework Directive waterbodies 2013 status (www.geostore.com/environment-agency)

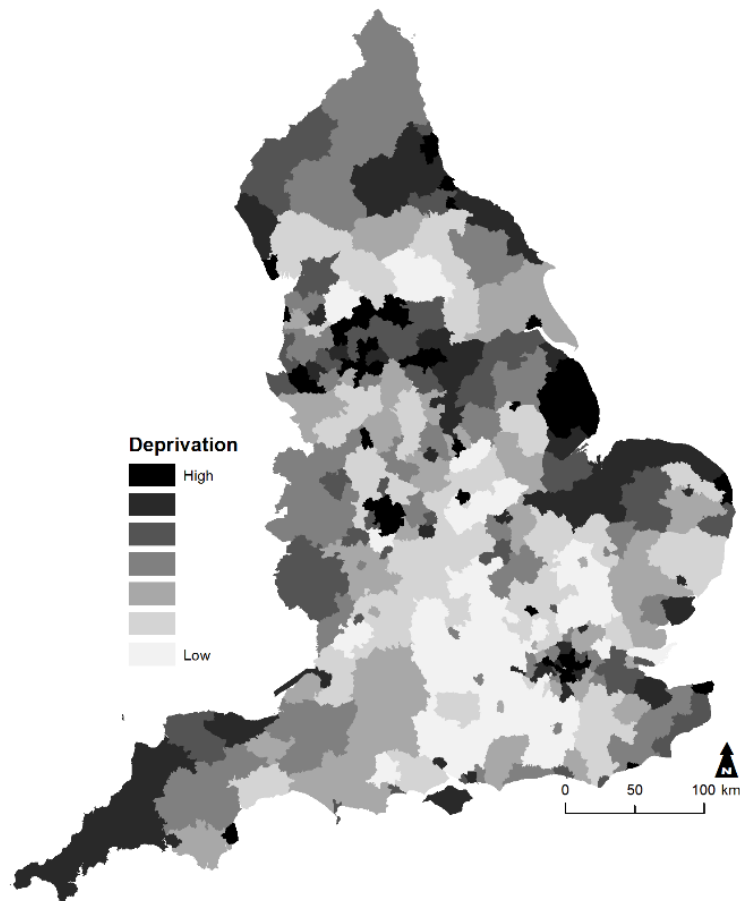
Ideally indicators would be used which describe the quality of all types of broad habitat and which are relevant for a range of ES including provisioning, cultural, regulating and supporting services. From a review of sources, some datasets were found to be explicit measures of the quality of specific types of natural capital e.g. water quality, quality of agricultural land and topsoil carbon. Although some of these potential data were omitted for reasons of parsimony. For example, multiple datasets are available which indicate soil quality (available from: eip.ceh.ac.uk/naturalengland-ncmaps) but once aggregated to districts they correlate closely with soil carbon ($\rho_{\min} = 0.7$, $\rho_{\max} = 0.956$). Of these, soil carbon was selected for use since this is also relevant for climate regulation. Other quality indicators are applicable across multiple broad habitats and are relevant to conditions defined by De Groot et al. (2003); Ecological status provides insight into the level of biodiversity of each type of natural capital (relative to that which is expected). Coverage of protected areas e.g. Sites of Special Scientific Interest, whilst a designation for management purposes, by definition relates to naturalness, biodiversity, uniqueness and/or fragility. Coverage of publicly accessible land is included as this is an important condition for several cultural ESs.

4.1.3 District-level deprivation data

As set out in Chapter 3, Local Authority Districts are the spatial units used for this analysis as the key unit of land use planning and which largely coincide with areas defined for ecological management and economic growth. Deprivation is determined from the relative IMD ranks which are calculated by the Office for National Statistics initially for Lower Super Output Areas (LSOA, areas defined to contain a mean population of 1500). From this, various IMD metrics are calculated for the 325 English Local Authority Districts (LAD) (Table 4.3). These measures reflect LAD deprivation in terms of a whole district average, and the amount and severity of deprivation relative to other LADs. A district average may hide within district extremes, and thus use of the two additional measures ensures within district variation in deprivation is better accounted for. This is especially relevant since there are some notable changes in district deprivation ranks dependent on the measure used, only six districts are ranked in the 20 most deprived districts for all three measures. For each measure LADs were ranked (low ranks represent high deprivation) then grouped into deciles of equal population for subsequent analysis.

Table 4.3 District level representation of IMD data

IMD measure	Description
Average rank	A population weighted mean of all LSOA ranks within the LAD. A whole area measure, but neglects within LAD variability.
Extent	Percentage of LAD population in the 30% most deprived LSOAs nationally. Within the 30%, progressively more weight is given to the more deprived LSOAs.
Local Concentration	Average rank of the most deprived LSOAs within which 10% of the population of the district live. Measure focuses on most severe deprivation.

**Figure 4.1** Deprivation in English Local Authority Districts (IMD average ranks). ONS (2015)

4.2 Methods

Having sourced relevant data, indicators were computed for LADs with subsequent analysis comprising of comparison of deprivation against individual natural capital indicators, then with an aggregation of those indicators. ESRI ArcGIS for Desktop 10.3.1 and QGIS were used, with statistical analysis and aggregation of natural capital indicators executed using IBM SPSS Statistics 22 and R software.

4.2.1 Computation of natural capital indicators

Computation of natural capital indicators at the LAD level required a range of geospatial processing steps dependent on the resolution and format of the input data. Given the variable size of LADs, indicators had to be made comparable; where possible percentages were computed, otherwise per unit area and area-weighted means were calculated. The extents of each type of natural capital (broad habitats), except freshwater, within each LAD were calculated using zonal statistics tools. The length of rivers and lake shorelines were calculated using a sum line length tool.

To generate values for quality indicators, different calculations were required to handle the different data types. In general, there were three approaches; for publicly accessible areas and land with protected status, the extent was given as a percentage of total LAD area. These areas were defined by multiple independent datasets which were merged before total extent was computed. For quality of water and agricultural land, the features classified as the highest and second highest quality were extracted and their extent relative to total classified waterbodies/agricultural land within each LAD were computed as a percentage. Soil carbon and ecological status are provided as continuous data values and therefore the area-weighted means were calculated.

4.2.2 Aggregate natural capital

The approach for aggregating the natural capital indicators for equity analysis should identify the relative similarities and differences of aggregate natural capital. Since the units of the indicators vary and placing them on a single scale requires further value judgements, a simple additive method was avoided. Instead, area classification using clustering methods was implemented, providing a comprehensive, quantitative and spatial

summary of natural capital which is less driven by value judgements, although such judgements cannot be wholly avoided. Clustering is a well-known geographical technique used to describe areas in terms of key characteristics, described by the mean value of each indicator, thus unlike aggregation to a single index, clustering allows information about component characteristics to be simultaneously conveyed.

Clustering techniques have been used previously to assess bundles of ESs (e.g. Hamann et al., 2015; Raudsepp-Hearne et al., 2010; Turner et al., 2014). Hamann et al. (2016) further demonstrate how creating clusters is a useful way to facilitate the comparison of ES bundles to social data. Whilst in the context of social equity and environmental benefits, several studies use clustering to explore who benefits from urban greenspace (Barbosa et al., 2007; Ibes, 2015 and Xiao et al., 2017).

Prior to applying a clustering algorithm, several techniques used within these studies were adopted to facilitate a more in-depth understanding of the spatial patterns and multivariate relationships of the natural capital. Firstly, the Moran's I global statistic was calculated for each indicator using a 150km threshold distance to test the degree to which they are spatially clustered and secondly (Appendix A1), pairwise relationships using Spearman Rank correlation were determined.

Clustering is sensitive to the algorithm applied, so to address some limitations of the different clustering methods, a two-step clustering approach was taken (Figure 4.2), similar to Green et al. (2014). Specifically, a hierarchical algorithm established the initial parameters and likely optimal number of clusters for subsequent k-means clustering. This reduces sensitivity to outliers and ensures replicability (Krieger & Green, 1999) but retains the advantage of the iterative k-means clustering algorithm. The k-means algorithm was repeatedly applied for between 2 and 15 clusters, from which the optimal solution was selected. Criteria for an optimal number of clusters includes relatively equal sized clusters, compact clusters, maximum separation between clusters and a stable solution (Green et al. 2014). Various cluster number optimisation metrics exist (see review by Halkidi et al. 2001), of which 30 were accessed in the NBClust Package in R (Charrad et al., 2014). Beyond these metrics, the number of clusters selected should be based on results that are sensible, interpretable and resolved to an appropriate level of detail (Green et al. 2014).

There was no single optimal clustering of England's natural capital. Metrics computed within NBClust most commonly indicate only three clusters is optimal, but this primarily identifies an urban/rural divide and the natural capital classification must reveal greater detail. The peaks in the NBClust plots for Hubert's Statistics (cluster compactness) and D Index (cluster homogeneity) (Figure 4.a and 4b) indicate stronger clustering performance for the corresponding number of clusters. The highest peak corresponds to 3 clusters, subsequent to this 5 or 6 clusters perform best. ANOVA showed that 14 clusters are required to obtain lower *within cluster* variation than *between cluster* variation for all indicators. However, in practice, such a high number of clusters could not all be clearly differentiated and interpreted. Variation in cluster size was also lowest with a greater number of clusters, however, this also creates multiple small clusters. The variation was greatest when less than 5 clusters are created.

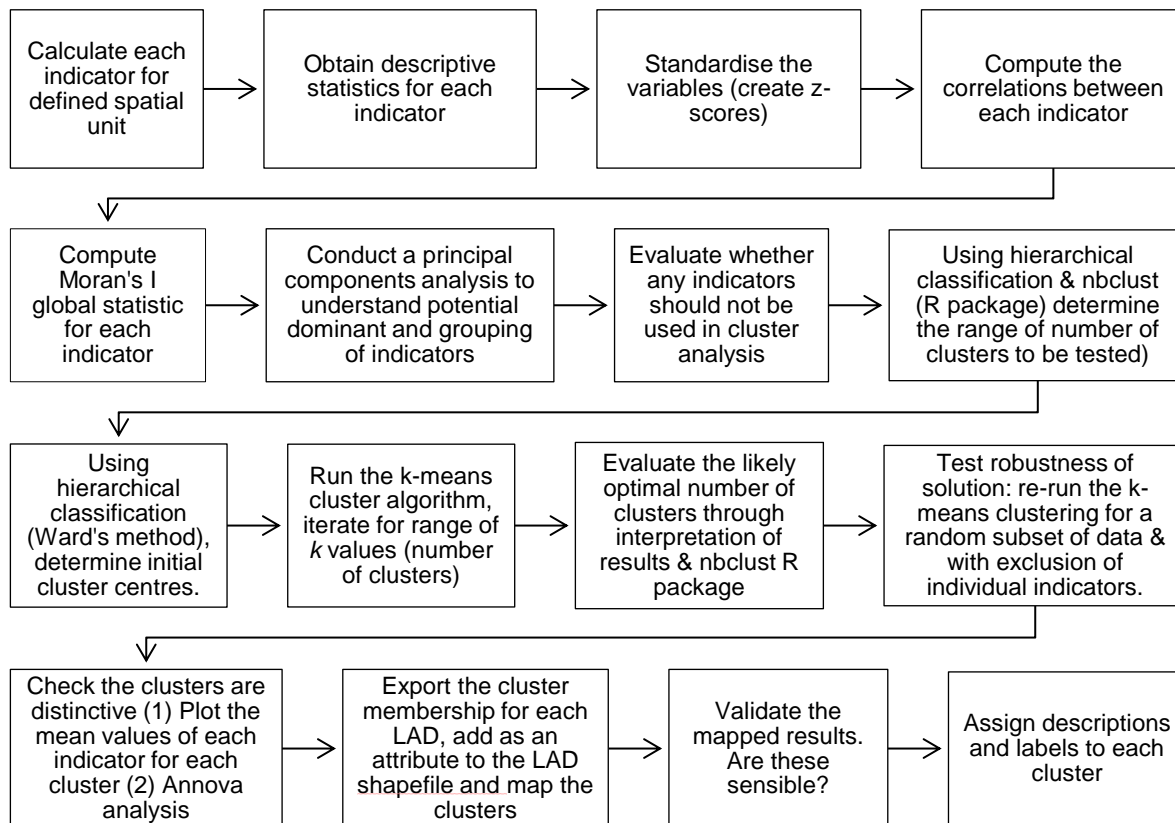
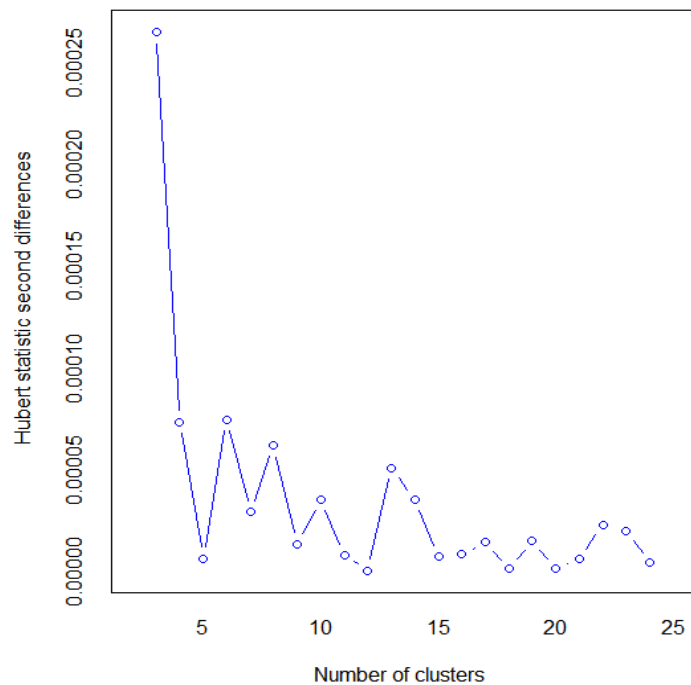


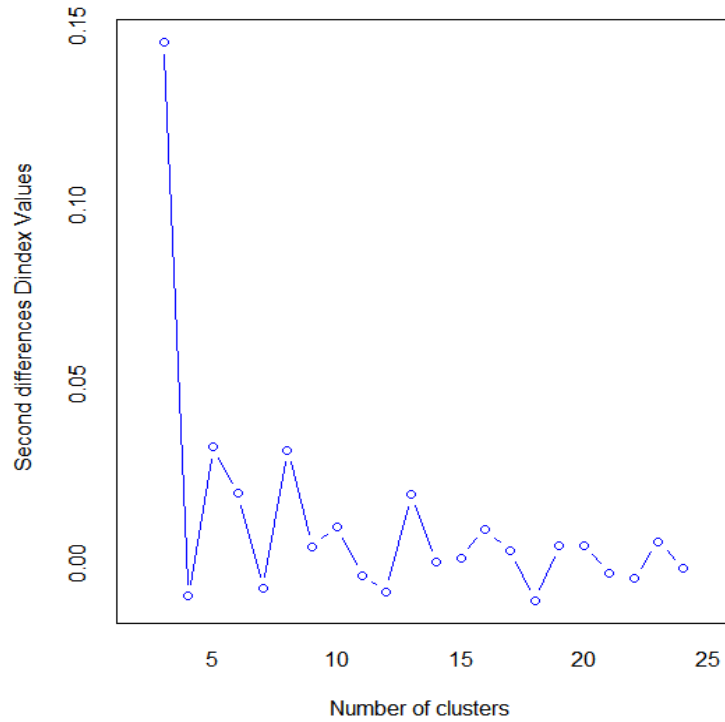
Figure 4.2 Natural capital classification of LADs using hierarchical and k-means clustering techniques

A final classification of six clusters was selected as this provides a clear classification that reveals an appropriate level of detail with acceptable clustering performance. Six clusters have a more evenly distributed cluster size (Figure 4.c), can be easily interpreted, and perform well in the Hubert's Statistics and D Index plots. Critically, the six clusters are meaningful, retaining sufficient distinction between clusters for descriptions to be assigned to each (Figure 4.), but are not overly complex. The clustering results were replicated for subsets of the data, thus indicating the classification is robust. For comparison, results for 5 and 7 clusters are given in Appendix A2.

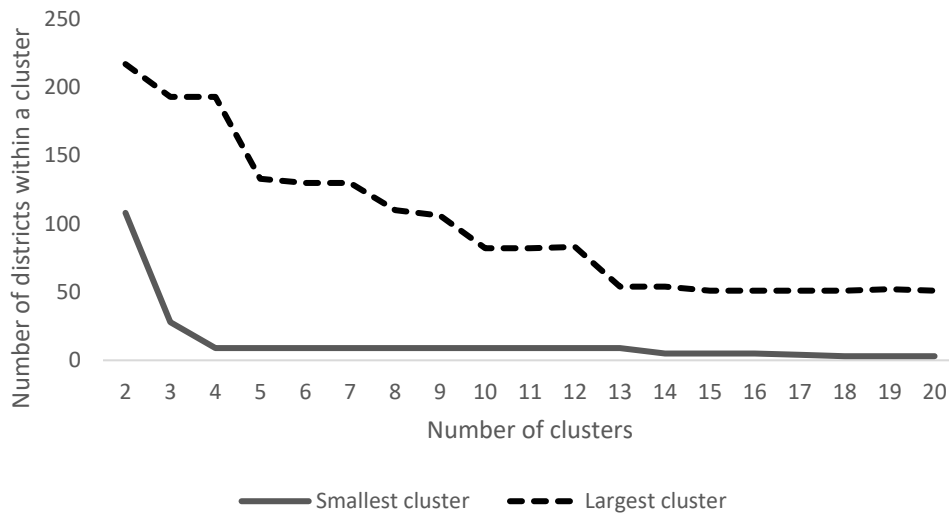


- a) Hubert Statistic (Hubert & Arabie, 1985) second difference plot. Indicates cluster compactness with increasing number of clusters

Figure 4.3 Graphical indicators of optimal number of clusters using k-means clustering. The first significant peak on the second difference plots indicates the statistical optimal solution (Charrad et al. 2014).



- b) D-Index (Lebart et al., 2000) second difference plot. Indicates 'cluster gain' which is change in homogeneity relative to increasing number of clusters



- c) Cluster size by cluster frequency.

Figure 4.3 Cont..Graphical indicators of optimal number of clusters using k-means clustering. The first significant peak on the second difference plots indicates the statistical optimal solution (Charrad et al. 2014).

The distinct natural capital profile of each cluster identifies which districts have a high or low extent or quality of each type of natural capital. The variation of deprivation with respect to natural capital was explored using the mean values of each IMD measure for the districts assigned to each cluster, a comparison of which clusters contain the most and least deprived districts, and visualisation of the distribution of IMD ranks in each cluster using boxplots. Kruskal-Wallis tests were applied to determine whether differences in deprivation between clusters are statistically significant. Sensitivity testing was carried out in which the clustering was re-run with each indicator removed (Appendix A3).

4.3 Results: The social distribution of natural capital in England

4.3.1 Natural capital indicators' Moran's I and pairwise correlations

All natural capital indicators exhibited significant spatial clustering (Appendix A1, with clustering greatest at 50km). High-density built-up areas are the most tightly clustered, whilst semi-natural grassland, soil carbon ecological status, agricultural land and quality of water have greater clustering tendency than the other indicators.

Pairwise correlations are listed in Table 4.4. Each natural capital indicator exhibits significant moderate correlation ($\rho < -0.4$, $\rho > 0.4$) with at least one other indicator (wetland/coastal and agricultural land quality) and with as many as eight (soil carbon). Independent variables are not a prerequisite for clustering but highly correlated variables may add 'weight' to a particular division. Whilst soil carbon is closely negatively correlated with built up areas ($\rho = -0.802$ and $\rho = -0.855$ for low density and high density respectively) this indicator is retained for clustering since it indicates quality rather than quantity and it emphasises differences beyond the rural/urban divide. Low and high density built-up areas, ecological status and agricultural land quality have predominantly negative correlations with other indicators, suggesting that these may form clusters distinct from areas with high levels of the other natural capital indicators.

Table 4.4 Spearman Rank correlation coefficients between natural capital indicators

	Mountain	Coastal	Woodland	Ecological Status	Agricultural land	Quality of agricultural land	Fresh water	Water quality	Publically accessible	Protected status	Semi-Natural Grassland	Soil carbon	Low-density built up	High density built up
Mountain	1	-	-	-	-	-	-	-	-	-	-	-	-	-
Coastal	-0.053	1	-	-	-	-	-	-	-	-	-	-	-	-
Woodland	.305**	-0.095	1	-	-	-	-	-	-	-	-	-	-	-
Ecological Status	-.223**	-.209**	-0.025	1	-	-	-	-	-	-	-	-	-	-
Agricultural land	.141*	-0.022	.162**	-.467**	1	-	-	-	-	-	-	-	-	-
Quality of agricultural land	-.152**	.242**	-0.093	-.149**	.421**	1	-	-	-	-	-	-	-	-
Freshwater	.439**	-.230**	.137*	-0.089	0.083	-.110*	1	-	-	-	-	-	-	-
Water quality	.461**	-0.074	.123*	-.379**	.309**	0.015	.374**	1	-	-	-	-	-	-
Publically accessible	.332**	0.004	.514**	0.042	-.330**	-.317**	0.105	0.078	1	-	-	-	-	-
Protected status	.165**	.532**	.148**	-0.048	-.271**	-0.032	-0.032	0.03	.496**	1	-	-	-	-
Semi-Natural Grassland	0.669**	0.076	0.276**	-0.454**	0.349**	-0.129*	0.389**	0.451*	0.273**	0.129**	1	-	-	-
Soil carbon	.531**	-0.023	.524**	-.445**	.642**	0.004	.306**	.449**	.240**	0.098	0.707**	1	-	-
Low-density built up	-.305**	-.129*	-.243**	.555**	-.745**	-.172**	-.124*	-.417**	-0.018	-0.093	-0.536**	-.802**	1	-
High density built up	-.333**	-0.036	-.411**	.469**	-.781**	-.189**	-.155**	-.425**	-0.046	-0.045	-0.536**	-.855**	.860**	1

Significance levels **p<0.05 ***p<0.01

4.3.2 Clustering

Characteristics of the six clusters generated are described in Table 4.5. The descriptions are based on the z-scores of the natural capital indicators for each cluster as shown in Figure 4.. These are generalisations for the clusters overall and there will be some variation from these descriptions within clusters with respect to the individual LADs. **Error! Reference source not found.** maps the six natural capital clusters and thus indicates the spatial distribution of natural capital nationally.

Table 4.5 Cluster descriptions based on average values of natural capital indicators across all LADs within each cluster. Clusters have been assigned simple labels which correspond to the most dominant natural capital indicator.

Cluster	Number of LADs	Key characteristics
1. Urban	13	Dominated by high density built-up land but also characterised by the highest ecological status. Lowest extent and quality of natural capital with respect to all individual indicators used except the extent of publically accessible land.
2. Suburban	90	Dominated by lower density built-up land. Further characterised by above national average of high density built-up land and ecological status. Low coverage of agricultural land and soil carbon.
3. Mountain	28	Dominated by montane broad habitats and semi-natural grassland. This cluster also features the highest extent of publically accessible land, extent of freshwater, large extents of land with protected status and highest water quality and soil carbon. Ecological status is lowest in this cluster.
4. Coast	9	Dominated by coastal habitats and land with protected status. Agricultural land is of the highest quality in this cluster but the extent of agricultural land is low. Soil carbon and freshwater extent and quality are also low.
5. Agriculture	138	Dominated by agricultural land, with high quality of agricultural land and water. Largely rural with the lowest extent of built up land but also the lowest extent of publically accessible land.
6. Woodland	47	Dominated by woodland with low extent of built up land. High extents of publically accessible land.

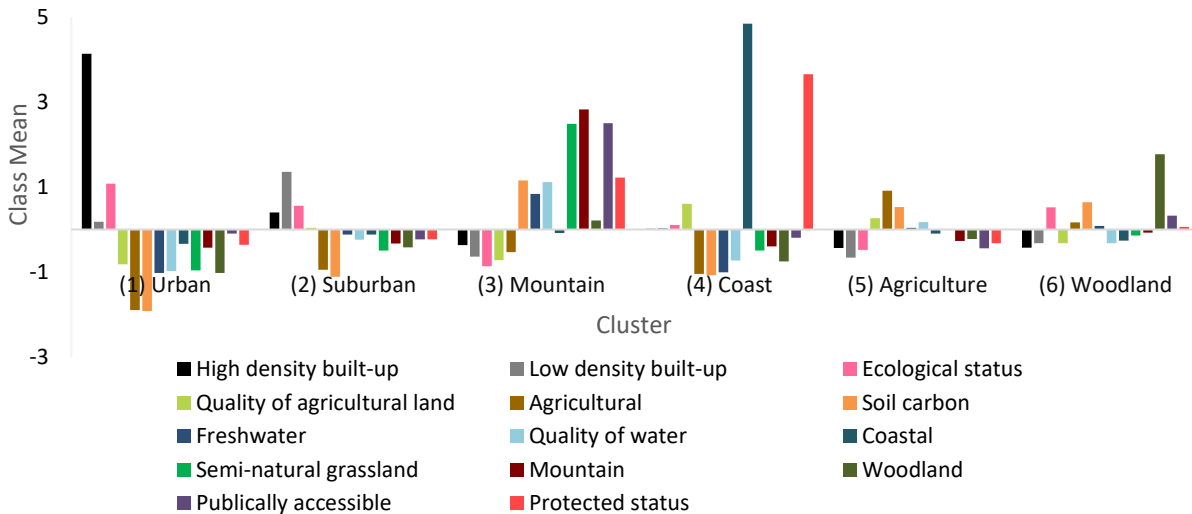


Figure 4.4 Mean z-scores for each indicator by cluster. A z-score of zero represents the national average, whilst a score of +1 is one standard deviation above the average. Mean values of natural capital indicator z-scores for each of the six clusters describe the characteristics of each cluster.

Clusters 1 ('urban') and 2 ('suburban') are characterised by higher proportions of built-up land and lower natural capital. The 'urban' cluster with the greatest extent of high density built-up land overall exhibits the lowest extent and quality of natural capital, but notably has the highest ecological status. Few LADs are assigned to the 'urban' cluster and these are spatially concentrated in central London only. The 'suburban' cluster has the highest extent of low density and above average high density built-up land, the type and extent of the majority of natural capital indicators is below average, except for ecological status. The second highest number of LADs are assigned to this cluster, and are they located in major towns and cities across England.

Clusters 3 ('mountain') and 4 ('coast') have the highest extent and quality of natural capital. The rural 'mountain' cluster exhibits the highest extent of mountain, freshwater, semi-natural grassland, land that is publicly accessible and has protected status. It also has the highest levels of soil carbon and water quality but the lowest ecological status. Although only 28 LADs are in this cluster, their rural character means they cover a large area, predominantly in northern England. The 'coast' cluster is characterised by the highest area of coastal habitat and land with protected status and whilst it has the highest quality of agricultural land, it has a low extent of agricultural land. This is the smallest

cluster which is scattered spatially in the northwest and southeast. The extents of built-up areas and ecological status are slightly above average for these areas. Clusters 5 and 6 are characterised by the highest extent of agriculture and woodland respectively. The 'agriculture' cluster, is the most commonly assigned cluster, predominantly rural it also has above average quality of agricultural land, water and soil carbon. The 'woodland' cluster, also rural, is further characterised by above average extent of freshwater, mountains, ecological status and soil carbon.

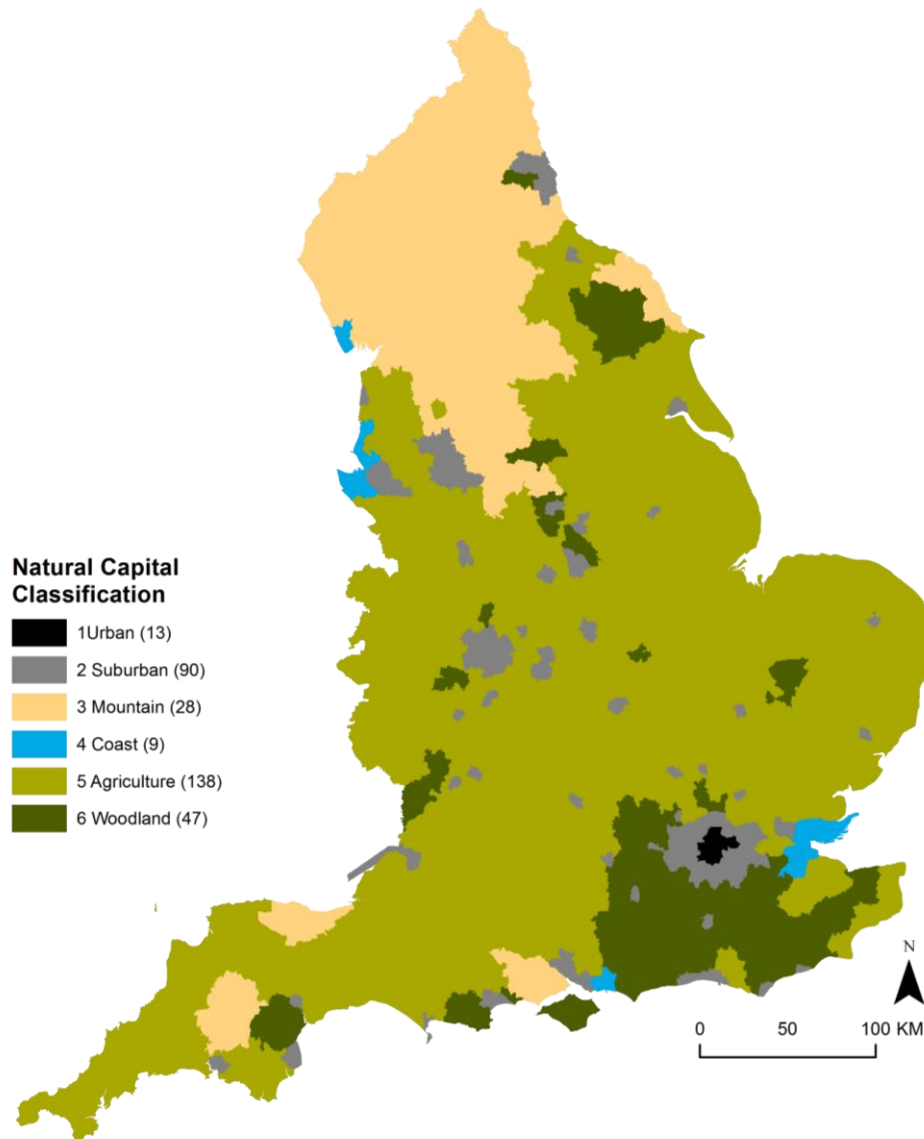


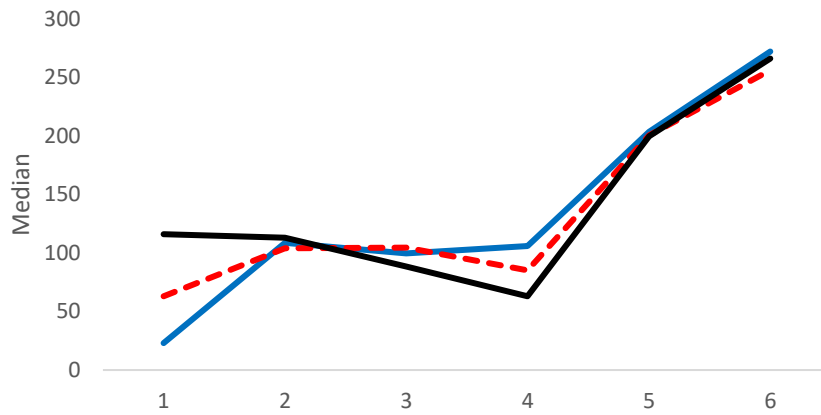
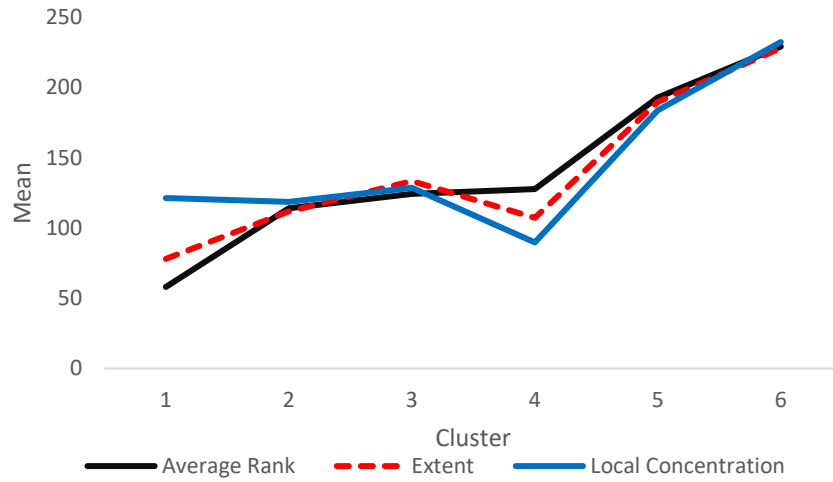
Figure 4.5 Natural capital clusters in England (see text). Parenthesis denote number of LADs within each cluster.

Systematic removal of individual indicators to test the sensitivity to indicator selection reveals overall a robust clustering (see Appendix A3, although some sensitivity was noted to the exclusion of high and low density built up land (resulting in 19% and 15% LADs changing cluster membership, respectively, compared to a maximum of 5% changing with the exclusion of other indicators).

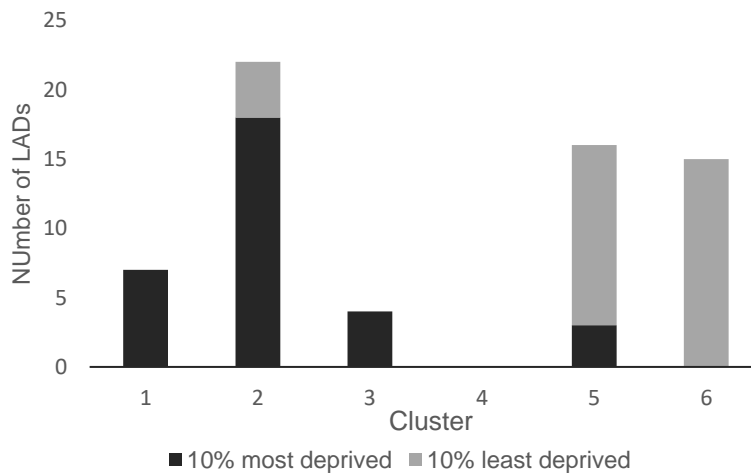
4.3.3 Deprivation and natural capital

IMD values for districts in each cluster are shown in Figure 4.7 (higher ranks indicate lower deprivation). In general, districts of lowest deprivation are located predominantly in southeast England, on the periphery of London and beyond. The most deprived districts are primarily, but not solely, in central London, Birmingham and the Northwest (Figure 4.1).

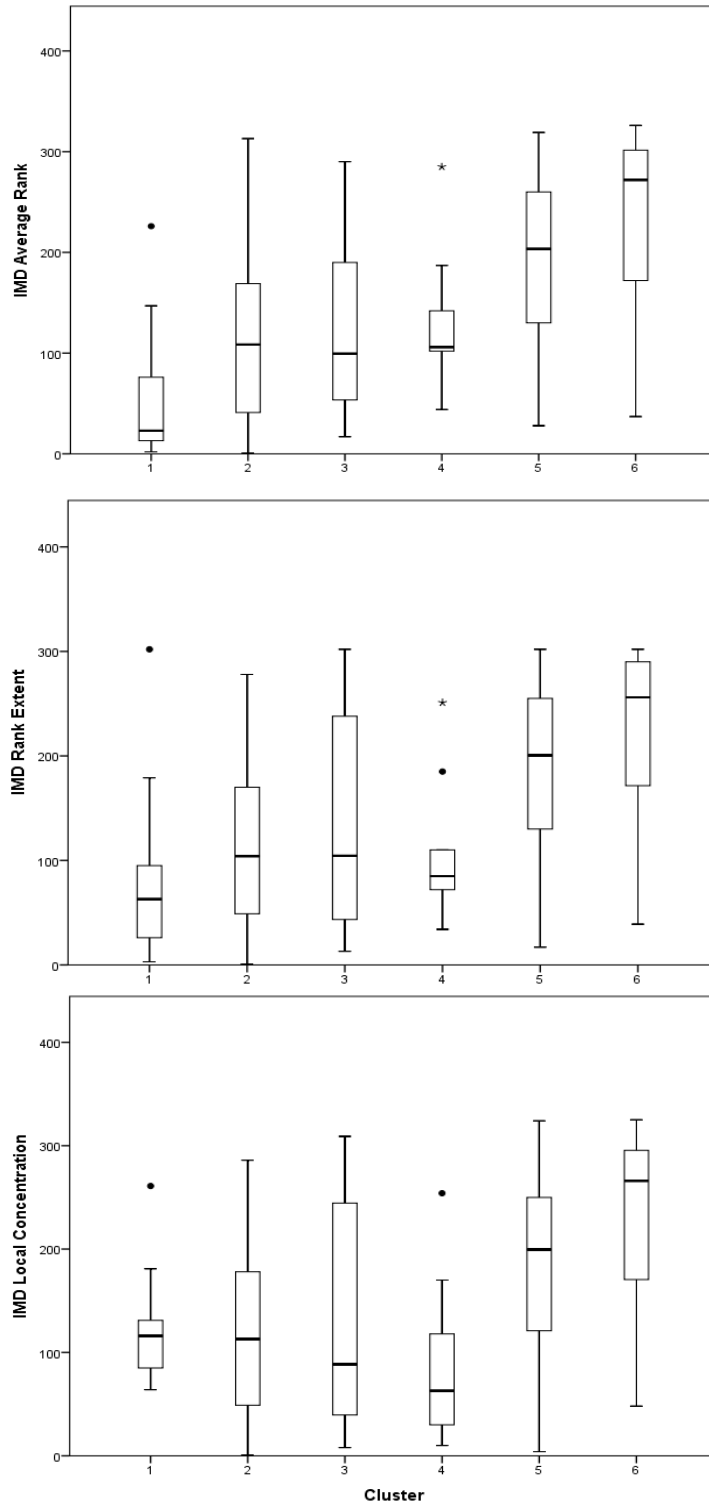
Prior to comparison of deprivation across natural capital clusters, boxplots were used to explore changes in deprivation against individual natural capital indicators. These illustrated that there is a clear decline in high density built-up area with declining deprivation reflecting a known predominance of high deprivation within urban areas relative to rural areas (ONS, 2009). However, none of the natural capital indicators show a similarly distinct social distribution, but differences in natural capital between most and least deprived districts exist. The most deprived districts have the lowest extents of woodland, agricultural land, semi-natural grassland and soil carbon. Conversely, the least deprived districts have the highest extent of woodland, whilst the highest soil carbon and extent of agricultural land is in the relatively un-deprived 2nd least deprived districts. Districts with mid-levels of deprivation tend to have higher quality natural capital compared to the most and least deprived deciles. Significant differences in median values for each IMD measure and the distribution of IMD values (Kruskal Wallis 1-Way Anova) across clusters are observed. Differences are consistently found to be significant under sensitivity testing whereby single indicators were systematically removed from the clustering process.



a) Average IMD values for each cluster



b) Number of the most and least deprived LADs assigned to each cluster



c) Boxplots of the values of each IMD measure for the 6 clusters.

Figure 4.4 Variation in IMD across clusters.

'Urban' districts are on average the most deprived and rural 'agriculture' and 'woodland' districts the least deprived. 78% of the 10% most deprived districts are assigned to 'urban' and 'suburban' clusters compared to 88% of least deprived districts assigned to the 'agriculture' and 'woodland'. None of the 30% least deprived districts are assigned to cluster 1 and only 5 (5%) of the 30% most deprived districts are assigned to the woodland cluster. The mean ranks (higher ranks indicate lower deprivation) of the three IMD measures - average rank, extent and local concentration - are consistently lowest for the 'agriculture' and 'woodland' clusters. The 'urban' cluster has the lowest mean and median IMD average rank and extent, the mean and median of IMD local concentration is lowest for the 'coast' cluster. This suggests that the severity of deprivation is highest in coastal areas which are moderately built-up. Although these are the smallest clusters with a much lower range of IMD values.

Of the more rural districts, the 'mountain' cluster has the highest deprivation (mean and median for all IMD measures), and has higher severity of deprivation than the 'urban' and 'suburban' clusters when median IMD rank average and local concentration are observed. Whilst the least deprived areas have higher extents of woodland (cluster 6), ecological status (cluster 6) and agricultural land (cluster 5 & 6) and agricultural land quality (cluster 5), they do not exhibit as diverse and overall as high quality and extent of natural capital.

Overall, the most deprived districts tend to have lower natural capital, but there are nuances and exceptions depending on the IMD measure used and with respect to some indicators of natural capital. Notably, some areas with very high extent and quality of a diverse set natural capital have higher levels of deprivation. The observed differences are largely noted with respect to measures of central tendency and there are large ranges in IMD ranks for most clusters. Thus the level of deprivation of districts within a particular cluster should not be assumed.

4.4 Evaluation of methodology

Selection of indicators has depended upon availability of data and consistent with other reporting (NCC, 2015), data availability beyond indicators of natural capital extent is limited. Future work would ideally encompass additional indicators of natural capital quality and other characteristics such as 'uniqueness' as identified by De Groot et al. (2003), and examine other social and demographic patterns.

The application of a classification technique has allowed spatial patterns of the types, quality of individual natural capital indicators, the mix of natural capital and overall natural capital, to be quantified and visualised. Although there may be some sensitivity to indicator selection and subjectivity in this selection and in the selection of an appropriate number and interpretation of clusters. Clustering is also subject to generalisation and indicators have been aggregated to administrative boundaries, an approach criticized by Dittrich et al. (2017). However, some degree of aggregation is necessary for comparability of economic and ecological data and districts were purposefully selected as they are coincident with local planning boundaries. They are also pertinent in the context of increased responsibility of local authorities, who have recently been critiqued by the Royal Town Planning Institute for neglecting issues of inequality (Pinoncely, 2016). Nonetheless, exploring the pattern of natural capital and deprivation for other administrative or landscape based scales may be revealing.

Data and methodological constraints of a national study mean only broad insight into the social distribution of natural capital assets is given. The actual benefits gained and services produced will depend on whether goods and services are directly or indirectly produced (Rova and Pranovi, 2017). Whilst other national studies (Turner et al. 2014; Dittrich et al. 2017) classify ES bundles rather than natural capital, these focus on service supply and do not include flows to beneficiaries for most services. At this scale, it is reasonable to speculate that the social distribution of natural capital reflects the social distribution of some ESs (with direct and local flows) generated by that capital, although this remains to be tested. As discussed in Chapter 3, section 3.1.3 scale dependent physical, social and economic processes will act to distribute benefits more widely, up to the global scale (Hein et al., 2006).

4.5 Implications for environmental equity

There is substantial environmental inequality in the UK but to date the evidence base neglects a full range of environmental benefits (see Chapter 2, section 2.4). The hypothesis that more deprived communities would occur in areas with less, and lower quality, natural capital, is supported by this district level analysis for some forms of capital, but a national social gradient is not universal across all forms natural capital. These findings echo the lack of a consistent conclusion across existing environmental justice analysis in the UK (Chapter 2, section 2.4).

Deprivation is found to be lowest in districts characterised by high woodland cover with greater accessibility and lower built-up land, with highest deprivation in urban areas with less natural capital, although high ecological status. In contrast, high deprivation occurs in coastal areas with large swathes of protected land and in upland rural areas which have high amounts of multiple types of high quality natural capital, this finding is consistent with literature discussing issues of and potential mechanisms behind rural poverty (e.g. Shucksmith, 2012). There are also some rural areas with lower deprivation but also lower aspects of several indicators of natural capital. For example, some urban districts have higher ecological status compared to agricultural dominated, rural districts, which also tend to be the least accessible. The higher ecological status in urban and suburban areas is likely due to their positive association with garden area and evidence of the high potential biodiversity of urban gardens (Goddard et al., 2010). It is therefore possible that benefits provided by urban gardens in more deprived areas could exceed those provided by agricultural land in less deprived areas, although this depends on how the land is managed (Power, 2010). These opposing patterns demonstrate the importance of exploring environmental inequality for different contexts, including within and between urban and rural areas, and for multiple types of natural capital.

Some of the observed associations between deprivation and natural capital are spatially driven and correspond with known north-south economic inequalities (Whitehead, 2014). However, social, political, economic and environmental processes have interacted historically and continue to shape variation in deprivation and natural capital across the country. To disentangle such complex intertwined processes is beyond the scope of this

analysis. Rather, I have sought to draw attention to current social patterns in natural capital distribution which have relevance for social objectives of natural capital strategies.

4.6 Implications for planning and land management

Building on Ostrom's (2009) socio-ecological system, Rova and Pranovi (2017) detail how people are both users and actors, and operate governance systems in the production and consumption of the goods and services derived from natural capital. On this basis, inequality in natural capital distribution has implications for sustainable management of natural capital in terms of the production of and demand for ESs (Andersson et al., 2007; Bennett et al., 2016; Ernstson, 2013).

De Groot et al. (2010) discuss the potential for the ecosystem approach to identify synergies for investments which produce ecological, social and economic benefits. An understanding of the relationship of natural capital (and ESs) to social characteristics of communities is an important aspect of identifying such synergies and maximising these benefits. This is especially so as not all ESs can be maximised concurrently and some trade-offs between social and ecological outcomes are inevitable (Daw et al., 2011; Seppelt et al., 2011).

For England, this chapter has shown that deprived districts have both high and low natural capital, and so emphasises that different, location specific approaches to maximising social, ecological and economic benefits are needed. This includes those focussed on encouraging the use of the available services from natural capital by deprived communities (O'Brien & Morris, 2014), and those where investment may be needed to increase potential natural capital and its services available to the poor. More in-depth discussion regarding synergies and trade-offs between natural capital, ESs, spatial scales and social and ecological outcomes is provided in Chapter 8, section 8.5.

4.7 Summary

This national analysis has developed some preliminary insights into the social distribution of ecosystem goods and service of a high-income, urbanised country through an environmental inequality analysis of natural capital in England, fulfilling objective 1 as established in Chapter 2 section 2.5. Overall, considering a wider range of natural capital than in previous EJ analysis reveals some interesting patterns nationally, and whilst some inequalities in the social distribution of environmental 'goods' are evident, this is not consistent in all places, nor for all types of natural capital. Thus a 'one-size fits all' national policy to address inequalities in ecosystem goods and services would not be appropriate, rather national policy should require incorporation of equity concerns within ecosystem management at sub-national scales. This aligns with the second research objective, analysis of the social distribution of ESs at sub-national scale, which has also been determined as necessary for enabling more spatially sophisticated modelling of a complex set of social, economic and ecological interactions which drive the production of ESs and environmental inequalities, as discussed in Chapter 3 section 3.1.3. The next three Chapters begins to address this challenge through analysis in English case study regions, whose selection is informed by national analysis presented in this chapter, as discussed in Chapter 5.

Chapter 5 The social distribution of air pollutant removal in regional case studies

5.1 Introduction to the analysis of the social distribution of ecosystem services

As the first of four analysis chapters within this thesis, Chapter 4 provided insight into the variation in natural capital characteristics and deprivation across the whole of England. This revealed that at the district level rural deprived areas featured some of the highest extent and quality of natural capital yet the converse is found in dense urban areas where deprivation and low natural capital coincide. The geographical variation in results supports the need for local analysis to better understand the social distribution of benefits which may be provided by the natural environment. This analysis is presented in the subsequent three chapters (5-7). These chapters present methods and results from the assessment of the social distribution of selected ESs for three diverse case study regions and address objective 2 (the social distribution of ESs) and objective 3 (sensitivity testing).

As outlined in the research design (Chapter 3, section 3.1.3), the approach to analysis in case study regions is more detailed than applied in the preceding national assessment. It uses the spatially defined flows of ESs to map the supply of benefits from natural capital to the population within the case study regions, how this is balanced with demand for ESs in those areas and in turn how these change across more and less deprived areas. The ESs selected for analysis are air pollutant removal, recreation and surface water runoff reduction (attenuation) (SWRR). These are chosen since they are pertinent for health and wellbeing in an English context, are key services for consideration in management of our natural environment and can offer insight into the effect of the spatial scale of ES flows on the social distribution of ESs. Reasons for selecting of these ESs were elaborated on in Chapter 3 section 3.2.2.

The results are grouped according to ES and thus are presented for all three case regions together. Descriptions of the study regions and reasons for their selection- Northampton, the South Pennines and Leeds – are given in the subsequent section (5.2). This chapter continues by presenting the analysis for the social distribution of air

pollutant removal. Chapter 6 and Chapter 7 examine recreation and SWRR respectively.

As highlighted in Chapter 3 section 3.3.1, a similar approach based on a common framework is used for mapping all ESs, however specific methods are individual to each ES, developed to provide a reliable estimate of the spatial distribution of that particular ES, given differences in their physical construct. Therefore Chapters 5-7 provide detail of the data and methods used to map the respective ES in terms of ES supply and flows, ES demand and the net ES index (objective 2), and the variants to these applied for sensitivity testing (objective 3). Each then continue by presenting results including a spatial description of the distribution of the ES supply, demand and resultant net ES index. Changes in ES across areas of higher and lower deprivation are reported using several global and spatial statistics (details provided in Chapter 3, section 3.3.2.3). These establish whether net ES increases or decreases with increasing deprivation and if so, what underlies this distribution (the spatial pattern of supply, demand or both), or if there is a more nuanced non-linear relationship between ES and area deprivation. Finally the sensitivity of results to the ways in which ES supply and demand are mapped is described for all case study regions collectively. Here I answer the question of whether the social distribution of ES changes if different values are assumed when quantifying ES supply, or when population density features as an element of ES demand. Specifically, the direction of any change is noted and implications for interpretation of the results are deliberated.

The primary aim of this chapter and Chapters 6-7 is to communicate the findings of all case studies together, for all ESs, so that a full narrative is presented. For this reason, the text is primarily descriptive; links to related research and the wider literature are drawn upon and the implications of the results examined in detail in the subsequent discussion chapter.

5.2 Case study region selection and descriptions

Case studies appropriate for assessing a social gradient in ESs should include areas of high and low deprivation and a range of natural capital. As highlighted in Chapter 2, section 2.5 inclusion of rural and urban areas is desirable to extend existing studies which have concentrated upon the distribution of environmental benefits within urban settings. Ideally, for extending current knowledge base of environmental inequalities,

the cases should be locations where previous analysis of inequalities in greenspace have not been undertaken.

The natural capital clusters established in Chapter 4 (Figure 4.4 & 4.5) and associated levels of deprivation provided the initial framework from which potential case studies could be identified. The rationale for case study selection firstly regards the clusters from which case study regions would be selected from, and subsequently the choice of specific location and boundary.

5.2.1 Selecting case study locations

Clusters 2, 3 and 5 were identified as the key clusters from which to select cases studies. Whilst high deprivation and diverse natural capital characteristics are observed for clusters 1 and 4, these comprise of few districts, with those in cluster 1 solely concentrated in London. The unique nature of these clusters indicates that analysis of these districts would be less relevant more broadly. Areas in cluster 6 are not considered since deprivation is lowest in these areas, whilst pockets of higher deprivation do exist, analysis of distributional environmental justice is most pertinent in areas where high deprivation is experienced.

Cluster 5 is the largest cluster, with the greatest geographical spread throughout England. Average deprivation of districts in cluster 5 is lower than clusters 2 and 3 and most indicators of natural capital lie close to the national mean, apart from agricultural land indicators which are higher than average. In contrast, districts in cluster 3 have been demonstrated to have the highest mean level of deprivation of a rural area and the greatest range in types of natural capital. Cluster 2 represents the more urban English districts which experience lower than average natural capital (except with respect to ecological status) and higher deprivation (which is similar to cluster 3). Three case study locations were therefore identified (Figure 5.1) to encompass these three clusters – the South Pennines (cluster 3), Northampton (clusters 2 & 5) and Leeds (cluster 5). Three case studies were deemed to be suitable on the basis that they provide insight into the social distribution of ESs in a range of contexts and were feasible to undertake within the research timeframe.

South Pennines

The South Pennines was selected as a region of interest on the basis that areas of high deprivation across cluster 3 are concentrated in this area. Moreover, across the area higher deprivation occurs in both rural and urban locations. It may therefore be expected that deprived areas experience similar levels of or higher ES than areas with a lower deprivation. This contrasts with areas with a clearer urban-rural gradient in deprivation (where high deprivation is concentrated in inner urban areas), where a

similar gradient in ESs may be expected. Unlike the majority of areas encompassed by the mountain clusters it is not designated as a National Park or Area of Outstanding Natural Beauty (which are subject to specific planning constraints). It is however under pressure from human activity as it lies close to many large urban conurbations, which through numerous ecological and social connections it also plays a critical role in supporting. In particular the extensive, openly accessible moorlands are important for their peat which stores carbon, the habits and species they support, for reducing flooding downstream and providing opportunity for recreation (Natural England, 2014).



Figure 5.1 Case study locations

Northampton

The town of Northampton was selected as an example of a 'typical' large town in England. Northampton is one of England's largest towns with a population of >212,000 (ONS Census, 2011). A historic county town, it has since experienced major development from industrialisation arising with the construction of Grand Union Canal in the 19th Century and planned expansion following New Town status designation in the 1960s. Its location between the UK's largest population centres of London and

Birmingham and connection with key infrastructure makes it attractive for but also under pressure from continued growth (West Northamptonshire Joint Planning Unit, 2014). Comparable to numerous urban areas in England, Northampton is surrounded by predominantly agricultural districts with small settlements (cluster 5), with those surrounding Northampton (e.g. South Northamptonshire) including some of the least deprived areas in England. These rural landscapes are characterised by gently undulating hills, mixed agriculture with historic hedgerow patterns and houses and wetland mosaics created from flooded gravel pits (Natural England, 2014). Therefore a case study in this location which includes the town and neighbouring rural areas surrounding Northampton can provide insight into whether a gradient in ESs is consistent with a rural-urban gradient in deprivation.

Northampton is situated within the Nene Valley, as such maps of ESs have been produced across the area as part of the development of a Nene Valley Nature Improvement Area (Rouquette, 2016). ES supply and demand maps at high resolution were made available subsequent to analysis for this research having commenced, these however do not spatially connect SPAs with SBAs or account for flows of services from natural capital beyond the immediate boundaries of the NIA.

Leeds

The city of Leeds is the only large urban area which is not assigned to cluster 1 or 2 in the national classification. This is due to the unique nature of the Local Authority District boundaries which are delimited beyond the city boundaries incorporating rural areas and small settlements. More commonly district boundaries are defined at the urban fringes. Consequently, Leeds district falls within cluster 5, however this conceals significant variation in natural capital and the presence of densely built-up areas. Thus, Leeds is a similar case study location in some respects to Northampton. However, it was selected because it also contrasts Northampton. Specifically, the district is surrounded by areas of high natural capital (cluster 3 to the north and west, cluster 6 to the south) and it is located close to the Yorkshire Dales National Park and therefore there is potentially high supply of ESs which may benefit Leeds district including the city. There are several distinct landscape characterisations across the district but the area covering the city itself and to the south is primarily a low lying urban landscape, with farmed open country. To the east the landscape is dominated by arable farmland based on fertile soils, with several country houses and large parklands with woodland areas. To the north of the district the area moves towards upland fringes and features more varied topography and greater woodland cover (Natural England, 2014). Despite a strong and growing economy overall (Centre for Cities, 2019) there are significant

clusters of high deprivation in Leeds city, which has prompted longstanding concerns regarding social inequalities (Boyle & Alvanides, 2003).

5.2.2 Case study extents

Following identification of case study locations, boundaries incorporating areas of higher and lower deprivation were determined. The case study extents were chosen to facilitate analysis of whether inequalities occur along a rural-urban gradient and to include a large enough sample size of LSOAs for robust statistical analysis (the minimum number of LSOAs included is 142 for the South Pennines region). Maps of the boundaries and of variations in deprivation across each case study are provided in Figure 5.2 to Figure 5.7.

For Leeds, as highlighted above, the district boundary extents more widely than for other large conurbations in England and thus incorporates the city and more rural surroundings characterised by smaller market towns. This boundary meets the requirements above and is thus used to define the Leeds case study region.

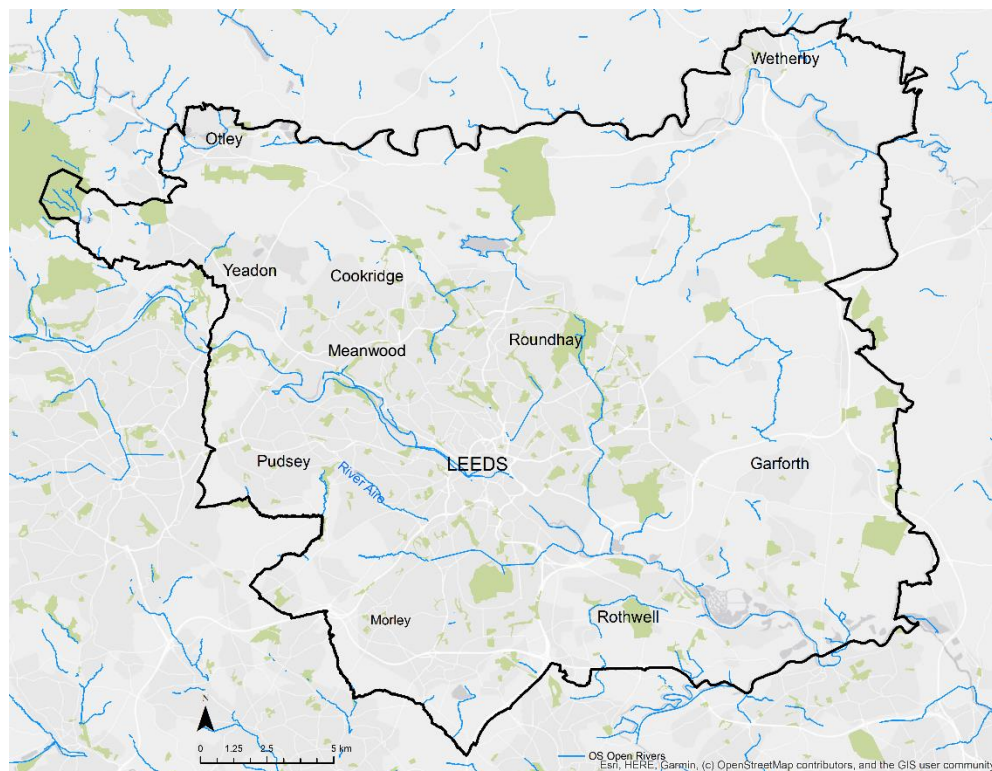


Figure 5.2 Leeds case study region boundary

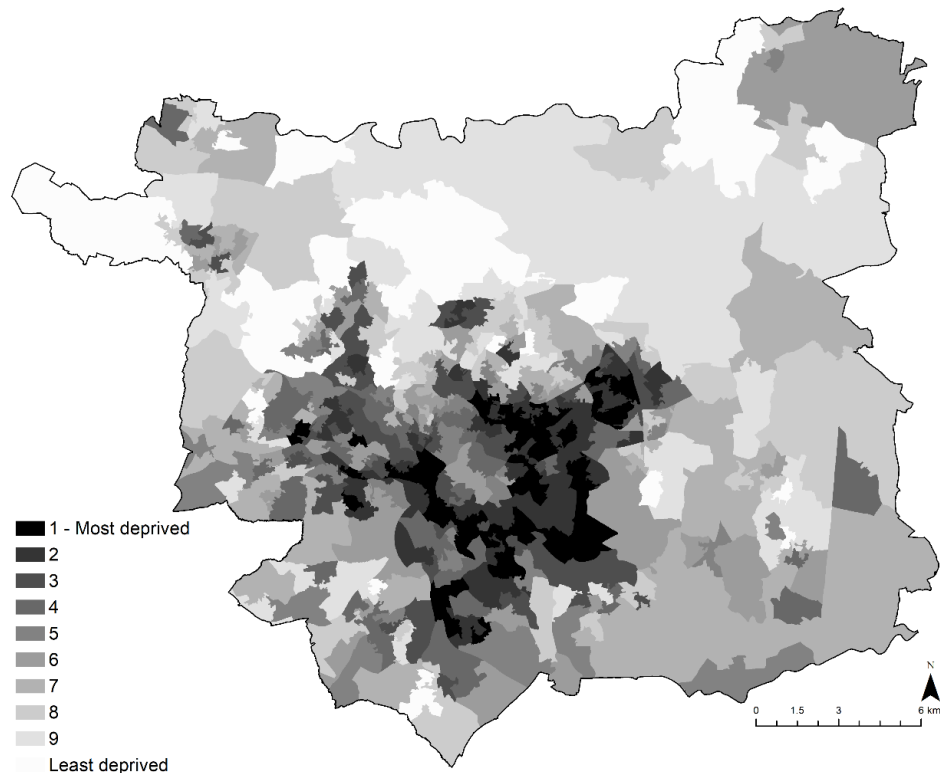


Figure 5.3 Patterns of deprivation across the Leeds case study region based on population weighted IMD deciles whereby 1 = most deprived 10% population, 10 = least deprived 10% population. Spatial units are LSOAs, and represent the spatial units for analysis of inequalities in ESs.

Boundaries of the Northampton and South Pennine case studies necessarily extended beyond a single district to meet the requirements set above and to cover a similar aerial extent. The South Pennines area is typically defined by the South Pennine Local Nature Partnership's (LNP) boundary (partnerships established to facilitate improvements to local natural environments), however analysis of the entire LNP area is not feasible in conjunction with other case studies. Thus a subsection of this was defined to primarily incorporate two districts - Calderdale and Rossendale – which correspond to cluster 3 only. This focussed analysis upon the southern part of the South Pennine LNP to maintain distinction from analysis for the Leeds case studies (ES supply to the northern areas of the LNP would likely overlap with ES supply areas for Leeds. The boundary of the Northampton Local Authority District formed the basis of the Northampton study, however, once more to meet the requirements above, this was extended to incorporate rural areas surrounding the town to the North and South. The boundary was not extended to the east of Northampton to retain a focus in West Northamptonshire, which is treated as a single unit from a planning perspective (West Northamptonshire Joint Planning Unit, 2014).

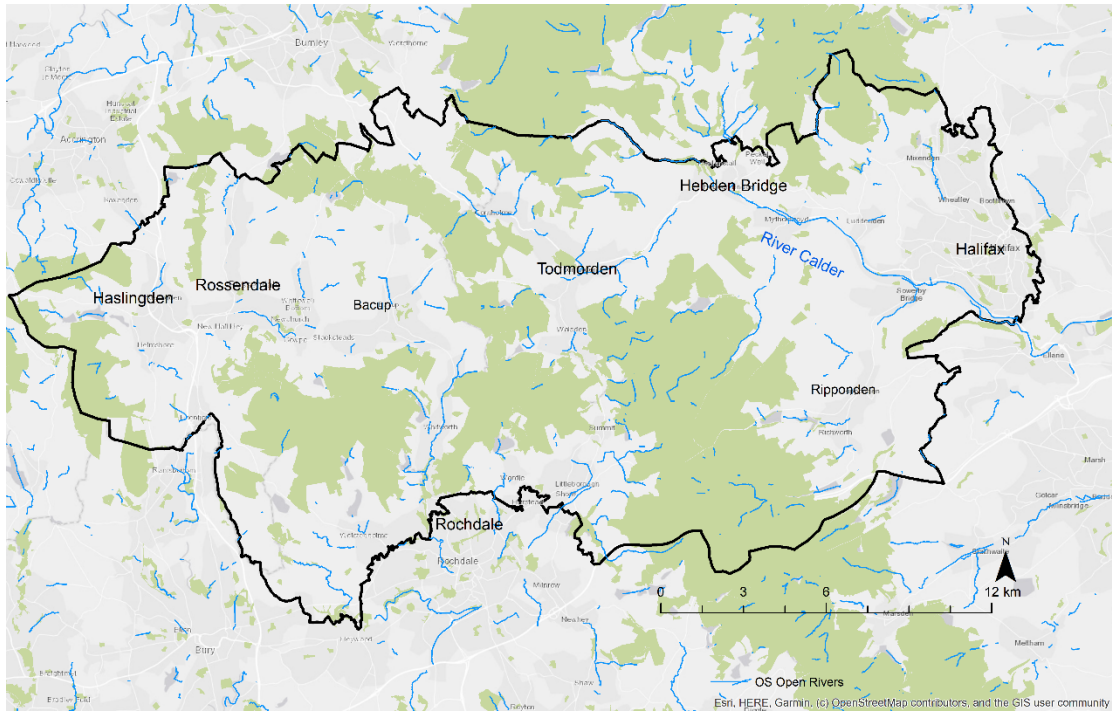


Figure 5.4 South Pennines case study boundary

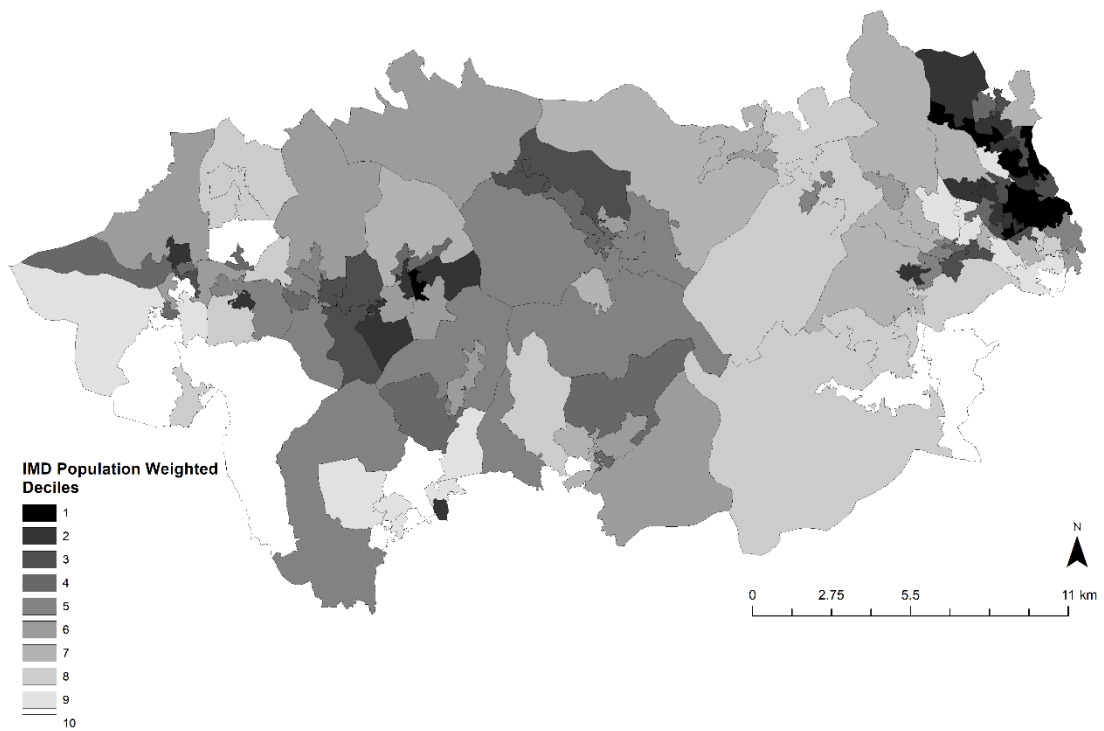


Figure 5.5 Patterns of deprivation across the South Pennines case study region based on population weighted IMD deciles whereby 1 = most deprived 10% population, 10 = least deprived 10% population. Spatial units are LSOAs, and represent the spatial units for analysis of inequalities in ESs.

Based on these case study extents, Table 5.1 and Figure 5.8 further characterise the districts encompassed by the case studies with respect to deprivation levels and the types of and quality of natural capital as determined by the indicators developed in the previous chapter.

Overall, the selected case study regions encompass urban deprived and rural deprived areas, together with wealthier areas in both contexts. The natural characteristics vary considerably, with some rural areas dominated by high quality agriculture and waterways and others which are largely accessible moorlands and semi-natural grasslands. Critically, in all regions there is considerable interest in managing the natural capital assets with respect to the ESs they can provide, the broader human-nature relationship and the need to provide suitable habitats to support biodiversity.



Figure 5.6 Northampton case study boundary

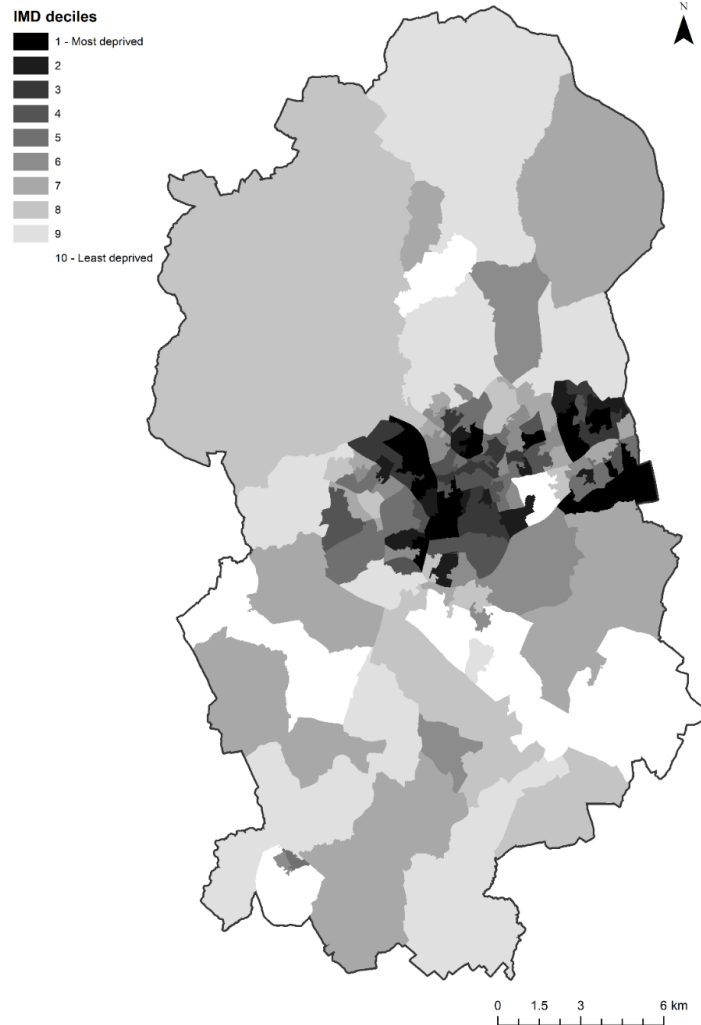


Figure 5.7 Patterns of deprivation across the Northampton case study region based on population weighted IMD deciles whereby 1 = most deprived 10% population, 10 = least deprived 10% population. Spatial units are LSOAs, and represent the spatial units for analysis of inequalities in ESs.

For example, in the South Pennines, Pennine Prospects together with the Local Nature Partnership drive numerous projects such as the 'Watershed Landscape' and the 'Woodland Heritage Project' (Pennine Prospects, 2019). Including the Leeds region and Calderdale in the South Pennines, iCASP (Yorkshire Integrated Catchment Solutions Programme, 2019) seeks to find effective solutions using the natural environment to challenges (such as flooding and climate change impacts in urban areas) faced with the Ouse Drainage Basin. With regards to Northampton, the development of the River Nene Nature Improvement Area emphasises the importance placed in the area on conserving habitats, primarily the wetlands in this case and promoting access to nature for the large nearby population (Bedfordshire, Cambridgeshire & Northamptonshire Wildlife Trusts, n.d.). Despite these interests, no

existing spatial analysis of inequalities in environmental benefits have been identified. The chapter now turns to the analysis of the social distribution of ES within these three regions. The next section presents analysis of air pollutant removal.

Table 5.1 IMD 2015 ranks for each district included within the case study regions. Lowest rank (1) indicates highest deprivation, highest rank (326) indicates lowest deprivation. District level IMD are aggregated from small area ranks; average indicates the mean IMD rank, extent indicates the prevalence of deprivation and local concentration indicates how clustered areas of high deprivation are.

Case study	Population	No. LSOAs	Local Authority District	IMD ranks		
				Average	Extent	Local Concentration
Northampton	267,506	167	Northampton	108	71	77
			South Northamptonshire	317	302	324
			Daventry	238	198	192
South Pennines	223,421	142	Rosendale	98	109	117
			Calderdale	96	83	58
Leeds	751,485	482	Leeds	100	58	24

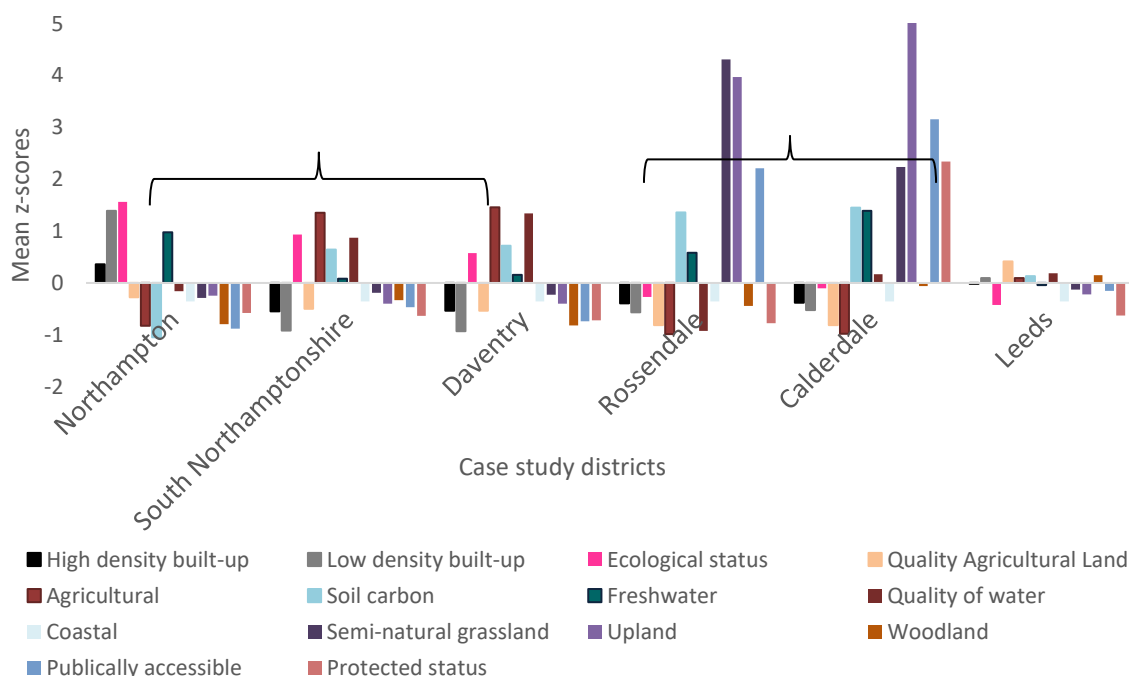


Figure 5.8 Natural capital characteristics of districts included in the case study regions (as derived in Chapter 4) indicated by mean z-scores for each natural capital indicator. A z-score of zero represents the national average, whilst a score of +1 is one standard deviation above the average.

5.3 Air Pollutant Removal

The air pollutant removal ES refers to the contribution of vegetation to improving air quality. Its importance in the context of this study arises from the considerable and unequal health impacts of air pollutants and thus its potential contribution to improving health, as reviewed in Chapter 3 section 3.2.2.2. Delimiting vegetation, in particular trees, is critical to mapping this service. Therefore input datasets should ideally be of higher resolution than is required for the other ESs addressed; these are detailed in the next section (5.3.1). Methods for mapping air pollutant removal ES (section 5) are based on the understanding that the spatial flows of air pollutant removal ES are local. Consequently supply is mapped within case study LSOA boundaries, using methods detailed in section 5. Methods for mapping ES demand are detailed in section 5.3.2.3, and are based on locating areas of higher air pollutant concentration. Calculation of a net ES index and the sensitivity tests applied are also outlined in these sections. Methods for assessing how equally the ES is distributed were detailed in Chapter 3 section 3.3.2.3.

The final sections in this chapter (5.3.3) present the findings from the analysis for Leeds (5.3.3), South Pennines (5.3.2.2) and Northampton (5.3.2.3). Responses to sensitivity tests are reviewed for all regions together in section 5.3.2.4.

5.3.1. Data

Vegetation is more effective at scavenging air pollutants compared to any other surface (Smith, 1981) and trees more so than other vegetation (Fowler et al., 2004), specifically coniferous trees which are not affected by seasonal variations (Freer-Smith et al., 2005). Thus the datasets required for mapping air pollutant ES supply must facilitate classification of land cover and in particular the identification of tree canopies and any additional information regarding tree characteristics. Existing data presenting land cover information such as the Land Cover Map of Great Britain LCM2007 (NERC CEH, 2011) and Ordnance Survey Mastermap Topographic Vectors (OS, 2014) only identify areas of woodland, not individual or small groups of trees. Whilst some studies of tree cover have been facilitated by data provided by local authorities (e.g. Barbosa et al., 2007), such datasets are not provided nationwide. On the other hand, high resolution aerial imagery is widely available across England and is therefore utilised for classification of vegetation. Notably, to distinguish trees from other vegetation and from manmade land covers.

For the academic community, colour infrared (CIR) aerial imagery at 0.5m resolution is freely available from the archived Landmap project (Landmap; Bluesky, 2014). Funded by the Joint Information Systems Committee, this imagery was acquired 2006-2010 by Bluesky (www.bluesky-world.com) and has good but incomplete coverage across England. The majority of imagery covering the case study regions was captured in 2009. Due to the commercial role in capturing the data, the imagery is also available to purchase for all of England and thus is also accessible by others and for areas not covered by the Landmap project. For large areas, image classification needs to be automated, this is best achieved using CIR imagery compared to imagery in the visible spectrum alone. This is due to the inclusion of Near InfraRed in addition to Red and Green bands; reflectance by green vegetated canopies is much greater for NIR wavelengths, which also penetrate deeper into the canopy thus revealing more regarding the density of vegetation canopies (Campbell & Wynne, 2011). As live green plants appear relatively dark in the photosynthetically active spectral radiation region and relatively bright in the near-infrared, differences in NIR reflectance can be used to emphasise and better differentiate denser vegetation from other land covers.

This is achieved using a Normalised Difference Vegetation Index (NDVI) where:

$$NDVI = (NIR-RED) / (NIR+RED)$$

Campbell & Wynne (2011)

The highest resolution imagery is desirable for achieving the highest classification accuracy, but given the total size of study regions imagery was resampled to a lower 5m resolution. This enabled classification for the case study regions to be undertaken within approximately one week for each region and to decrease the high data storage requirements. For a 10km² test area, classification accuracy for 5m opposed to 0.5m was only 5% lower when assessed against 'ground truth points' assigned through visual interpretation of the higher resolution imagery.

Image classification was carried out using the rule based feature extraction workflow in ENVI 5.2 software. This utilised spectral information including NDVI and textual information across a 7 pixel distance to distinguish between trees, grass and non-vegetated land. Classification rules were tailored to each case study region on account of the diversity of land covers, such as the moorland in the South Pennines and highly variable land cover of urban Leeds. Classification was verified as 80% accurate using ground truth points assigned by visual interpretation of 50cm resolution imagery. Image classification did not reliably differentiate coniferous and deciduous trees, which are ideally distinguished since coniferous trees have a longer leaf-on period during which

pollutants can be sequestered. For larger areas of trees OS Mastermap topographic vectors provide this information. Individual and small areas of trees were therefore assigned as deciduous, a conservative approach to ensure ES supply is not overestimated. OS Mastermap data is further applied to refine the boundaries of buildings and roads of the classification.

Mapping air pollutant ES demand requires identification of areas of higher concentration of air pollutants. This is also applicable for ES supply since vegetation is more effective at removing air pollutants where there is a higher concentration of pollutants. Estimates of relative air pollutant concentrations are generated using OS Mastermap Integrated Transport Network (ITN) vector layer and Average Annual Daily Flow traffic (AADF) count data from the Department for Transport (DfT, 2017). Section 5.3.2.3. details reasons for not using available national air quality data. The ITN data comprises vectors of all types of roads from motorways to pedestrianised streets. Attributes of these provide road classifications and names (OS, 2007). AADF data gives the average number of vehicles per day of major roads. 2017 AADF data was assigned to ITN roads using spatial joins, giving mean annual traffic counts for motorways and major roads. Since traffic count data are available for only a sample of minor roads, the mean traffic count is assigned to these.

The next sections outline the methods used for quantifying the ES from these datasets to producing maps of ES supply, demand and net ES index.

5.3.2 Methods

5.3.2.1 Air pollutant removal flows and supplies

Air pollutant removal, in the context of ES supply, describes the role of vegetation in removing pollutants from the air, thus providing cleaner air. The ES supply benefits the immediate area (Burkhard et al., 2014) and therefore the service providing areas (SPAs) share the same boundaries as service benefitting areas (SBAs) i.e. the case study regions' LSOAs. Air pollutants are more effectively scavenged by vegetation than other land surfaces (Smith, 1981).

Vegetation absorbs gaseous pollutants and intercepts or dissolves air particulates (in dry and wet conditions respectively) (Vos et al., 2013). Given the weather dependence of wet deposition, the effectiveness of vegetation at removing pollutants is normally considered with respect to their dry deposition rates (Nowak et al., 2006). These rates

are dependent upon several factors, which include (Broadmeadow & Freer-Smith, 1996; Beckett et al., 2000; UKNEA, 2011; Tallis et al., 2011; Vos et al., 2013):

1. Vegetation type and species – large trees have been shown to remove between two and ten times the amount of particulates from the air compared to grassland (Broadmeadow & Freer-Smith, 1996).
2. Tree canopy area and tree structure – the exterior area of trees controls the air turbulence and the impact of pollutants with the tree which enables their deposition.
3. Length of leaf-on period - the continual leaf area of evergreen trees enables more consistent removal of pollutants than deciduous trees.
4. Air pollutant concentration - greater concentrations lead to greater pollutant removal
5. Distribution of the pollutant particle sizes - very large and very fine particles deposit faster.
6. Windspeed

Detailed modelling which estimates how much air pollutants are scavenged by vegetation ideally utilises knowledge of these conditions combined with known deposition rates for a particular species and air pollutant. In addition to these factors, larger vegetation can act as a physical barrier to air flow which impacts the dispersion/concentration of pollutants. However, this is not often incorporated in models covering city-wide and larger areas since this is a highly localised process, usually modelled at street level (e.g. Vos et al., 2013).

The i-Tree modelling toolset would be an ideal means of modelling air pollutant removal to incorporate these factors (Nowak & Crane, 2000; Nowak et al., 2006). Originally developed for the USDA Forest Service using deposition rates appropriate to North America, its application has been focussed on this region (e.g. Nowak et al., 2006; 2007a,b; Nowak et al., 2013; Ning et al., 2015) but is expanding worldwide (e.g. Escobedo & Nowak, 2009; Selmi et al., 2016; Tiwary et al., 2016). However, in 2015, when the analysis for this research commenced, application in the UK was limited to two completed projects in Torbay, Devon and London (Rogers et al., 2011 & Treeconomics London, 2015). Subsequently Forest Research, the principal organisation for forest research in the UK, in collaboration with external researchers and organisations have developed the applicability of the toolset in the UK (see www.forestresearch.gov.uk/research/i-tree-eco; Hand & Doick, 2018). Despite this, the data requirements present a limitation to the use of this toolset for this analysis, in

particular, the requirement for knowledge of tree species and structures. Therefore for large areas, sampling strategies are used which result in aggregate estimates of pollutant removal. For this reason – given data regarding tree structure is not ubiquitously available across the study regions and a sampling approach will not accurately generate spatially explicit results - these tools were not used in this analysis.

Beyond the use of i-Tree software and in the absence of inventories detailing tree species and structure information, a dry deposition equation (Equation 5.1) using more generalised tree cover information can be applied (Jim & Chen, 2008; Zulian et al., 2014; Escobedo et al., 2015; Baró et al., 2016; Manes et al., 2016; Song et al., 2016). This approach combines data regarding the local or regional air pollutant concentration and averaged dry deposition rates with either the leaf-area index (ratio of leaf-area compared to the canopy size when projected onto the ground) and/or the canopy cover combined with data (Vos et al., 2013).

The dry deposition equation is given as:

$$F_{ii} = F_i \times A \times T \dots\dots\dots \text{Equation 5.1}$$

Where:

F_{ii} = total flux

A = tree cover

T = time

F_i = pollutant flux ($g/cm^2/s$)

Pollutant flux F_i is calculated as:

$$F_i = V_d \times C_i \dots\dots\dots \text{Equation 5.2}$$

Where:

V_d = deposition velocity (cm/s)

C_i = concentration of pollutant (g/m^3)

Normally, Equation 5.2 would be applied individually with respect to one or more air pollutant(s) of concern using known deposition rates. Deposition rates may be determined through laboratory based experiments using methods appropriate to the pollutant, such as washing and filtering of particulates from sample leaves to compute their mass of deposited particulate, e.g. Dzierżanowski et al. (2011), Liu et al. (2018).

However, for modelling air pollutant removal across streets, cities and regions rates are commonly transferred from other studies or databases. Consequently, rates are subject to issues of value transfer as discussed in Chapter 3 section 3.3.1. Potential uncertainties arise due to the high variation of reported deposition rates, given the varied environmental contexts in which they are measured. For example in Norway, deposition rates for particulates varied by up to a factor of 15 dependent on tree species, whilst in Poland differences were observed across species by up to a factor of 20 (Sæbø et al., 2012). Broadmeadow & Freer-Smith (1996) estimate the effectiveness of trees they cover in removing particulates compared to the ground varies between 2 and 12 fold. In addition some tree species are greater emitters of biogenic volatile organic compounds (BVOCs), which decrease air quality (Tiwary et al., 2016). Therefore the reliability of deposition rates is contested (Grundström and Pleijel, 2014).

Nevertheless, the use of average deposition rates reflects a consensus that trees are more effective than other vegetation types and land covers at removing air pollutants. The dry deposition model (Equation 5.1) estimates the actual mass removal rate of pollutants, however for this analysis only relative removal across the study region and not absolute quantities are required. Therefore actual deposition rates are not calculated and an alternative approach is taken which accounts for the effectiveness of vegetation at removing pollutants and an increase in this effectiveness in areas of higher pollutant concentration. The method adopted in this analysis applies a proxy to land cover data to account for the influence of vegetation presence and type upon the removal of pollutants. An additional weighting is then applied according to areas of higher or lower air pollutant concentration based on road proximity. It therefore is a GIS based approach which is informed by the relationships expressed in equations 5.1 and 5.2.

As Escobedo & Nowak (2009) observe, this relationship between tree cover, tree type and ambient air pollutant concentration is the foundation of the majority of studies examining the effect of urban forests on air quality. The chosen method is considered appropriate since ambient air pollutant concentration data covering all case study regions has a 1km² spatial resolution, hence use of this data (i.e. as required for direct calculations of the dry deposition equation) could potentially conceal differences in relative pollutant sequestration between neighbouring LSOAs in urban areas, where LSOA areas are often smaller. In addition, spatially resolved data on tree types is not available for the case study regions which limits the accuracy with which the deposition equations could be applied. That is, the derivation of average deposition rates for a broad classification (e.g. presence of coniferous or deciduous trees) from a large range

of estimates would be akin to assigning proxies to tree cover data. It should also be noted that land cover proxies have previously been applied for quantifying air pollutant removal ES (Burkhard et al., 2012; Ecoserv ES mapping tools, Winn et al., 2018). The approach taken in this study builds on these since it further accounts for the influence of ambient air pollutant concentration. It is accepted that this approach leads to inclusion of proximity to roads within estimates of ES supply and demand, and therefore the net ES index is subject to some ‘double counting’. However, the supply of air pollutant removal is dependent on where the vegetation is, not only its extent, and therefore is it considered that this is reflective of actual processes.

Table 5.2 Scores assigned as proxies for the effectiveness of land cover classes at removing pollutants for the baseline ‘best estimate’ scenario and two additional scenarios used for testing uncertainties (sensitivity testing).

Land cover class	Air pollutant removal scores		
	Best estimate	Low influence	Tree dominant
Coniferous trees	10	10	10
Deciduous trees & shrubs	1.5	0.8	4
Grassland	0.5	0	0
Non vegetated land covers	0	0	0

To implement the method a unitless score between 0-10 is applied to a highly spatially resolved land cover dataset; a tailored classification for estimating air pollutant removal (Table 5.2). This indicates the effectiveness of the vegetation in removing air pollutants; 10 denotes the most effective and 0 indicates no pollutant removal. Scores were derived from multiple sources which provide information of tree species’ and land covers’ deposition rates considering multiple pollutants (primarily PM₁₀, PM_{2.5}, SO₂, NO₂ and O₃), notably Nowak et al. (2013), Tiwary et al. (2016) and Zhang et al. (2002). The groups of vegetation were established based on those that could be identified from available data and in acknowledgement of the key determinants in different deposition rates. Notably, the difference between coniferous and non-coniferous trees (Freer-Smith et al., 2005). Aligning with the information regarding tree cover which can be extracted from available data (discussed in greater detail below), deposition rates were compared for species grouped as coniferous trees, deciduous trees, shrubs and grassland. The average differences in deposition rates across these groups were then compared to determine their scores. This revealed differences between coniferous and deciduous from approximately 2.5 fold to 13 fold, aligning with the observations for deposition of particulates made by Tiwary et al. (2009). Indexes which consider the overall contribution to air quality by examining deposition rates and emission of BVOCs given by Tiwary et al. (2015) indicated little difference between deciduous trees and

shrubs and therefore these were grouped together. Overall consensus regarding the much greater influence of trees in removing air pollutants means that little information is available regarding the influence of grasslands. However Fowler et al. (2004) observe that woodland is more effective than grassland by a factor of approximately 3.

As stated above, in line with the dry deposition model, the land cover scores are multiplied by a road density surface which serves as a proxy for air pollutant concentration. This generates air pollutant removal supply values as a 5m raster surface. The use of the road density surface for estimating ES demand leads to an inextricable link between supply and demand which Hegetschweiler et al. (2017) observe is a common occurrence. Details of how this surface is generated are provided in section 5.3.2.3 below. Briefly, values between 0-1 are assigned based on the density of the road network combined with traffic count data. A value of 1 is assigned to locations proximal to the greatest number of roads with the highest traffic counts, this decreases with distance until a threshold of 500m, beyond which locations are given a value of 0. The use of a proxy for air pollutant concentration is preferred due to the high spatial resolution which can be achieved, ensuring that the importance of the location of vegetation is accounted for. For example, a high proportion of coniferous trees within an LSOA, would not necessarily mean that there is high ES supplied by this LSOA. Rather, ES supply is highest in areas where there is a relatively high presence of this vegetation within the proximity of roads (i.e. vegetation which is in the 'right' place). Available air pollutant concentration data at 1km² resolution would not provide the detail required to achieve this.

The area weighted mean of ES supply values are computed for LSOA polygons using the intersect tools available in Geospatial Modelling Environment software. Final ES supply for LSOAs is given as values normalised from 0-10, whereby 0 indicates the lowest and 10 the highest supply.

5.3.2.2 Air pollutant removal sensitivity testing

Vegetation's role in improving air quality is considered both potentially underestimated (UKNEA, 2011) and overestimated (Grundström and Pleijel, 2014). In addition, potential errors arise from the use of simple proxies to quantify ES, as highlighted by Eigenbrod et al. (2010) and deliberated in greater detail in Chapter 3 section 3.3.1. On this basis, and given the range of deposition rates encountered in the literature, potential uncertainties in the proxies applied are addressed in sensitivity testing. A social distribution of air pollutant removal ES that is robust to these uncertainties will

provide greater confidence that the conclusions reached are not simply a product of the chosen model.

The baseline ES supply is considered to be a 'best estimate' scenario; variants of the scores applied represent a 'tree dominant' scenario whereby all trees and shrubs are given a greater weight in comparison to grassland and manmade land covers; and a 'low influence' scenario where overall the effectiveness of grassland, shrubs and broadleaved trees in removing air pollutants is minimised (Table 5.2). It is necessary to place values on a scale of 0-10, since it is consistently shown that coniferous trees are the most effective pollutant scavengers they are assigned a value of 10 for all scenarios. The different scenarios were constructed to enable mapping of possible variations in the spatial distribution of ES supply and thus whether the social distribution was sensitive to these different estimates. It was further tested how results from the use of readily available statistics on land use such as the generalised land use statistics correlate with the more detailed approach used for this assessment. This can help inform whether these datasets can be used reliably when resources or availability of data is limited. This comparison is also pertinent as many studies concerned with inequalities in the distribution of environmental benefits solely use data on coverage of greenspace (as reviewed in Chapter 2 section 2.2).

5.3.2.3 Air pollutant removal demand

A risk reduction perspective is normally adopted by studies examining the demand for regulating ESs (Wolff et al., 2015). Thus the higher the risk comparative to the desired conditions, the greater the demand. For air pollutant removal, the level of risk is directly related to the level of air pollution since a higher concentration of air pollutants leads to greater potential harm to human health (see Chapter 2, section 2.1.2). Additional factors may also be included in the definition of risk, including the number of people potentially affected and the vulnerability of those people (Wolff et al., 2015).

Few studies modelling air pollutant removal by vegetation simultaneously account for the demand for this service. This is exemplified by Wolff et al.'s (2015) review of studies mapping ES demand, which contains no specific reference to studies mapping demand for air pollutant removal. Furthermore, Wei et al (2017) select only one study examining air pollutant removal - Baró et al. (2015) - in their review of ES literature integrating both supply and demand. That said, there are several studies which do account for air pollutant removal demand including Burkhard et al. (2012), Baró et al. (2015), Larondelle & Lauf (2016), Song et al. (2016) and Verhagen et al. (2016). Each take a different approach, although they all relate to Burkhard et al.'s (2014) selection

of three appropriate indicators of air pollutant removal ES demand: air pollutant concentrations, air quality standard deviation and critical loads exceedance.

Baró et al. (2015) provide an example of equating demand to exceedance of air quality standards, which can be viewed as the desired environmental condition. They estimate city-wide demand for five European cities as the reduction in air pollutant concentration for three pollutants (PM_{10} , NO_2 , O_3) required to reach standards defined by EU Air Quality Directive (2008/50/EC) and WHO air quality guidelines (2006) from current pollutant concentrations. Verhagen et al.'s (2016) continental scale assessment offers an example of demand based on ambient air pollutant concentration by using modelled concentrations of NO_2 . Song et al. (2016) build on this by incorporating population density together with ambient air pollutant concentrations in their analysis of the spatial distribution of air pollutant reduction by South Korean forests.

Alternatively, Larondelle & Lauf (2016) equate demand for PM_{10} removal in Berlin with modelled traffic emissions on the basis that this represents the biggest single source of PM_{10} city wide emissions. This assumes that locations with greater traffic emissions have a greater concentration of air pollutants. This aligns with reporting by DEFRA (2017) that when UK Air Quality Management Areas are declared for exceedances of Clean Air Strategy targets, traffic is the major source of PM_{10} (75%) and NO_2 (96%). DEFRA (2017) further identifies road traffic as the main or a key contributor of $PM_{2.5}$, Benzene and Carbon Monoxide. Furthermore, pollution from road traffic is a key concern as particulates (PM_{10} and $PM_{2.5}$) which penetrate the respiratory system are identified by WHO (2013) as presenting the greatest risk to human health.

Other sources of pollution include power stations, industrial facilities, households and agriculture (Guerreiro et al., 2014; DEFRA, 2017) and these may be more important in some, especially rural, areas. There are national datasets available which present modelled concentrations of multiple air pollutants (DEFRA: <https://uk-air.defra.gov.uk/data/pcm-data>) and account for these various pollutant sources. However these datasets are not used in this study since they are spatially resolved to 1km only, which is lower than the area of many LSOAs, especially the dense urban areas within Leeds (376/482 LSOAs < 1km²) and Northampton (127/167 LSOAs < 1km²). Use of these datasets would generalise demand across urban areas, thus limiting identification of differences between neighbouring LSOAs. Identification of these spatial differences is important since the supply and deprivation of neighbouring LSOAs can diverge considerably. Instead, for this study, as Larondelle & Lauf (2016), an approach based on the importance of traffic related emissions for air quality is used since it enables creation of a more spatially resolved dataset. This aligns also more

closely with the spatially detailed assessment of air pollutant removal ES supply. Additionally, as expressed in Chapter 3 section 3.3.1.2, it is the relative pollution level which is important, actual pollutant concentration information is not required.

To estimate traffic emissions, Larondelle & Lauf (2016) use a detailed modelled incorporating traffic counts with the types of streets, engines and fuel. Although this data is not available for all streets and they use a simple buffer analysis to extrapolate to adjacent streets. In this analysis, a simplified model is used given the availability of data across the study sites. This includes the road network, road type and traffic count data. A gridded 5m resolution surface representing road density weighted by traffic counts (see section 5.3.2.3) is created by application of the kernel density tool in ESRI ArcGIS 10.3. This assumes greater air pollutant concentration given a closer proximity to roads with high traffic flows and in areas where there is a dense road network. A cut-off distance of 500m is applied which assumes that pollutants emitted from road traffic disperse no further than this distance.

As Song et al. (2016) highlight, for the removal of air pollutants to be an ES – that is beneficial to human health – there must also be a human presence. Therefore only the areas which are considered as locations used by people are considered in the calculation of ES demand. Data on the actual presence of people across the entire study area is not feasible and is highly variable in time, instead this analysis uses locations where a human presence is likely at some time on a daily basis. This comprises of a 25m buffer surrounding built-up areas extracted from the OS Mastermap Topographic datasets and areas accessible for recreation purposes as determined for analysis of the recreation ES (Chapter 6). Demand for air pollutant removal is therefore based on these areas extracted from weighted road density surface. For all other areas, primarily agricultural fields where there is no public right of way or open access rights, demand is set to 0. The area weighted mean demand is computed for each LSOA using intersect tools in Geospatial Modelling Environment (GME).

As highlighted in Chapter 3, section 3.3.1.2, for each ES, a utilitarian concept of demand, whereby population is incorporated within demand estimation, is also computed. Thus areas of highest demand are where air pollutant concentration modelled by the road density surface, and number of people per km², are greatest

A net ES index is calculated for each LSOA through subtraction of demand from supply, rescaled to 0-10, whereby 0 indicates in relative terms the area of highest demand/lowest supply and 10 indicates lowest demand/highest supply.

5.3.3. Results

5.3.2.1 The spatial and social distribution of air pollutant removal in Leeds

The highest supply of the air pollutant removal ES is in the northeast of the study area covering the small town of Wetherby, to the west on the outskirts of Leeds city, covering Yeadon, in some city suburbs to the north, covering parts of Cookridge, Meanwood and Roundhay, and to the southeast of the study area covering Garforth (Figure 5.9). Low supply can be found in the city centre and in larger rural districts to the northwest. There are however scatterings of higher and lower supply across the district. Demand is highest in the more densely populated LSOAs, notably near the city centre, this is expected given that demand is based on the road network. When combined into an overall ES Index this creates a spatial pattern where the air pollutant removal ES index is low in Leeds City centre and highest in rural areas to the north and east. Compared to LSOAs within the city, the ES index remains higher for some of the small towns in the district, such as Wetherby and Garforth, although there are a few LSOAs with a lower ES index in Yeadon and Otley in the west.

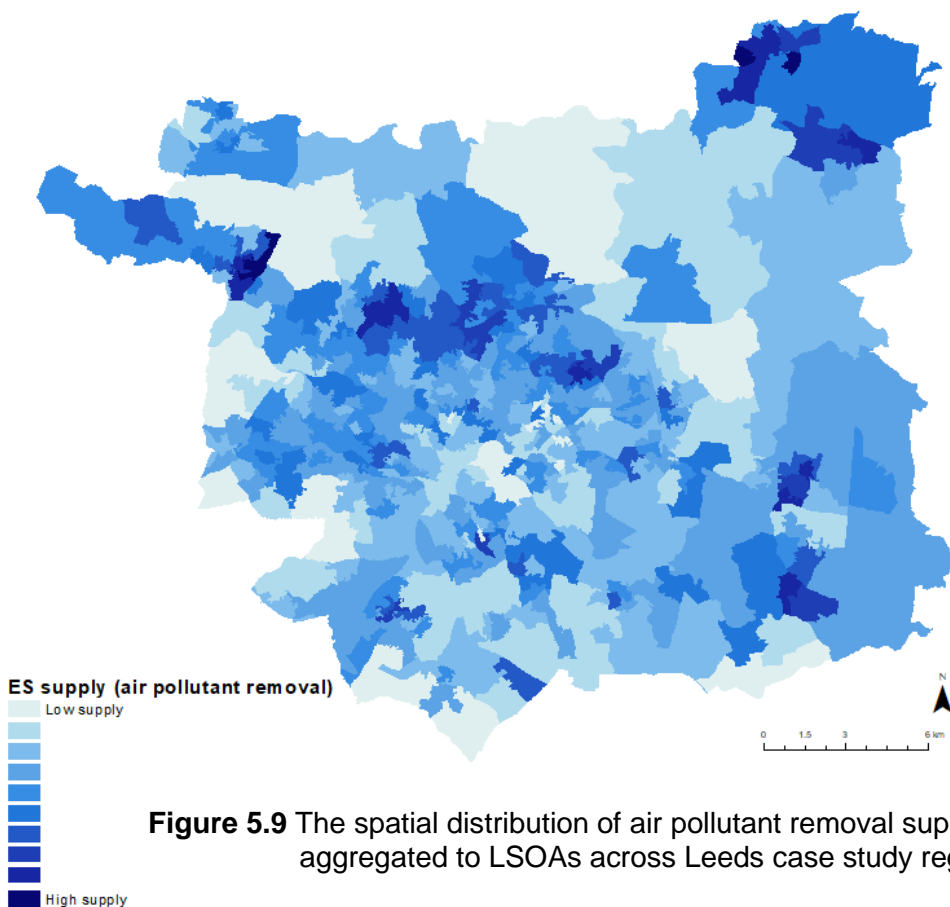


Figure 5.9 The spatial distribution of air pollutant removal supply aggregated to LSOAs across Leeds case study region

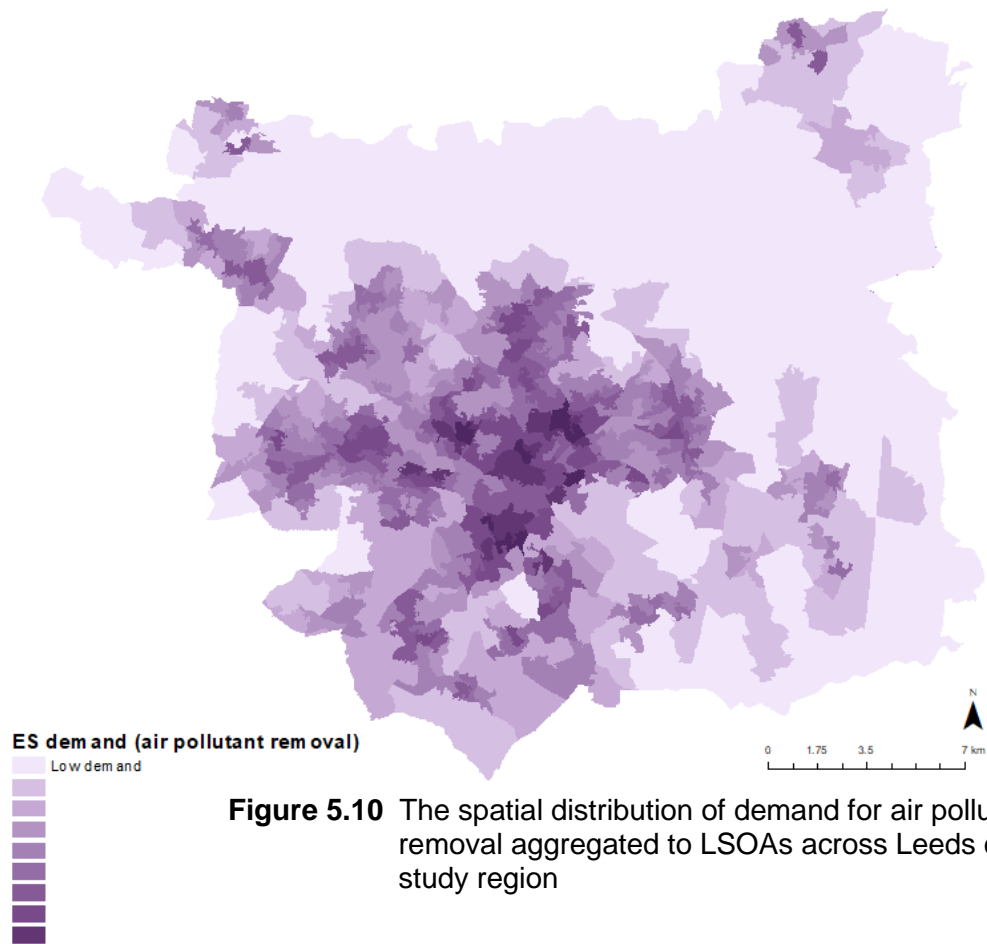


Figure 5.10 The spatial distribution of demand for air pollutant removal aggregated to LSOAs across Leeds case study region

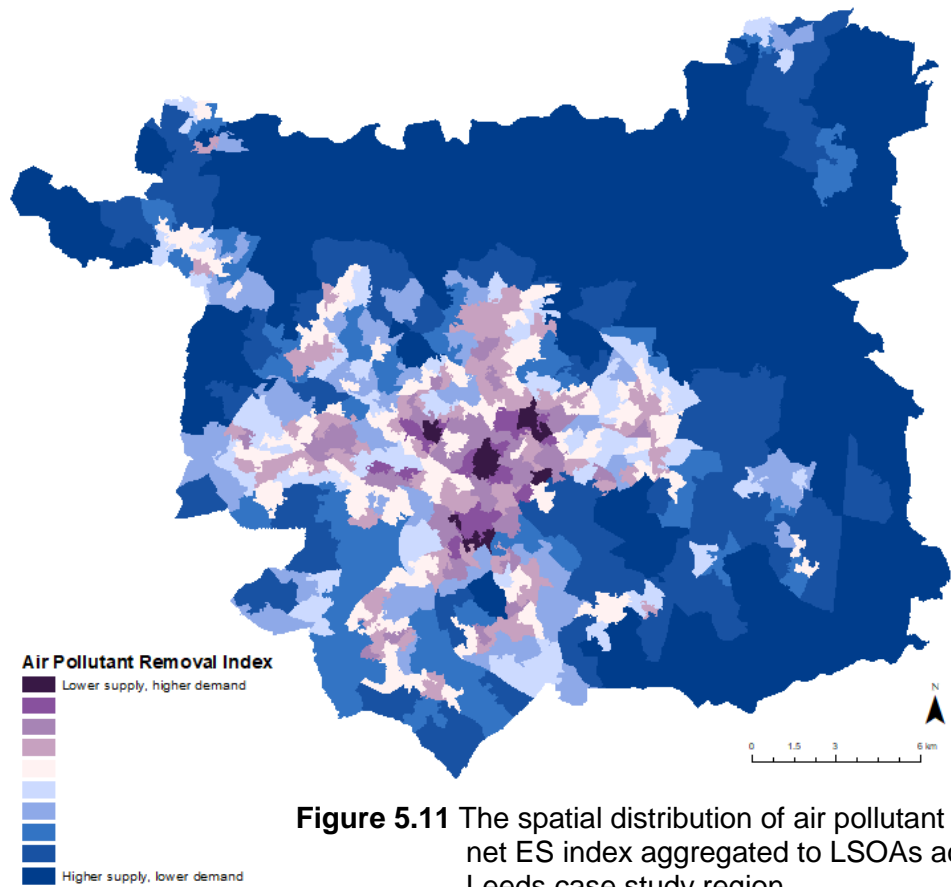


Figure 5.11 The spatial distribution of air pollutant removal net ES index aggregated to LSOAs across Leeds case study region

Correlation between ES demand and IMD ranks is significant, strong and negative ($\rho = -0.489$), and there is weak significant correlation between supply and deprivation ($\rho = 0.144$) (Table 5.3), thus demand is greater and supply is lower for the most deprived areas. This combines to generate a significant moderate positive correlation between the air pollutant removal net ES index and IMD ranks ($\rho = 0.514$, Table 5.3), meaning that net ES indexes (i.e. exceedance of supply relative to demand) are higher for LSOAs with lower deprivation. This pattern is further illustrated by the boxplot (Figure 5.14), showing a relatively linear increase in ES for each increasingly deprived population weighted decile. The interquartile range of net ES index values are similar for each decile. Computation of the concentration index (0.18) also confirms an unequal distribution, although this is relative low (with 0 indicating an equal distribution). The concentration curve illustrated in Figure 5.30, being closely tied to the concentration index, shows a similar pattern as cumulative air pollutant removal (based on net ES index) very closely matches the line of equality for the three least deprived deciles and falls below this line for the more deprived deciles, but there is still not a large disparity.

ES demand is the main driver of the observed correlation between the air pollutant removal net ES index and deprivation; a stronger magnitude of correlation is exhibited between demand and deprivation compared to supply and deprivation. Additionally, Figure 5.13 shows median demand values for deprivation deciles almost consistently decrease as deprivation increases, in comparison Figure 5.12 shows low median values and interquartile ranges for supply values across all deprivation deciles. LSOAs with higher supply tend to be outliers, but once more these are present across most deciles despite being slightly less common amongst the four most deprived deciles. The area with the greatest supply – a notable outlier- is in the 8th decile (less deprived), this is a relatively small LSOA in the suburban village of Yeadon and it's likely that the high supply is due to the lower amount of built-up area relative to LSOA size (it's located on the edge of open countryside) yet proximal to roads where the pollution is generated which increases efficiency of air pollutant removal.

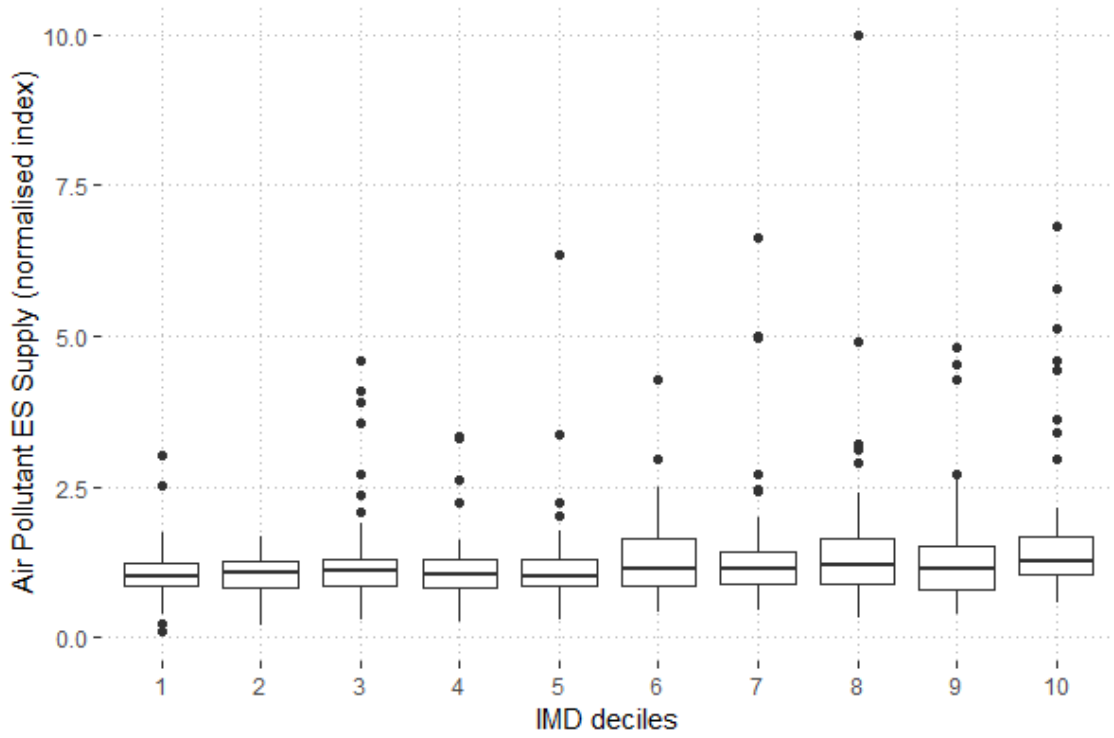


Figure 5.12 Air pollutant removal supply per population weighted IMD deciles for Leeds case study region. Deprivation decreases from decile 1 to 10.

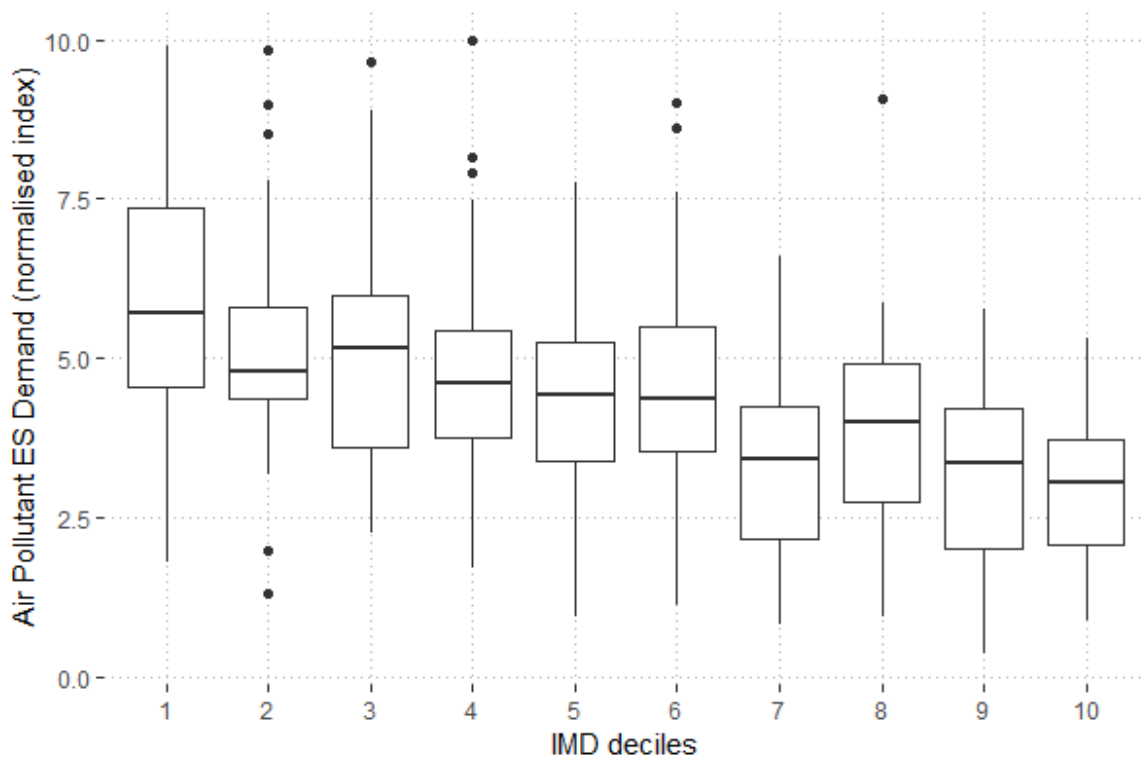


Figure 5.13 Air pollutant removal demand per population weighted IMD deciles for Leeds case study region. Deprivation decreases from decile 1 to 10

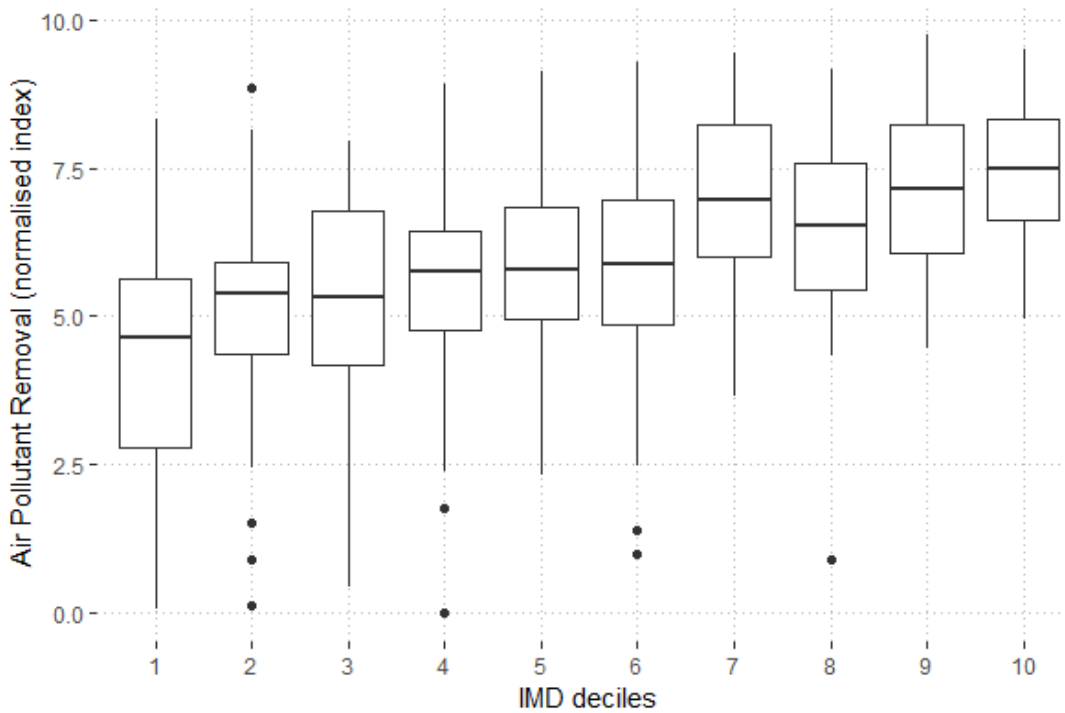


Figure 5.14 Air pollutant removal net ES index per population weighted IMD deciles for Leeds case study region. Deprivation decreases from decile 1 to 10.

Figure 5.15 identifies LSOAs with significant local indicators of spatial associations (LISA) between IMD ranks and net ES index for air pollutant removal. This further confirms the patterns in deprivation and air pollutant removal described above and potentially identifies target areas for prioritising intervention which addresses issues of air pollutant removal if it is deemed desirable to prioritise more deprived areas. Spatially, the LSOAs which experience both higher deprivation and lower air pollutant removal ES are clustered in Leeds' inner city to the north, east and south of the centre (Figure 5.15). Conversely, significant spatial association between low deprivation and high ES is found in the LSOAs located in the city peripheries to the north, east and south. There is only a scattering (11 LSOAs) of areas where local associations are significant between low deprivation and low ES.

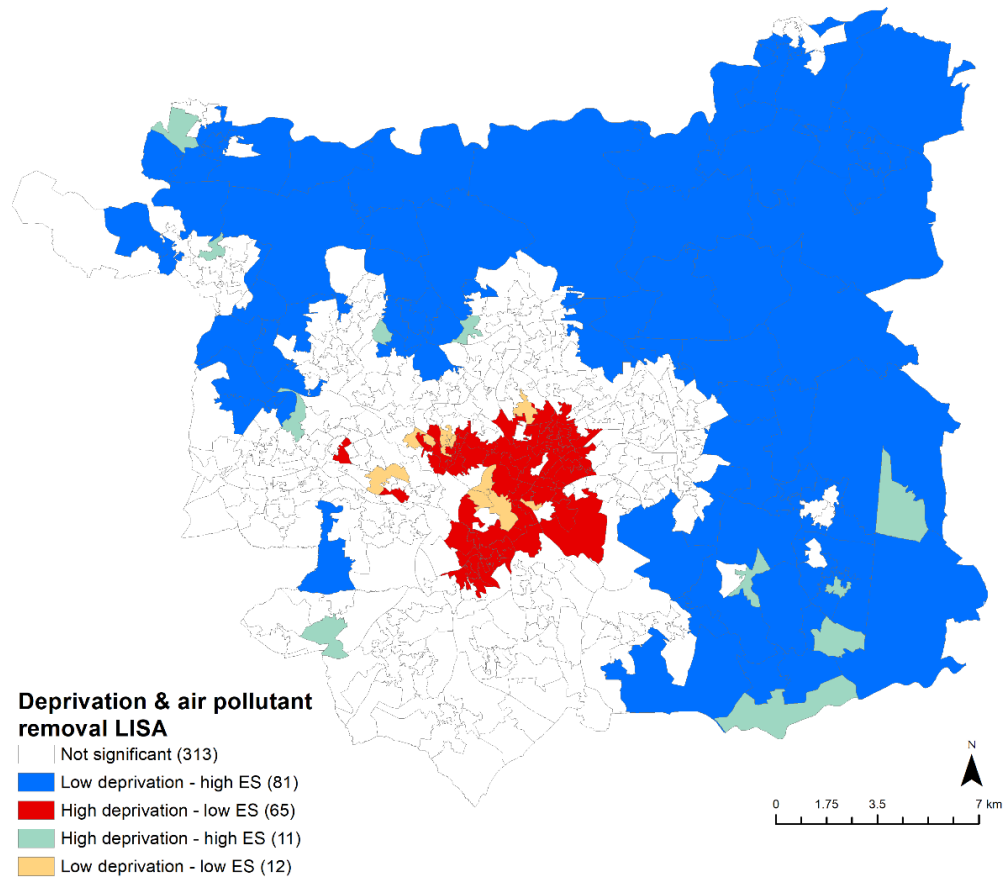


Figure 5.15 Leeds LSOAs with significant local spatial associations between IMD ranks and air pollutant removal. Identifies clusters of areas where there are significant associations between ES indexes and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class.

5.3.2.2 The spatial and social distribution of air pollutant removal in the South Pennines

Highest air pollutant removal by vegetation (Figure 5.16) occurs linearly from the centre of the case study region to the western edge, in particular in Rossendale from Todmorden to Haslingdon; this largely follows the valleys and consequently also the roads. Highest supply is also found in two small areas, specifically on the edge of the towns of Rochdale to the south and Hebden Bridge in the north, in general it is also higher in Halifax. Low supply is evident across large rural areas of the study area. The spatial pattern of demand (Figure 5.17) is parallel to that of supply; low in rural areas and high in the small valley towns and the larger town of Halifax.

However, ES supply values tend to be closer to the maximum value of 10, whilst most demand values are closer to zero. Thus once subtracted from each other to produce a

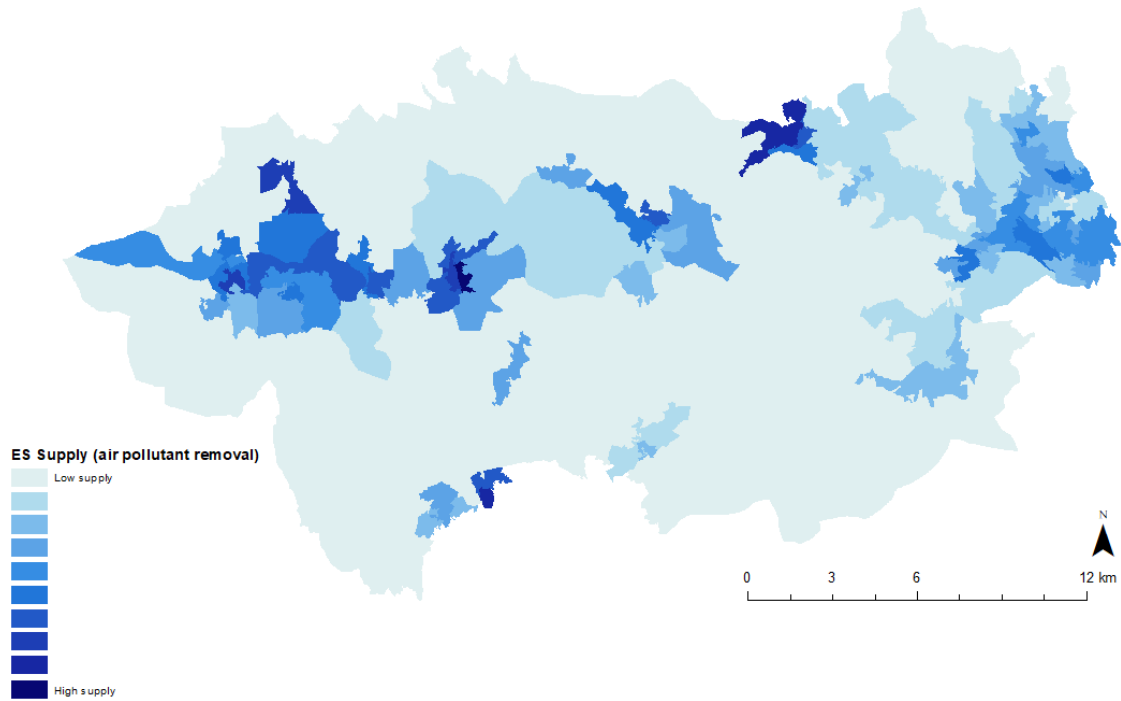


Figure 5.16 The spatial distribution of air pollutant removal supply aggregated to LSOAs across South Pennine case study region

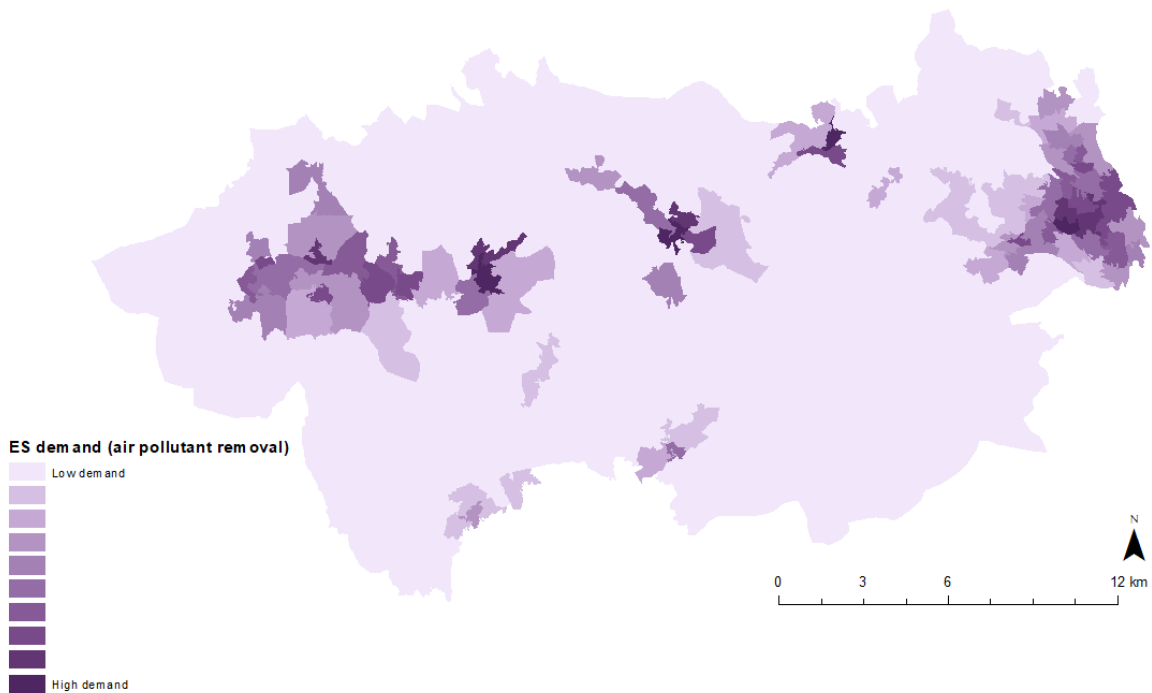


Figure 5.17 The spatial distribution of demand for air pollutant removal aggregated to LSOAs across South Pennine case study region

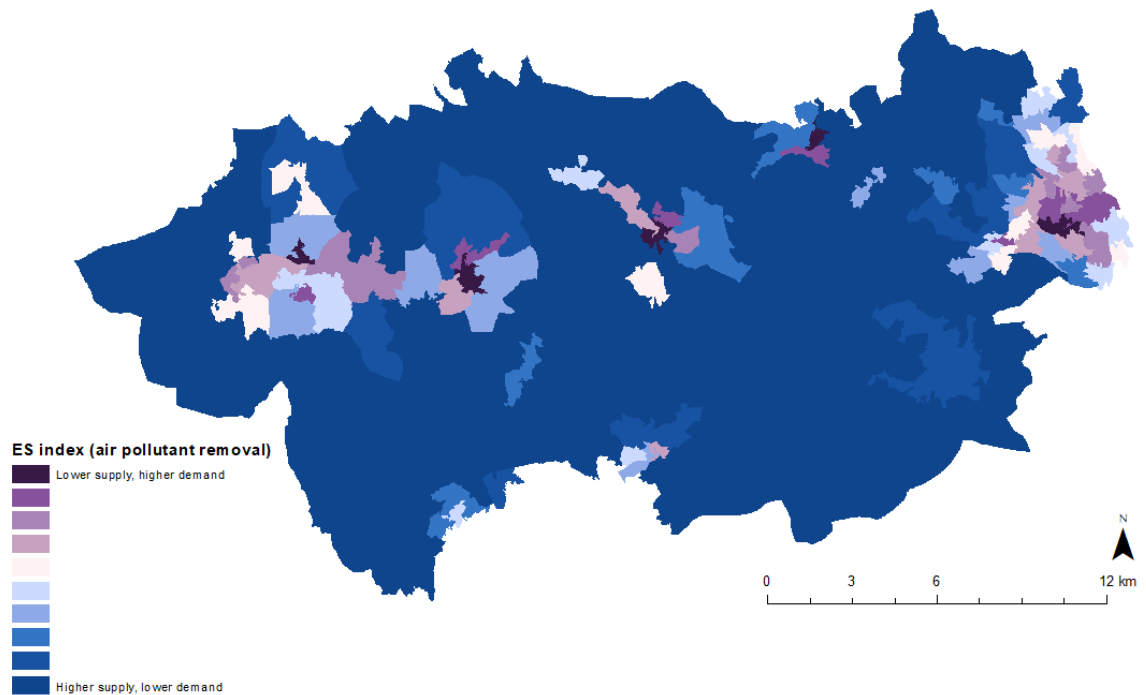


Figure 5.18 The spatial distribution of air pollutant removal net ES index aggregated to LSOAs across South Pennine case study region

relative ES index, relatively higher supply compared to demand is found in the rural areas and supply is lower relative to demand in the more urbanised areas. LSOAs with the lowest net ES index are found scattered across the study area in Bacup, Constable (near Haslingdon), Hebden Bridge and Halifax (Figure 5.11).

There is significant moderate correlation between air pollutant removal ES index and IMD ranks ($\rho=0.444$), thus there is some increase in the ES as deprivation decreases. This is driven almost entirely by the pattern of demand (correlation coefficients between net ES index and demand and supply are -0.999 and -0.711 respectively). ES demand has a moderate negative correlation with IMD ranks ($\rho=-0.434$), indicating that demand increases with deprivation. There is no significant correlation between ES supply and deprivation levels. These findings are reflected in the boxplots (Figure 5.20 and Figure 5.19), which show little variation in ES supply across population weighted deprivation deciles but a higher ES demand for the four most deprived deciles. There is however no clear decrease in demand as deprivation decreases across the other deciles (deciles 5-10). The concentration index for air pollutant net ES index is 0.16, which similarly indicates some inequality in the social distribution of this ES for the South Pennines. The associated concentration curve (Figure 5.30) lies below the line of equality for more deprived LSOAs, indicating that they have a lower share of ES.

Although there is only a small disparity between the concentration curve and line of equality and the relatively low concentration index echoes this, indicating that the inequalities present are not acute.

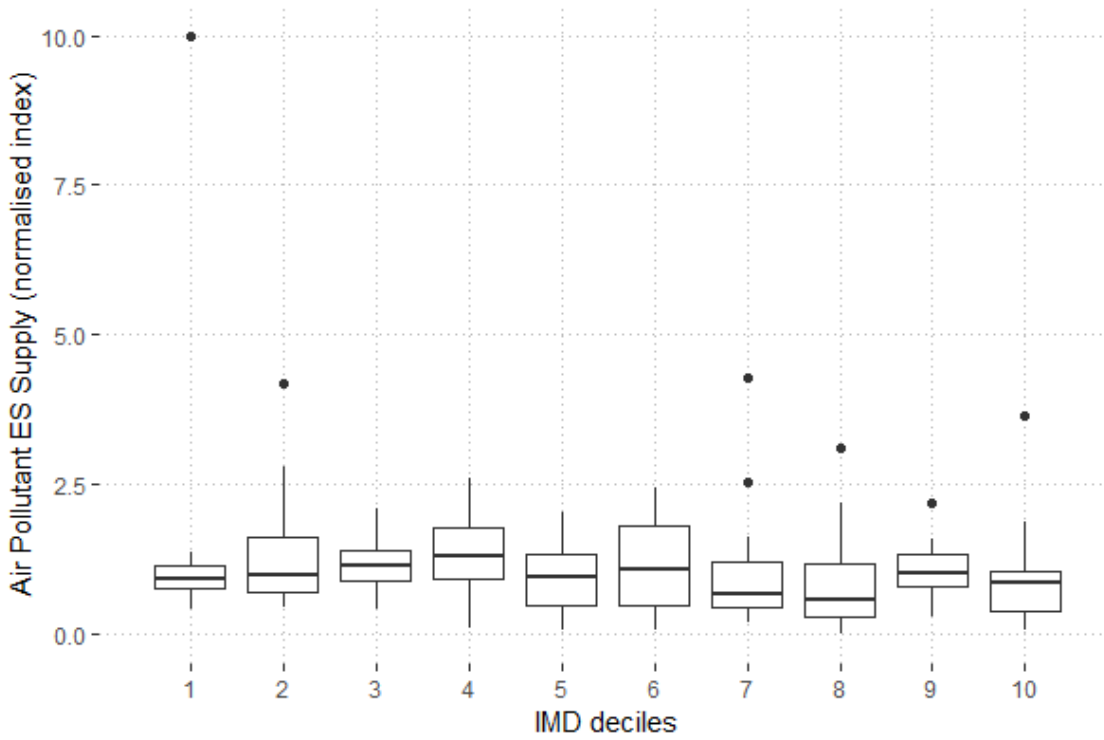


Figure 5.19 Air pollutant removal supply per population weighted IMD deciles for South Pennines case study region. Deprivation decreases from decile 1 to 10.

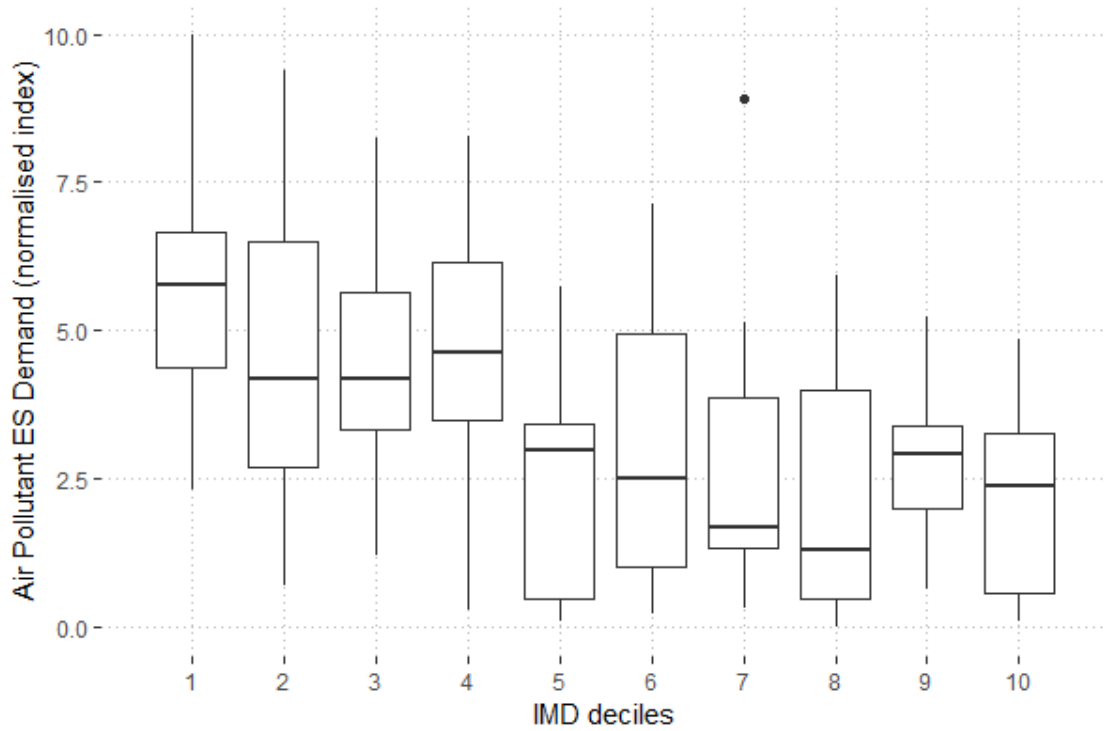


Figure 5.20 Air pollutant removal demand per population weighted IMD deciles for South Pennines case study region. Deprivation decreases from decile 1 to 10.

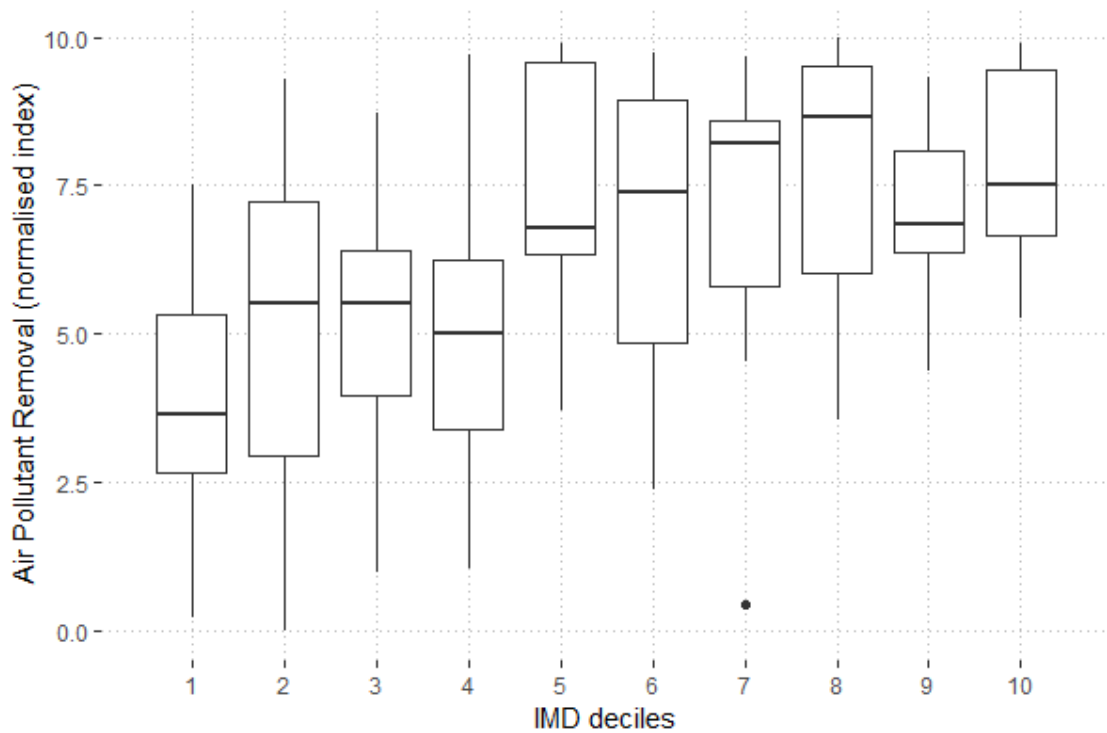


Figure 5.21 Air pollutant removal net ES indexes per population weighted IMD deciles for South Pennines case study region. Deprivation decreases from decile 1 to 10.

Local indicators of spatial association (Figure 5.22) indicate that areas where high deprivation is significantly associated with low air pollutant removal ES (a total of 17 LSOAs) are concentrated solely in the large town of Halifax in the east of the study site and the centrally located small town of Bacap. There are slightly more (22) LSOAs whereby significant local association between low deprivation and high ES are found. Since these are primarily rural, they also cover a large proportion of the South Pennine region.

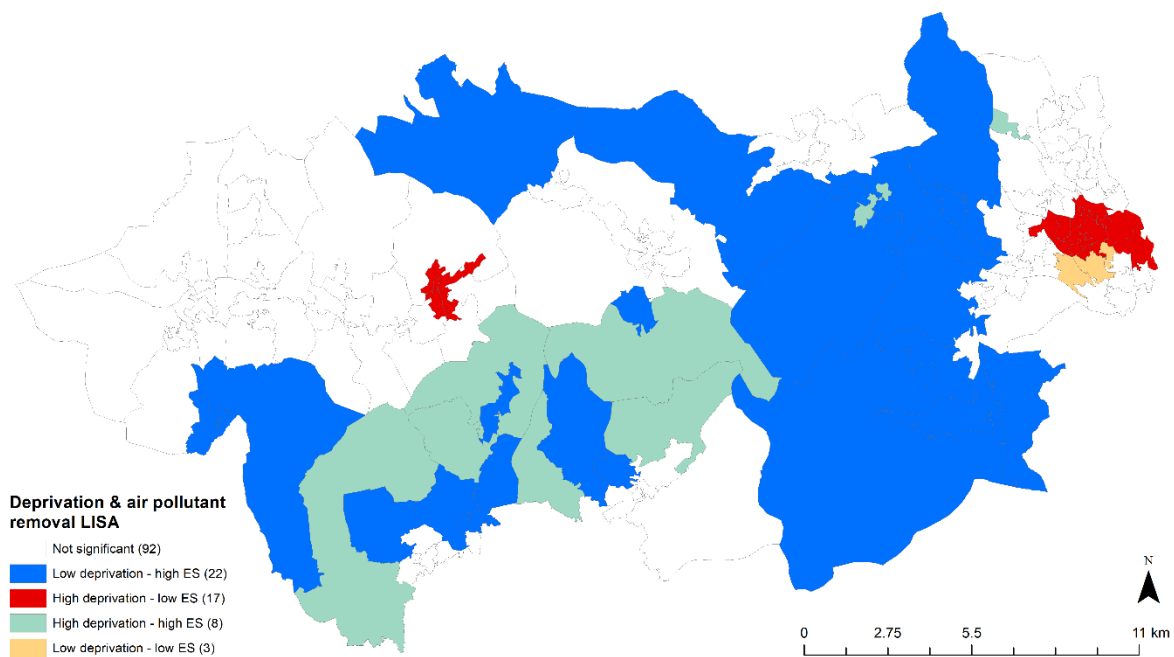


Figure 5.22 South Pennine LSOAs with significant local spatial associations between IMD ranks and air pollutant removal. Identifies clusters of areas where there are significant associations between ES indexes and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class.

5.3.2.3 The spatial and social distribution of air pollutant removal in Northampton

There is a distinct rural-urban spatial pattern in the Northampton region's air pollutant removal. Areas with the lowest supply of (Figure 5.23) and lowest demand for (Figure 5.24) air purification are rural areas lying outside of Northampton town. Conversely, the highest supply and demand is present in LSOAs within Northampton and Towcester.

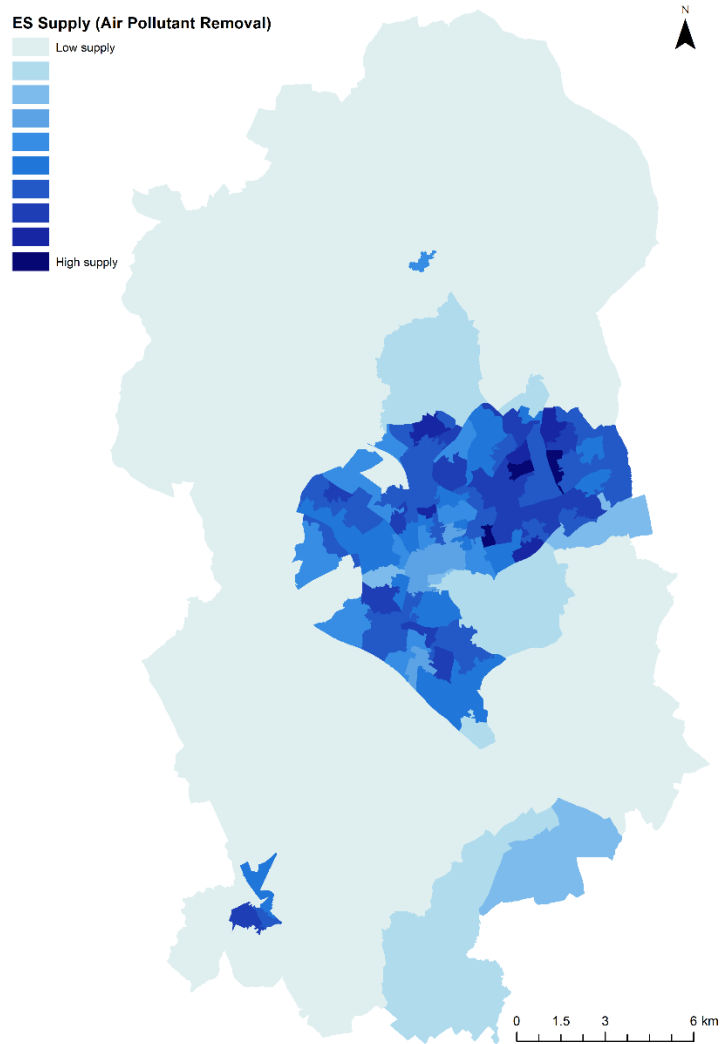


Figure 5.23 The spatial distribution of air pollutant removal supply aggregated to LSOAs across Northampton case study region

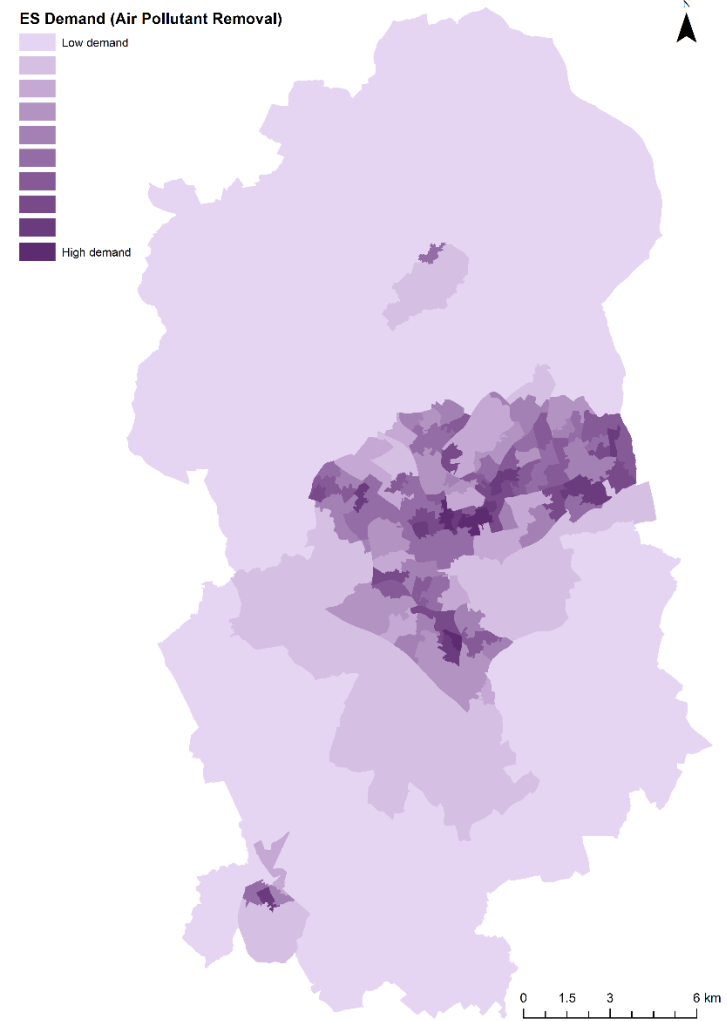


Figure 5.24 The spatial distribution of demand for air pollutant removal aggregated to LSOAs across Northampton case study region

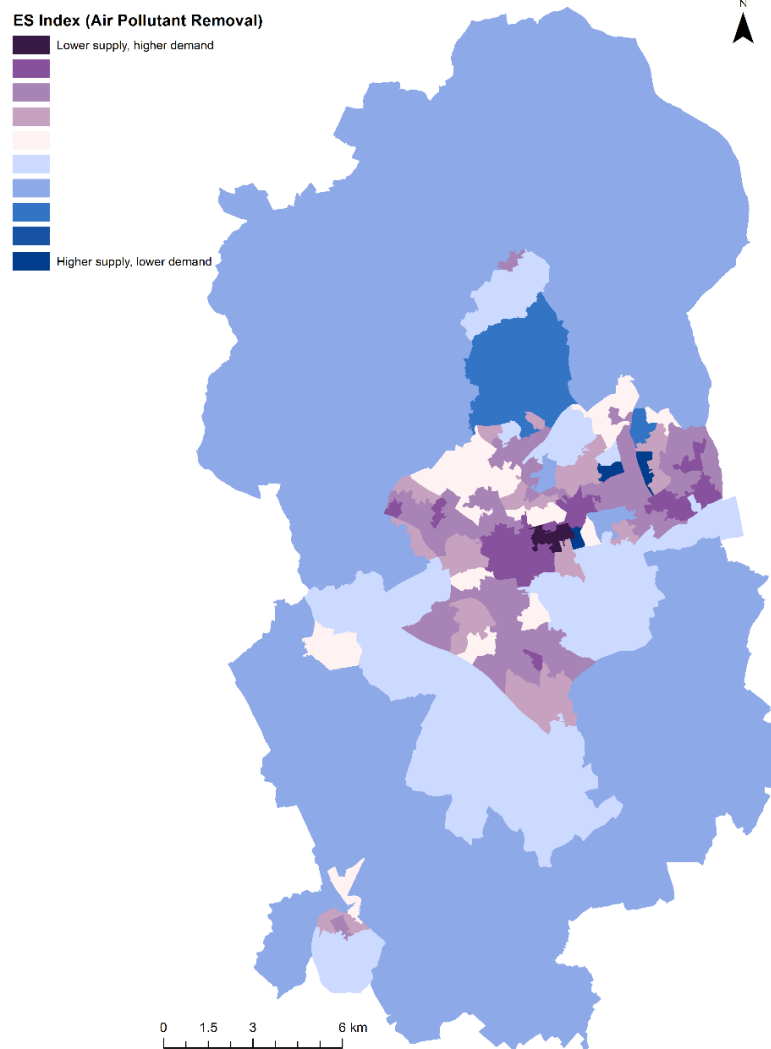


Figure 5.25 The spatial distribution of air pollutant removal net ES index aggregated to LSOAs across Northampton case study region

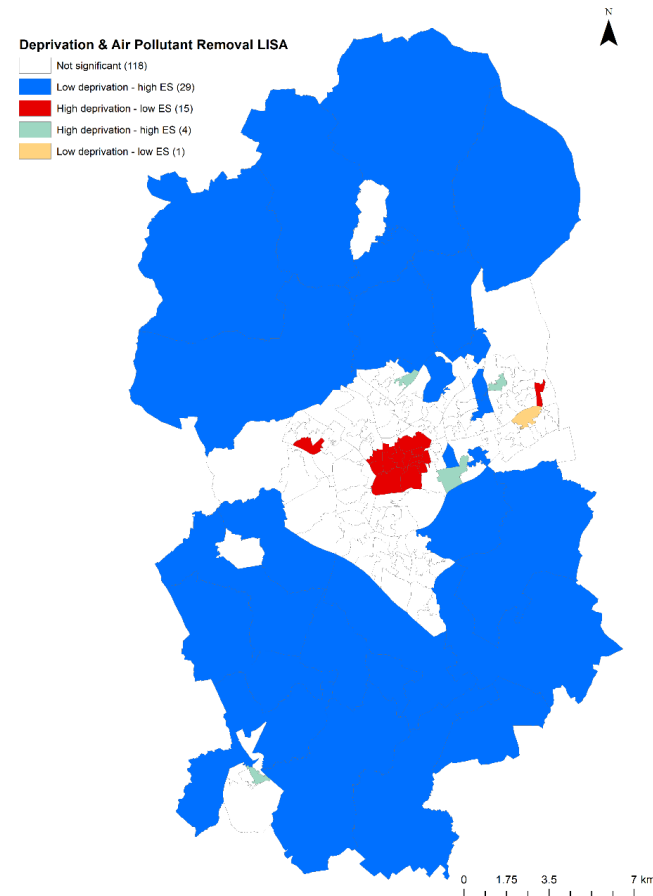


Figure 5.26 Northampton LSOAs with significant local spatial associations between IMD ranks and air pollutant removal. Local indicators of spatial association (LISA) identify clusters where there are significant associations between net ES indexes and IMD ranks in the same direction (95% confidence level). Directions of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class

Lower supply in rural areas can be attributed to fewer trees amongst the agricultural fields compared to the urban parks and gardens combined with its reliance upon demand. ES demand is greater in the urban areas due to the higher density of road network, which dominates the smaller urban LSOAs. Once subtracted to generate the net ES index, this presents a more spatially fragmented pattern (Figure 5.25). Rural areas have moderately higher supply and lower demand in comparison to other areas. LSOAs with the highest supply relative to lowest demand are scattered within Northampton town as are those with the lowest supply relative to highest demand.

Overall there is a significant but modest linear relationship whereby air pollutant removal net ES index decreases as deprivation increases ($\rho=0.322$). The distribution of net ES indexes across IMD population weighted deciles (Figure 5.29) shows the relatively consistent increase in ES as deprivation decreases, but that the most and least deprived deciles (1st and 10th respectively) are inconsistent with this pattern. The median of net ES indexes for LSOAs within the most deprived decile (1st) is higher than those in the 2nd – 5th deciles. LSOAs within the 9th deprivation decile (low deprivation) have the highest ES indexes overall as indicated by the highest median value. Consistent with these findings the concentration curve (Figure 5.30) shows some divergence from the line of equality, suggesting that the 2nd-7th deprivation deciles have less than an equal proportion of that service. The concentration index (0.12) quantifies this, indicating some inequality but that these inequalities are modest.

Local indicators of spatial association (Figure 5.26) show a distinct rural-urban pattern, whereby significant association between high deprivation and low ES is clustered within central Northampton, whilst significant association between low deprivation and high ES is experienced across the majority of rural LSOAs.

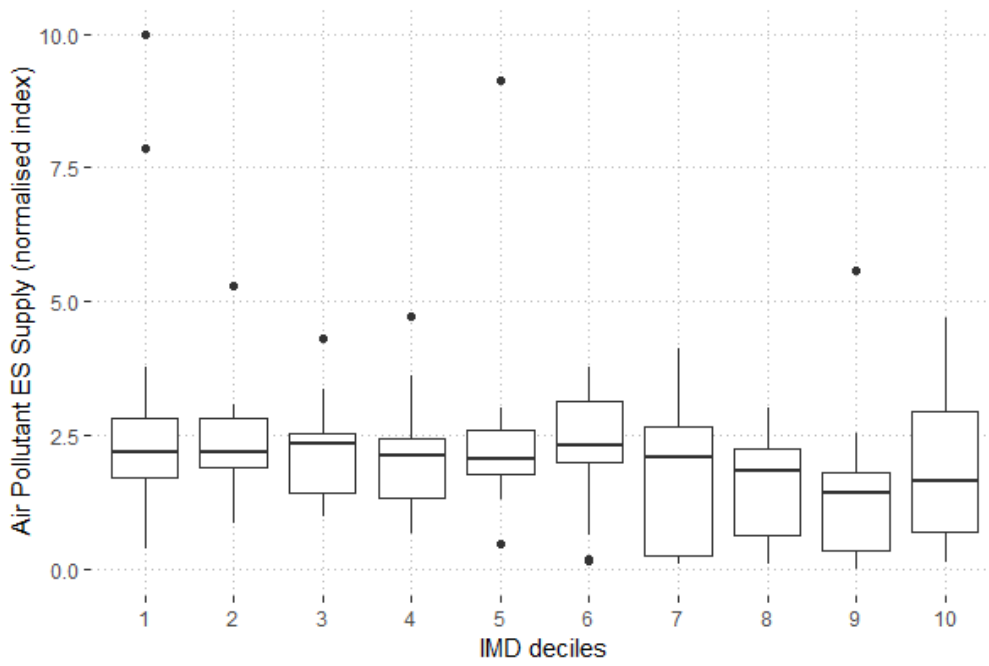


Figure 5.27 Air pollutant removal supply per population weighted IMD deciles for Northampton case study region. Deprivation decreases from decile 1 to 10.

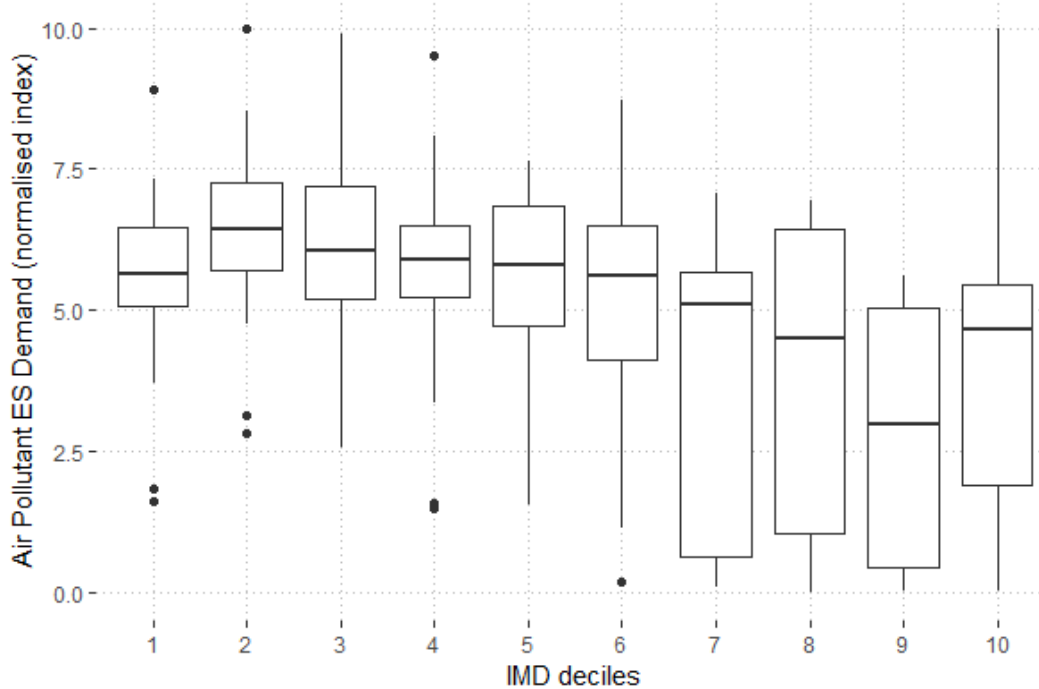


Figure 5.28 Air pollutant removal demand per population weighted IMD deciles for Northampton case study region. Deprivation decreases from decile 1 to 10.

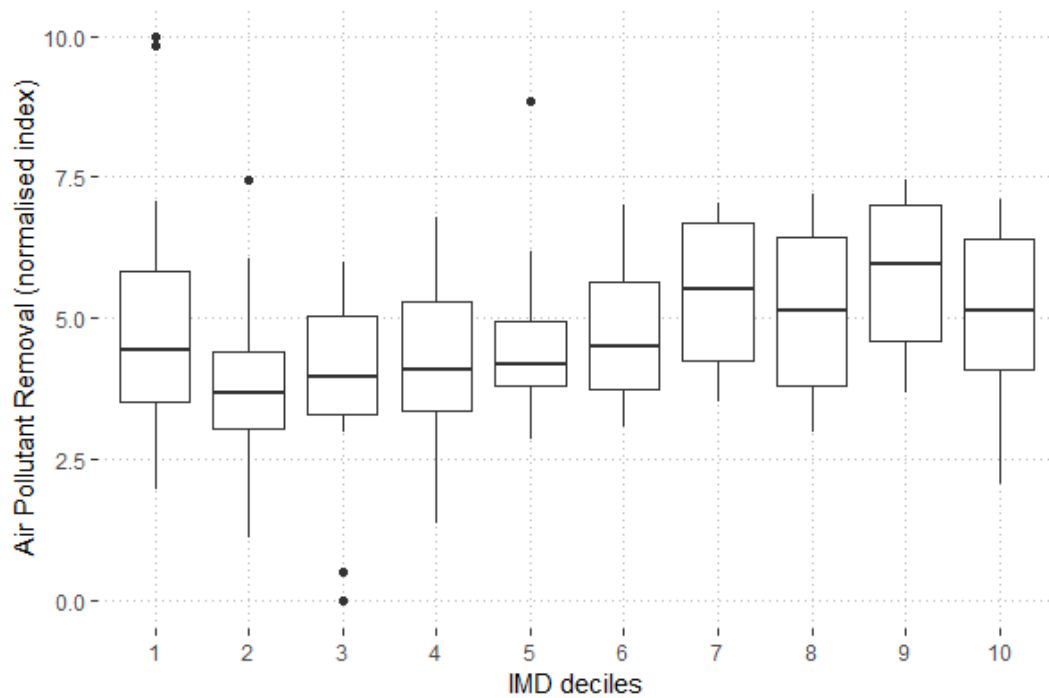


Figure 5.29 Air pollutant removal net ES indexes per population weighted IMD deciles for Northampton case study region. Deprivation decreases from decile 1 to 10.

5.3.2.4 Sensitivity tests

For all case studies, different ES supply parameters (Table 5.1) had negligible effect on the association between net ES index and deprivation. Regarding correlation coefficients maximum variation of ρ is 0.044 (Table 5.3) and maximum variation of the concentration index is 0.02. This indicates that the results are relatively robust to reasonable variations in values (based on other studies) describing the effectiveness of different land cover in supply of air pollutant removal. This confirms the finding that a net ES index in air pollutant removal is unequally distributed in the case study regions. Nevertheless, it is important to note that there other factors which have potential to contribute to this limited sensitivity of the social distribution of net ES index to variations in ES supply estimates including; the consistency in assigning coniferous trees as the most effective scavenger of air pollutants and the weighting of supply by road density which is consistent across all three scenarios, dominance of ES demand not supply in determining the social distribution of net ES, aggregation to LSOAs.

Correlation coefficients given in Table 5.3 indicate there is inequality in the social distribution of greenspace coverage as determined using the GLUD 2005 database for all case studies. This aligns with the correlations determined for the air pollutant

removal net ES Index, although the strength of association is found to be much lower than that for the net ES index for Leeds and slightly lower for Northampton. These findings suggest that the assumption often encountered within environmental justice research that local greenspace coverage is representative of environment benefits available (Chapter 2 section 2.2) is reasonable in consideration of air pollutant removal, but risks underestimating inequalities. Furthermore, this is largely due to the association of the proportion of greenspace with demand for air pollutant removal, and not supply. For example, the proportion of greenspace is very strongly correlated with population weighted demand ($\rho = -0.931$) for the South Pennines. The use of proportion of greenspace as a proxy for environment benefits is further considered with respect to the results for recreation and SWRR ES in the subsequent chapters.

Assuming a utilitarian conceptualisation of demand for air pollutant removal, whereby the number of potential beneficiaries is also accounted for, has a notable impact upon results for Leeds (Table 5.2). However, for Northampton and the South Pennines results were similar to those for baseline net ES indexes. For Leeds, the magnitude of the correlation coefficient between demand and IMD rank only slightly reduces ($\rho = -0.433$ compared to $\rho = -0.489$) as ES demand is population weighted. However, the population weighting impacts upon the spatial distribution of ES demand, and thus once subtracted from ES supply to produce the net ES index, there is a substantial effect on the correlation between deprivation and net ES index. Specifically, the direction of association between net ES index and deprivation changes, thus a net ES index incorporating a population weighted demand is higher for more deprived areas. This suggests that land use change to increase net ES index for more deprived areas would not benefit the greatest number of people.

Conversely, for South Pennines and Northampton the strength of correlation between deprivation and net ES index is reduced slightly once demand is weighted by population density, but significant positive correlation coefficients persist ($\rho = 0.326$ and $\rho = 0.208$ for South Pennines and Northampton respectively). This suggests that changes in land cover to improve air pollutant removal that focus upon more deprived areas would simultaneously reduce inequality in its distribution and benefit more people.

Table 5.3 Spearman rank correlations between air pollutant removal and IMD, for the three case study regions ranks, and including results of sensitivity tests.

(a) Leeds

		IMD ranks	ES supply	ES demand	ES index
	IMD ranks	-	0.144**	-0.489**	0.514**
	ES supply	0.144**	-	NS	NS
	ES demand	-0.489**	NS	-	-0.985
	ES index	0.514**	NS	-0.985	-
Sensitivity	'Tree dominant' ES index	0.516**			
	'Low influence' ES index	0.512**			
	Population weighted ES demand	-0.433**			
	Population weighted ES index	-0.206**			
	% Greenspace (GLUD)	0.239**			

(b) South Pennines

		IMD ranks	ES supply	ES demand	ES index
	IMD ranks	-	NS	-0.434**	0.444**
	ES supply	NS	-	0.733**	-0.711
	ES demand	-0.434**	0.733**	-	-0.999**
	ES index	0.444**	-0.711**	-0.999**	-
Sensitivity	'Tree dominant' ES index	0.442**			
	'Low influence' ES index	0.441**			
	Population weighted ES demand	-0.466**			
	Population weighted ES index	0.326**			
	% Greenspace (GLUD)	0.424**			

(c) Northampton

		IMD ranks	ES supply	ES demand	ES index
	IMD ranks	-	-0.231**	-0.370	0.37**
	ES supply	-0.231**	-	0.656**	-0.286**
	ES demand	-0.376**	0.656**	-	-0.862**
	ES index	0.322**	-0.286**	-0.995**	-
Sensitivity	'Tree dominant' ES index	0.299**			
	'Low influence' ES index	0.366**			
	Population weighted ES demand	-0.417**			
	Population weighted ES index	0.208**			
	% Greenspace (GLUD)	0.227**			

IMD ranks increase as area deprivation decreases; a positive correlation indicates ES increases with decreasing deprivation.

Correlations between IMD and net ES index, supply and demand separately are provided for 'best estimate' calculations. For sensitivity tests (Table 5.1), correlation with only IMD ranks are computed for net ES index given variations in supply ('tree dominant' and 'low influence') and in demand ('population weighted demand'). In addition, for comparison correlation is computed between IMD and percentage greenspace coverage obtained from GLUD.

NS- no significant correlation **Correlation significant at 99% confidence level
*Correlation significant at 95% confidence level.

5.4 The social distribution of air pollutant removal in the case study regions

The results discussed in the previous sections reveal that air pollutant removal ES is unequally distributed across areas of high and low deprivation in all case study regions. In addition to the results reported above Figure 5.30 portrays the similarity in social distribution of air pollutant removal ES for Northampton, Leeds and South Pennines. In all cases the concentration curves lie below the line of equality for the more deprived deciles, converging with the line of equality for the least deprived deciles. Thus there is statistically strong evidence that more deprived deciles receive a disproportionately lower share of the air pollutant removal net ES, although the relatively close proximity to the line of equality and associated low concentration indexes indicate a more modest inequality comparative to other known inequalities, for example in income (ONS, 2019).

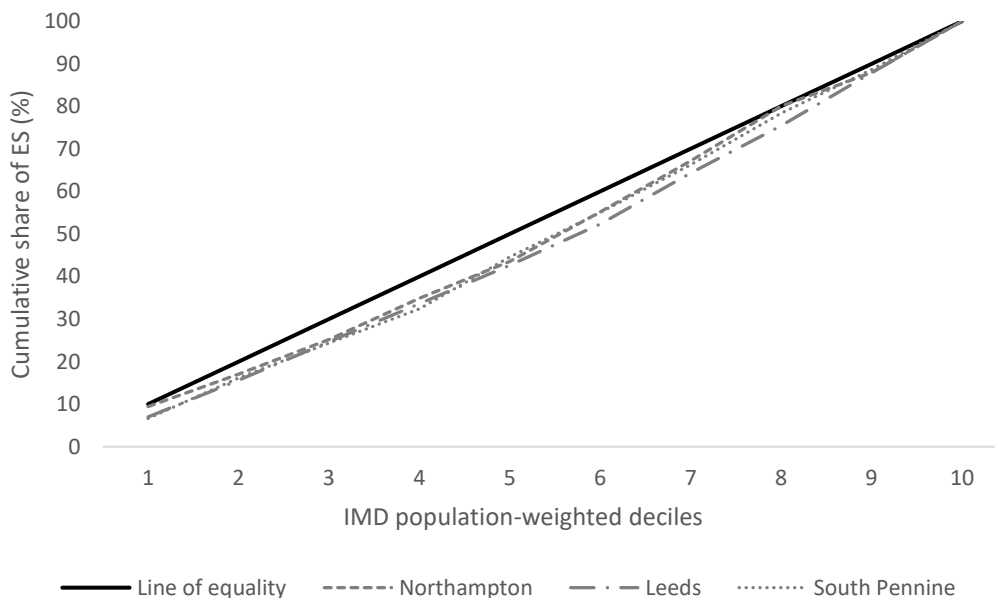


Figure 5.30 Air pollutant removal net ES index concentration curves.

IMD decile 1 represents most deprived population-weighted decile; IMD decile 10 represents least deprived population-weighted decile.

Line of equality shows theoretical equal distribution across all deciles. Concentration curves show actual distribution of air pollutant removal net ES index according to deprivation for each case study. Less than 'equal' share is indicated by the curve falling below the line of equality.

Importantly, it is observed that in all regions the social distribution is driven by the unequal distribution of demand as approximated by proximity to road networks weighted by average traffic flows. It should be noted that IMD 2015 which is used to assess the inequalities does incorporate air quality, it is therefore likely that the

association between ES demand and deprivation is slightly emphasised. However, the results are still considered reliable since the weighting given to air quality is very low (it forms one aspect of the environmental domain which contributes 9.3% to the overall index) and since the results align with findings from previous studies (e.g. Mitchell et al., 2015).

The air pollutant removal which is supplied by vegetation is lower for more deprived areas for Leeds only, and regardless demand is still the dominant factor in determining the distribution of this service. For the South Pennines, there was no significant association between the supply of air pollutant removal ES and deprivation. Whilst in Northampton more deprived areas were demonstrated to have greater supply than the less deprived areas. These findings were consistent under sensitivity testing of uncertainties in the proxies describing the effectiveness of vegetation in removing air pollutants. Nevertheless, given demand is higher in the more deprived areas, interventions which change the distribution of supply to favour the more deprived areas in all case study regions could help reduce the overall inequalities observed. This is illustrated in Northampton where the supply is greater for more deprived areas and helps to offset the higher demand in these areas, although it is acknowledged that this has not been determined in absolute terms.

Given the importance of demand in the social distribution of this ES, the normative position which dictates how demand is conceptualised plays an even greater role. The sensitivity tests revealed that this can (such as for Leeds) change the direction of association between the ES and area-level deprivation. It is therefore crucial, that the normative position is explicit in any social distribution assessment of air pollutant removal ES and with regards to the desired outcomes in increasing this ES in any location.

The social distribution of air pollutant removal as described in this chapter provides an example of a regulating ES which is supplied locally. The implications of this scale factor will be discussed further in Chapter 8. The role of more distant areas on the supply and distribution of an ES is addressed in the Chapters 6 and 7, with the next chapter examining the social distribution of recreation ESs.

Chapter 6 The social distribution of recreation in regional case studies

Recreation is an activity carried out for enjoyment, in leisure time and may or may not involve physical activity. Chapter 2, section 2.1.2 highlighted evidence regarding the importance of recreation in natural environments for human health and wellbeing through, for example, reducing stress or reducing obesity due to physical exercise. The following section examines the supply and demand for recreation as an ES across all three case studies. As outlined in Chapter 5's introduction this includes a description of data, methods and results which describe both the spatial and social distribution of recreation ES and how the latter changes under different modelling assumptions.

It is first necessary to clarify how recreation ES is defined in this research. Here recreation is perceived as a general 'day-to-day' activity which is conducted by a more local population, whilst tourism may occur in distant, more specific locations. There is considerable overlap between the two terms with the difference largely due to the participant (McKercher, 1996), therefore some spaces for local recreation may simultaneously provide opportunities for tourism. Only outdoor nature-based recreation constitutes an ES (i.e. leisure centres and other 'grey' infrastructure provide opportunities for recreation but are not an ES); this includes greenspaces and bluespaces. Provision for recreation ES may be via public or private greenspace or bluespace, however since this study is concerned with inequalities in the context of spatial planning and environmental management, I focus upon public spaces. Importantly, everybody has access rights to these spaces without entry cost. A focus on public areas only is also pragmatic since contemporary data on private garden area and its' vegetation coverage is not readily available nor practical to identify across several large areas. Despite this, private green and blue spaces are relevant for environmental equity and this is acknowledged when the findings here are discussed. Henceforth recreation will refer to recreation ES which takes place in green or blue spaces where the general public have free access rights.

Recreation is one of the most tangible and commonly mapped cultural ES (Hernández-Morcillo et al., 2013). Mapping access to and monitoring the recreational use of green and blue spaces has a long history that predates its attention with the ES discourse (see Butler, 2004 for a review). In England, several organisations, concerned with the health and economic benefits of outdoor recreation, have made notable contributions in this regard (e.g. CABE Space, 2010; Sport England, 2016; Natural England's MENE surveys e.g. 2018a). From an equity perspective, accessibility of parks, woodland or

urban greenspace for recreational purposes is one of the more commonly mapped ES (e.g. Barbosa et al., 2007; Molteno et al., 2012; Woodland Trust, 2017; see Chapter 2 section 2.4 for a more detailed review). Consequently, there are multiple existing methodologies and datasets; these are drawn upon for informing the selection of datasets (discussed in the next section) and for quantifying recreation supply and demand for the case study LSOAs (discussed in section 6.2).

6.1. Data

Mapping recreation service providing areas (SPAs) requires identification of any greenspace or bluespace where the general public can access for free and carry out recreational activities. In England, this is most appropriately achieved using a site based approach where a range of datasets are used to identify all sites and their boundaries which match this description. Alternative approaches use land cover data combined with other indicators of 'quality' or 'naturalness' such as presence of protected areas or water quality. However these tend to cover very large areas (e.g. Maes et al., 2011; Willemen et al., 2008), identify landscape potential (also termed recreation suitability mapping) rather than existing ES supply (e.g. Haines-Young et al. (2012), Kienast et al. (2012), Weyland & Laterra (2014)), or are appropriate given the context of their study area, such as Lankia et al.'s (2015) analysis in Finland where 'Everyman's rights' in principal gives access to all public and private greenspace, although there are constraints on activities..

To fully quantify recreation supply, it is desirable that the SPAs constitute the full range of greenspace types used for a variety of recreation activities relevant to the preferences of individuals of different socio-cultural backgrounds. Identification of recreation sites and corresponding datasets to be included here is based on those identified in UK based research and surveys including Barbosa et al. (2007), CABS space (2010) and Natural England (2018). All types included are listed in Table 6.1.

In addition to sites designated for recreation, other greenspaces which may have a different primary purpose but are nonetheless used for recreation activities are included. This includes, for example, cemeteries which are openly accessible maintained greenspaces for which evidence of everyday recreational use, such as dog walking, has been documented in Scandinavia, a comparable high-income European country (Swensen et al., 2016; Nordh & Evensen, 2018). Informal spaces, such as larger areas of grass in housing estates which may not offer a full variety of functions are often overlooked, but are included here since they may nonetheless be used by children, dog walkers or others.

Public right of ways (PROWs) which include paths, cycleways and bridleways are recorded as the second most popular type of destination for recreation in England, representing 14% of total visits in the 2017/18 MENE survey (Natural England, 2018c). They are of particular importance in this analysis given that both urban and rural areas are included, and in some areas dominated by agricultural land, footpaths may constitute the main access for outdoor recreation. This approach is more representative of actual access compared to those studies which have assigned all open countryside, including agricultural areas, as publicly accessible (e.g. Ferguson et al., 2018).

PROWs also provide access points to inland waterways and lakes. Hegetschweiler et al. (2017) identify 45% studies in their review of recreational ES mapping that account for water or access to water, with identification of footpaths proximal to water accounting as water access points. Thus blue spaces, including lakes, streams and rivers are not separately defined, but are accounted for through the inclusion of PROWs proximal to them. This avoids double counting and is appropriate since access rights are not assigned to all blue spaces and data showing boundaries on where rights are granted are not available for all areas. Additionally, where blue spaces occur within other defined greenspaces such as country parks, they are already included within the boundaries. It should also be noted that none of the case studies are close to a coastline.

Table 6.1 Land uses/land covers where public recreation may take place (service providing areas).

Service providing area	Example activities	Dataset/source
Parks, town or village greens	Walking, playing with children, running, sitting enjoying surroundings, fieldsports	defra.opendata.arcgis.com (Doorstep Greens, Local Nature Reserve, Millennium Greens, National Nature Reserves, Registered Parks and Gardens, Registered Common Land) Ordnance Survey Points of Interest, Ordnance Survey Mastermap Topography: digimap.edina.ac.uk Manual digitisation Google Earth
Path, bridleway, cycleway (PROWs)	Walking, running, cycling, horse riding	www.rowmaps.com or www.openstreetmap.org (PROWs) defra.opendata.arcgis.com (National Trails)
Woodland, forest	Walking, cycling, horse-riding, sitting enjoying surroundings, wildlife watching	england@woodlandtrust.org.uk (Woods for people)
Country park	Walking, running, playing with children, sitting enjoying surroundings, wildlife watching	naturalengland-defra.opendata.arcgis.com
Playing field, recreational area	Playing with children, fieldsports, informal sports	Ordnance Survey Points of Interest, Ordnance Survey Mastermap Topography: digimap.edina.ac.uk Manual digitisation Google Earth
Cemetery, church grounds	Walking, sitting	Ordnance Survey Points of Interest, Ordnance Survey Mastermap Topography: digimap.edina.ac.uk Manual digitisation Google Earth
Open countryside	Walking, running, hiking, wildlife watching	CRoW Act 2000 layers: naturalengland-defra.opendata.arcgis.com
Other informal greenspace	Walking, playing with children, informal sports	Manual digitisation: Google Earth
River, lake, canal	Swimming, fishing, watersports	Ordnance Survey Meridian

Developed from Natural England (2018) and Barbosa et al. (2007). Note that any type of location is included only if the general public have access without charge.

To obtain data regarding all these recreation SPAs requires use of a variety of sources and necessitates generation of some new data (a full list of data sources is given in Table 6.1). Where possible freely available GIS datasets with national coverage are used. These identify the majority of formally designated parks, country parks, nature reserves, woodland and open access land. Whilst such data is relatively plentiful for England in comparison to other countries, comprehensive spatial data on many smaller or informal sites such as cemeteries, recreation grounds and sports grounds is lacking. In an approach similar to Barbosa et al. (2007), these latter SPAs were primarily extracted from OS Points of Interest data combined with feature boundaries defined by OS Mastermap topographic polygons. Validation using Google Earth imagery revealed that many informal spaces, as well as some parks and cemeteries remained unidentified, and were thus manually vectorised to ensure the completeness of the final dataset. Only spaces with visible open access points (verified if necessary in Google Street View or via internet searches) were added.

Recent data were available for the majority of datasets as most are regularly maintained by the relevant organisations. Manually defined areas were based on imagery from 2015 onwards. The final dataset is thus considered representative of current recreation SPAs.

Assimilation of data for analysis required conversion to shapefile format, merging of individual datasets and dissolving of boundaries to ensure areas were not double counted. Data was cleaned to ensure correct topology and clipped to a 10 km boundary around the study areas (a threshold discussed below).

6.2. Methods

Recreation, as with many cultural ESs, requires physical interaction with the ecosystem (Hegetschweiler et al. 2017). Unlike other ESs whereby benefits may be passed onto humans passively, recreation requires a user to actively choose to access the benefits of recreation. This then generates user related movements that constitute ES flow between Service Benefitting Areas SBAs (i.e. where the demand is) and SPAs (the recreation sites) which are not necessarily in the same location (Hegetschweiler et al., 2017). Increasingly, the need to model this ES flow, and explicitly address supply by assessing the accessibility of SPAs, and demand by accounting for the factors which influence the decision to use this ES, has been acknowledged, although conceptualisations of this flow and methods used to assess it vary considerably (Hegetschweiler, 2017). In the next section the approach taken to conceptualising and mapping ES flow and supply is explained, and subsequent sections discuss ES

demand with sensitivity tests applied to address the third objective. Methods applied to test the social distribution of recreation are not outlined below since these are consistent for all ESs and are therefore described in detail in Chapter 3.

6.2.1. Mapping recreation flows and supply

As indicated previously, recreation supply requires identification of accessible greenspace (and/or blue space). The criteria set above states that this includes spaces which are free and open for public use, but 'accessibility' is an ambiguous concept (Lotfi & Kooshari, 2009). It may more broadly be conceived as how 'easily' the greenspace can be used for recreational purposes (Nicholls, 2001). This 'ease of use' is constructed through multiple factors including physical characteristics, human perception and preference, and interaction of these factors, important for ES supply, flows and demand.

A key factor in assessing whether a particular greenspace is accessible is its distance from the potential user (often equated with travel time) (Bateman et al., 1996; Erkip, 1997; Rossi et al., 2015; Liu et al., 2018). Most visits to the natural environment occur to nearby locations (Paracchini et al., 2014). Hooper (2015) found that the recreational service of parks decreases with increasing distance (Liu et al 2017), although there is a threshold distance beyond which the greenspaces would not be considered for use (Spinney et al., 2013). Using distance as the key determinant for modelling the availability of greenspaces for recreation for an area enables a clear method for establishing ES flows between SPAs and SBAs for larger study areas.

Whilst the nuances which in reality shape recreation supply, demand and flows are important, basing recreation supply and flows primarily on distance is a reasonable generalisation across larger areas. Distance has long been considered an important factor in studies of recreation access, value and demand (e.g. Wood, 1961). Indeed more than one third of studies examined in Hegetschweiler et al. (2017) review of approaches to modelling supply and demand of cultural ESs include distance to greenspace. Consideration of the characteristics which influence more subjective notions of accessibility is rarer and would require an understanding of how these are shaped through socio-cultural, economic and demographic factors as well as the psychophysical relationship between the user and greenspace characteristics (Casado-Arzuaga et al., 2014; Hegetschweiler et al., 2017). As Hegetschweiler et al., 2017 emphasise, this requires a linked supply-demand model.

Whilst preferences for some greenspace characteristics are near universal, for example tidiness (Gobster & Westphal, 2004), most are user or place specific. For

example, greater 'naturalness' or biodiversity, is often assumed to be a positive characteristic, but several studies find no evidence between sites with designated conservation status, greater species richness nor 'attractiveness' and their use for recreation (Qiu et al., 2013; Casado-Arzuaga et al., 2014; Dallimer et al., 2014; Andrew et al., 2015; Hornigold, et al., 2016). Moreover as Hornigold et al. (2016) observe, greater biodiversity does not preclude the use of a site for most recreational purposes. There is a wealth of literature attempting to tease apart psychophysical linkages but they often relate to specific areas (e.g. Giles-Corti et al., 2005; Dallimer et al., 2014) or specific greenspace types such as parks (Kaczynski et al., 2014). Generalisations on preferences developed from more comprehensive sources such as the MENE surveys (e.g. Hornigold et al., 2016) are based on existing usage and therefore are linked to existing supply, notwithstanding the recent study by Boyd et al. (2018) attempting to tease apart these factors. Thus findings from this research cannot be appropriately extrapolated for this analysis given its concern with distributional equity. Further incorporation of socio-economic data in the quantification of ES supply could preclude an independent assessment of the social distribution of ES supply and net ES (i.e. there would be some circularity). Moreover, data on these types of characteristics are not available across large areas for a full range of greenspace types. Given these constraints distance is used as the main criterion is a simplification of recreation ES flows, with recognition that this is a simplified model.

Studies which similarly employ a distance based approach vary with respect to the distance considered relevant and how distance is measured. Talen (1998) offers an early example of a methodology for assessing equity in access to parks using spatial analysis but there have since been a wealth of studies generated which build on this, including Barbosa et al., 2007; Oh & Jeong, 2007; Paracchini et al., 2014; Macedo & Haddad, 2015; Zank et al., 2016, and Ferguson et al. (2018). Kim and Nicholl's (2016) review classifies the algorithms used into five main approaches. These include network analysis which models actual travel distance/time from origin to the greenspace point of access (e.g. Comber et al., 2008; Ala-Hulkko et al., 2016), which is considered a more accurate way to model distances (Comber et al., 2008). However this is less applicable in this analysis since LSOAs are used as the minimum spatial unit for the SBAs and therefore there is no single point of origin, and to define one would be especially problematic for larger LSOAs. The network method also overlooks the multiple possible modes of transport, and in particular walking time which is critical for nearby greenspace access, and travel by public transport, and personal vehicle (Chen et al., 2017). A 'minimum distance' approach identifies the nearest greenspace, but discounts use of a variety of greenspaces and so is again less appropriate for larger LSOAs (Kim

and Nicholls, 2016). A ‘container approach’ which calculates the number or areal coverage of greenspace within a chosen spatial unit (e.g. Larondelle & Lauf, 2016; Nesbitt et al., 2019) is commonly used to assess differences across census tracts but does not account for the availability of more distant, although often still nearby, greenspace.

The method applied in this study is most akin to Kim and Nicholl’s (2016) ‘spatial interaction model approach’ whereby consideration is given to a additional criteria such as size, attractiveness combined with distance from origin. This approach is used since it aligns with criteria used by policy guidance on minimum provision of greenspace for England. Specifically, this minimum standard is the ANGSt specification, outlined by Handley et al. (2003). This standard is used to define recreation ES flows and considers two related factors – greenspace size and distance from user - since distances to larger parks tend to be further than for smaller parks (Liu et al., 2017; Giles-Corti et al., 2005), although they also serve the local population. ANGSt guidelines present a minimum standard stating at least one space of a certain size within a defined distance should be available:

- “no person should live more than 300m from their nearest area of natural greenspace of at least 2 ha in size” (Rule 1);
- “there should be at least one accessible 20 ha site within 2 km from home” (Rule 2);
- “there should be one accessible 100 ha site within 5 km” (Rule 3); and
- “there should be one accessible 500 ha site within 10 km” (Rule 4).

(Handley et al., 2003)

The maximum distance used assessed is 10km and this is suitable given that the latest MENE survey reports 82% of visits to the natural environment in England occurring within 8 km (note that the survey reports in miles and visits within 10 km are not defined) (Natural England, 2018c). Importantly, the use of ANGSt encompasses multiple distances, considered more revealing than use of a single distance (Kim & Nicholls, 2016), since it realises the contribution of both the closest and slightly more distant greenspace. For example, findings from MENE 2017-2018 indicate that whilst 42% of outdoor recreation visits occur within 1 mile (1.6 km), a further 40% occur within 5 miles (8 km). Other studies in England which have in some way adopted the use of ANGSt guidelines to assess access to public greenspace include Comber et al. (2008) for Leicester, Holt et al. (2015) for Sheffield and Ferguson et al. (2018) for Bradford. Larondelle & Lauf’s (2016) study in Germany offers an example of a policy-led

approach to determining recreation supply in a different country through application of Berlin's policy requirement of 6 m² per capita of close to home greenspace, where close to home is specified as within 500 m (Senatsverwaltung für Stadtentwicklung und Umwelt, 2013). More widely, Hegetschweiler et al. (2017) find that over half of studies researching recreational ES supply and demand use data regarding the size and/or shape of the recreational spaces to determine ES supply.

To implement the ANGSt specification, distance is measured using Euclidean distance calculated for multiple buffers around each LSOA boundary. The boundaries of LSOAs are used as opposed to the centroids as some of the rural LSOAs cover large areas. The use of buffers has been applied in several studies such as Nicholls and Shafer (2001), Ferguson et al. (2018), Wüstermann et al. (2017). Using the SPAs as defined in section 6.1, the percentage of total area of all SPAs within each buffered distance that meet the respective specified minimum size is calculated and rescaled to a 0-10 scale.

ANGsT does not cover requirements for the provision of PROWs, yet as highlighted in section 6.1, their inclusion is important given their high usage and role in providing access to open countryside and bluespace. Since it is likely that footpaths beyond the immediate LSOA are used, an appropriate distance for considering the provision of PROW needed to be identified. The furthest (10 km) buffer is used since the MENE survey (Natural England, 2018b) reported 87% visits along paths/bridleways occurred up to 6 miles (10 km) in 2013-2016, compared to 69% for up to 3 miles (5 km). The distance of 10km is consistent with the limit of the ANGSt guidelines for greenspaces and thus it is appropriate to combine both the footpath and space analysis within a single ES index. All PROWs within a 10 km buffer of each LSOA and located within or alongside natural land cover (including bluespace) are accounted for. The PROW data is filtered using a 5 m buffer around non-manmade land cover in OS Mastermap Topographic vectors and those running through the identified greenspaces are omitted to avoid double counting. The length of PROW per km² of the buffered LSOA area is calculated, which accounts for the diverse sizes of LSOAs, this figure is then normalised to a 0-10 scale.

The final recreation supply value is calculated through weighted addition of the normalised PROW per km² and greenspace per km² values:

$$\text{Recreation net ES index} = 0.75 \times \text{Greenspace}_{nm} + 0.25 \times \text{PROW}_{nm}$$

The weighting aligns with the assumption that greenspaces are the most important locations for recreation, and that they also offer opportunity for a greater variety of activities. The specific weighting was based on the different proportion of visits according to MENE survey results from 2013-2016 whereby visits to paths/bridleways

and farmland (it is assumed that farmland is primarily accessed using PROWs) constitute 24% of all visits recorded (Natural England, 2018b).

6.2.2. Recreation sensitivity testing

Sensitivity tests can reveal how the metrics indicating the social distribution change according to how recreation supply is conceived or quantified. Here sensitivity tests are applied to understand how the two main types of recreation SPAs and the criteria for measuring access to them influence the findings.

ES supply is based on two main types of SPAs: PROWs and greenspaces. Since these fulfil different functions and are managed in different ways, it is pertinent to explore the social distribution of these separately. This not only provides greater knowledge regarding the social distribution of recreation within the study areas but also substantiates findings given the selection of a single weighting for combining these into a single index. Furthermore PROWs are often overlooked in studies assessing recreation equity, thus comparing the distributions of each in addition to their combined value could also help inform methodological choices in future work.

The key factor in defining ES supply is the use of the distance criteria with previous studies showing that the distribution of recreation is sensitive to distance (Comber et al., 2008; Kim & Nicholls, 2016). Therefore additional quantification of ES supply for three separate distances is carried out; within LSOAs only, within 1 mile (1.6 km) from LSOA boundary and within 2 miles (3.2 km) from LSOA boundary. Assessment at the LSOA level is important since many studies report the distribution of greenspace with respect to its coverage over census tracts with the implication that this is representative of access to ESs, including recreation (e.g. Larondelle & Lauf, 2016; Nesbitt et al., 2019). The 1-mile and 2-mile distances represent 42% and 25% of all trips to the natural environment as reported by the 2013-16 MENE surveys (Natural England, 2018b). They are also considered representative of 'walking distance', important given that 63% recorded visits were made on foot for the same time period (Natural England, 2018). Furthermore the area of parks within 1 mile has been linked to greater physical park use and is thus important for health outcomes (Kaczynski et al., 2014). Only a single distance is used in the calculations which are again measured using buffers with all greenspaces above 2ha in size and footpaths included. The use of different distance criteria is methodologically important since whether or not each of these reveal a similar social distribution can help shape development of future assessments and facilitate comparisons to findings from other studies. Finally, it can enable greater consideration of the equity of recreation distribution for all groups since it is known that

socio-economic, cultural, health and demographic factors can influence distances travelled to greenspace.

Although additional sensitivity tests could investigate other methodological decisions (e.g. potential errors in input data, use of buffers to estimate travel distance), additional scenarios or subsets of data. It is believed that the tests outlined above will provide a strong insight into the robustness of the results and the implications of methodological decisions for policy and other research.

6.2.3. Mapping recreation demand

Demand for recreation ES is typically modelled with respect to the number of people requiring the services, people's socio-demographic and socio-economic characteristics and their individual values and preferences, or the actual recorded use of a greenspace (Wolff, 2015; La Rosa et al., 2018). As highlighted in section 6.2.1, this creates an ES supply-demand interdependency whereby actual characteristics of greenspaces (e.g. their facilities, condition and accessibility) contribute to the demand for recreation (Brainard et al., 2001; Plieninger et al., 2013; Hegetschweiler et al., 2017).

Hegetschweiler et al (2017) find that when supply and demand are linked (58% of studies identified in their literature search) they use on site questionnaires, or interviews combined with GIS data. These approaches are less appropriate for larger scale studies across a range of landscapes in different locations, especially when data on characteristics of all the greenspaces is lacking. Whilst there are examples of studies across large areas which have estimated recreation demand based on a population's socio-economic characteristics, these are based on knowledge of current use (through for example MENE survey) which are influenced by the current ES supply and may therefore not be a true representation of people's desires or needs. Recent work by Boyd et al. (2018) has started to disentangle this, seeking to understand the factors influencing lack of visitation to the natural environment controlling for environmental conditions.

In this thesis recreation demand is, in the first instance, considered uniform, regardless of social-economic profiles and population density. Several factors informed the decision to not base demand on socio-economic, cultural or demographic characteristics of populations. Specifically, that it is not considered appropriate to make assumptions across the study areas, data availability is a limiting factor, and such an approach would also confound subsequent comparison to deprivation data for equity assessment. Of concern is the assumption that particular socio-demographic groups have lower demand for accessible greenspace which could result in underestimating their demand (e.g. Natural England (2018a) finds that those from lower socio economic

groups have been shown to make fewer visits to the natural environment). Furthermore the characteristics of sites have not been found to influence demand in consistently the same way across different studies (Dallimer et al., 2014; Hegetschweiler et al., 2017). The approach taken here is in line with numerous studies mapping recreation which simply examine physical availability of greenspace and do not examine characteristics nor preferences similarly (e.g. Laurendelle & Lauf, 2016). Nonetheless, the many socio-economic and cultural factors which shape recreation demand are important for equity and will be reflected on in the interpretation and discussion of results.

Although recreation demand models are most commonly based on population distribution (e.g. Bateman, 1995; Lankia et al., 2015; Larondelle and Lauf, 2016), such studies are based on a utilitarian conception of demand. By not assuming population density shapes recreation demand it is assumed that each member of the population has an equal requirement for recreation whether they live in a high or low populated areas. This is also in line with the application of ANGSt in quantifying recreation supply, since this is intended as a minimum standard, and is consistent with the approach used for the other ESs in this study. However, as noted in section 3.3.1.2, the effect of this assumption on the social distribution of recreation ES is also tested. This involves additionally mapping recreation demand as population density and including this within an alternative calculation of net ES index. The population weighted net ES index is computed by scaling LSOA population density values between 0-10 and subtracting these from ES supply.

6.3. Results

The following sections present the main results from analysis of the social distribution of recreation for all three case study regions. This comprises: maps of the spatial distribution of the net ES Index for LSOAs and of the association with deprivation using Local Indicators of Spatial Association (LISA); statistics indicating the association (or lack of) between recreation and deprivation including correlation coefficients and concentration indexes; and graphs to provide further insight including boxplots and concentration curves. Findings for Leeds, Northampton and the South Pennines are described in turn, providing the key information in relation to objective 2. Subsequently for objective 3, the results of sensitivity testing are described for all case studies together. A summary synthesising similarities and differences in how recreation ES is socially distributed across all three regions concludes the recreation analysis.

6.3.1. The spatial and social distribution of recreation in Leeds

In terms of spatial distribution, there is a clear gradient in recreation with highest supply in the northwest of the study area and lowest supply in the east (In terms of spatial distribution, Figure 6.1). This is likely due to the proximity of the north western areas to larger open access spaces in the Yorkshire Dales and the South Pennines. The higher values continue in a south eastern direction covering large areas of Leeds city, due to the higher number of smaller, more local spaces in the form of city parks. Conversely the land cover in the east is more rural but dominated by agricultural fields where accessible recreational areas are sparser.

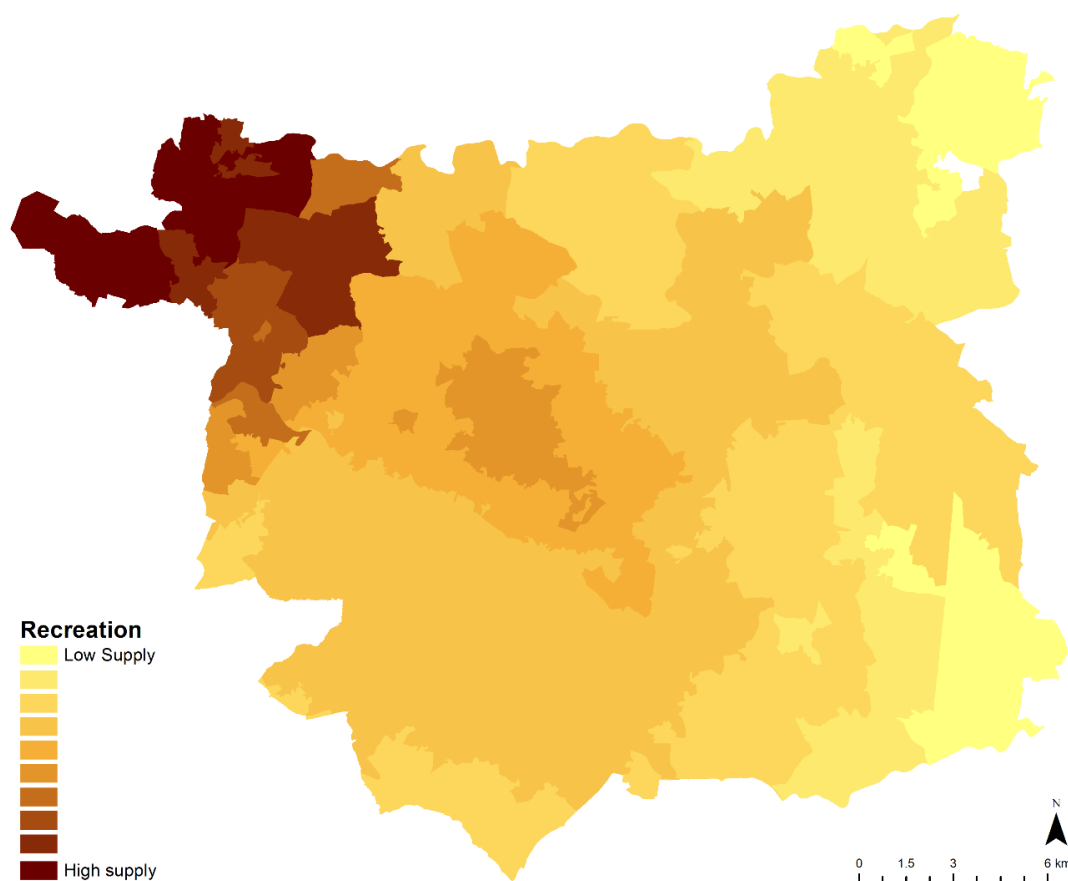


Figure 6.1 The spatial distribution of recreation ES aggregated to LSOAs across Leeds case study region.

The dominant spatial pattern of deprivation in Leeds is radial and thus when set against the dominant east—west pattern of recreational ES, there is no clear social distribution of recreation. There is no significant correlation between the recreation index and IMD ranks (Table 6.2), the concentration index is 0.01 as visualised by the concentration

curve closely following a line of equality (Figure 6.11) and ANOVA tests indicate no significant difference in mean recreation across deprivation deciles. For the majority of LSOAs (378 out of a total 482) there is no significant association between IMD ranks and recreation when Local Moran's I is computed. Together these indicate that overall there is a relatively equal distribution of recreation across the Leeds study area.

However, once the separate types of SPAs are examined separately, it's evident that there are differences in recreation ES for areas of high and low deprivation. When only greenspaces are considered, there is weak but significant correlation, meaning that the more deprived areas have lower access to recreation. Conversely, there is greater availability of PROWs in the more deprived areas, due to a large network of footpaths in the urban areas (with natural land cover).

Interestingly, recreation accessibility is more variable for the least deprived areas, likely due to these areas being located in both the northwest and east of the study area. For instance, the boxplot (Figure 6.2) shows relatively consistent median values across each population weighted deprivation decile, but the range of values (considering both the interquartile range and outliers) increases as deprivation decreases.

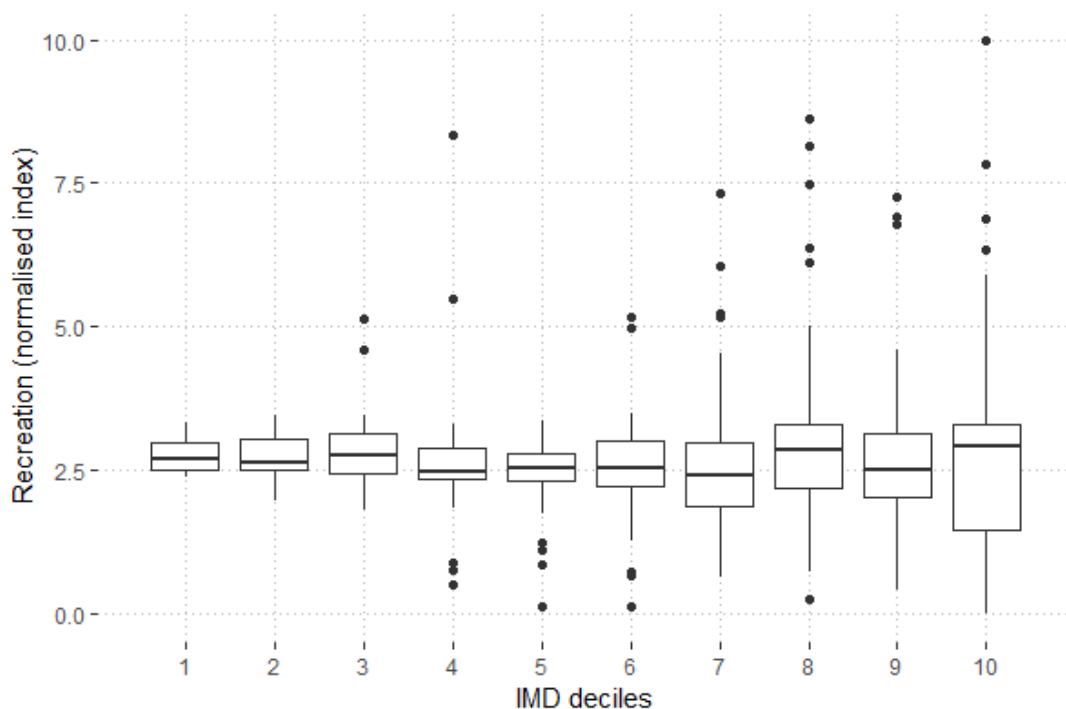


Figure 6.2 Recreation ES indexes per population weighted IMD deciles for Leeds case study region. Deprivation decreases from decile 1 to 10.

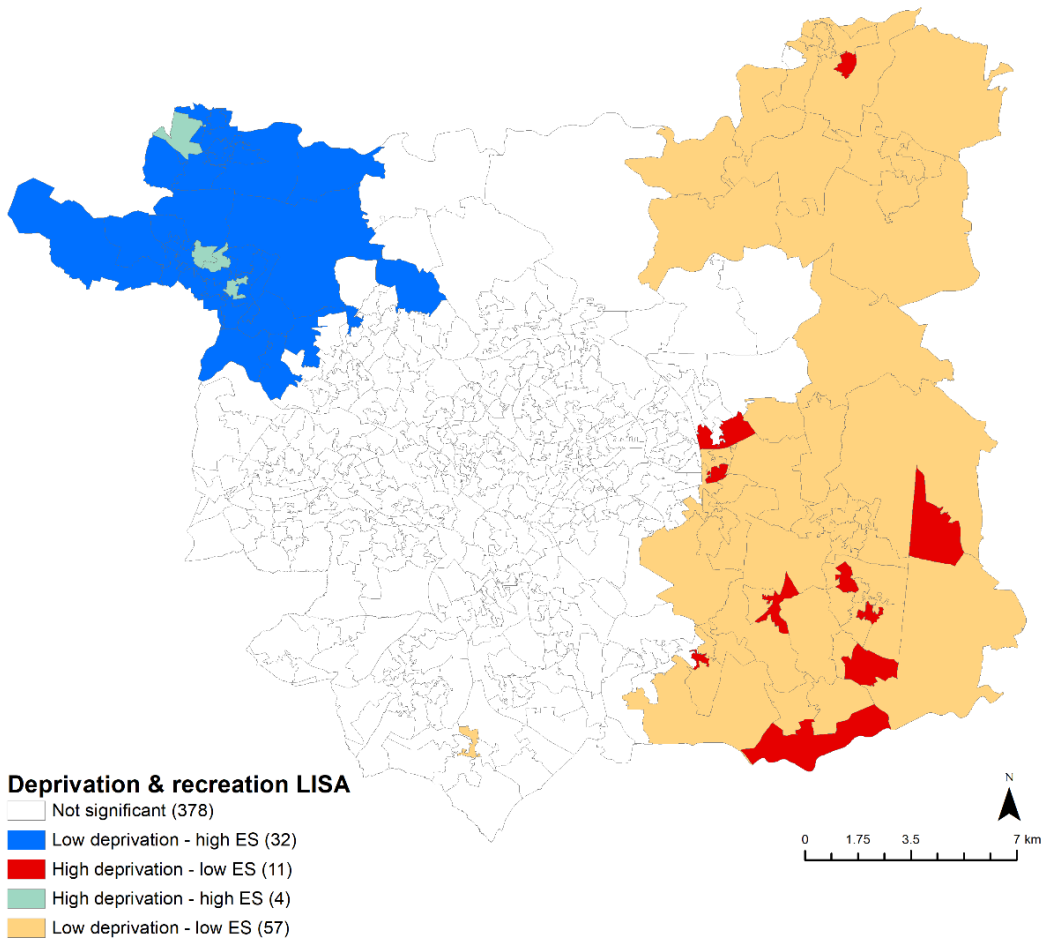


Figure 6.3 Leeds LSOAs with significant local spatial associations between IMD ranks and recreation. Identifies clusters of areas where there are significant associations between ES indexes and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class.

This is further exemplified by the local indicators of spatial association (Figure 6.3). For example, there are a relatively high number (52 out of 482) of LSOAs with significant local association between low deprivation and low ES supply. These are solely located in the east of the study area. Only 11 out of 482 LSOAs are found to have a significant local association between high deprivation and low ES. Principally these are scattered in the south east of the study area. Even so, it is pertinent to note that even fewer LSOAs (4) have a significant local association between high deprivation and a high recreation (compared to 32 LSOAs with low deprivation and high ES).

6.3.2. The spatial and social distribution of recreation in the South Pennines

High recreation in the South Pennines case study region is found centrally stretching from north to south across many of the rural LSOAs and is lowest in the east across the more densely populated LSOAs in the large town of Halifax (Figure 6.4). The high supply in the central areas is created by the large (>500ha) openly accessible moorlands of the Pennine uplands which run in a north-south direction across northern England and across the centre of the study area. There are further open access spaces in the West Pennine Moors to the west of the study which also contribute to the recreation supply for LSOAs in the west of the study area, compared to Halifax in the east which although it has many smaller spaces for recreation has large conurbations within the 10 km radius to the east. The mean ES index for rural areas is 5.3 compared to 3.2 for urban areas (scale 0-10).

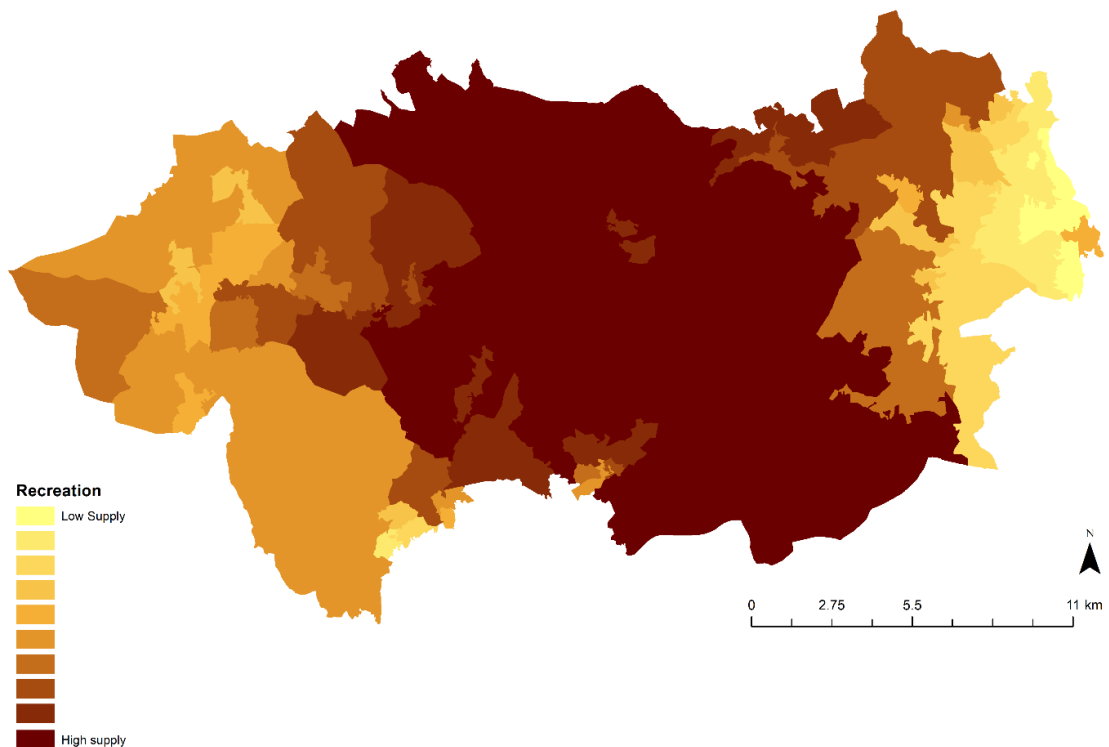


Figure 6.4 The spatial distribution of recreation ES aggregated to LSOAs across South Pennines case study region.

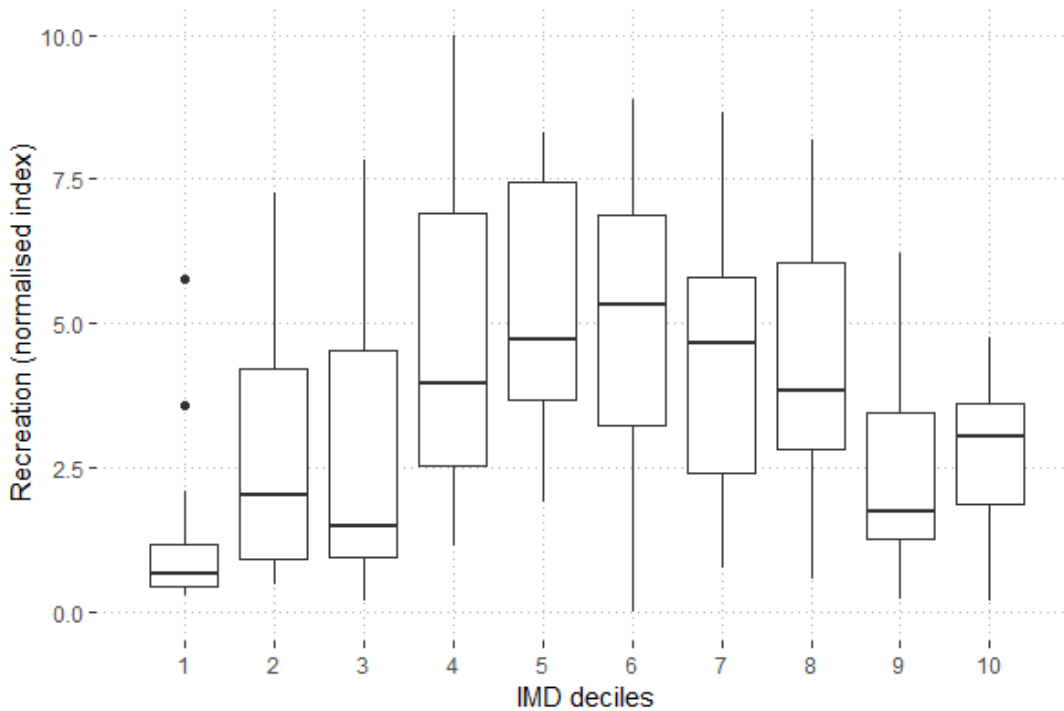


Figure 6.5 Recreation ES indexes per population weighted IMD decile for South Pennines study area. Deprivation decreases from decile 1 to 10.

Considering the social distribution of recreation in the South Pennines, the most deprived LSOAs within the case study are concentrated in Halifax and in rural areas around the centre and west of the study area. Whilst the least deprived LSOAs are scattered in the south west and south east. Thus the spatial pattern of deprivation is not wholly coincident with the spatial pattern of recreation. This is reflected in the significant but weak Spearman Rank correlation coefficient ($\rho = 0.207$) and a concentration index of 0.04. The differences in recreation are confirmed as significant by ANOVA tests of the mean ES index of deprivation deciles. Examination of PROWs and greenspaces separately reveals that the distribution is driven solely by greenspaces ($\rho = 0.287$) whereas there is no significant correlation between PROWS and IMD ranks (Table 6.2).

Plotting the distribution of recreation indexes across IMD deciles, as shown in Figure 6.5's boxplot, reveals a non-linear relationship where low recreation indexes are experienced by both the most and least deprivation deciles. Recreation values peak for the 4th – 8th deciles. Correspondingly, the concentration curve fluctuates so that for the four most deprived deciles the curve lies below the line of equality (a less than 'equal' share), before rising for the deprived 5-8th deciles, indicating their greater 'share' (Figure 6.11). Local Moran's I calculations reveal significant association between high deprivation and low recreation for 29 of 142 LSOAs, all concentrated in eastern urban

LSOAs in Halifax. In contrast, significant local spatial association between low deprivation and low recreation are found in several areas on the outskirts of Halifax. Where recreation is highest in the central region of the study area there is significant local spatial association with both high and low deprivation (21 and 19 LSOAs respectively) (Figure 6.6).

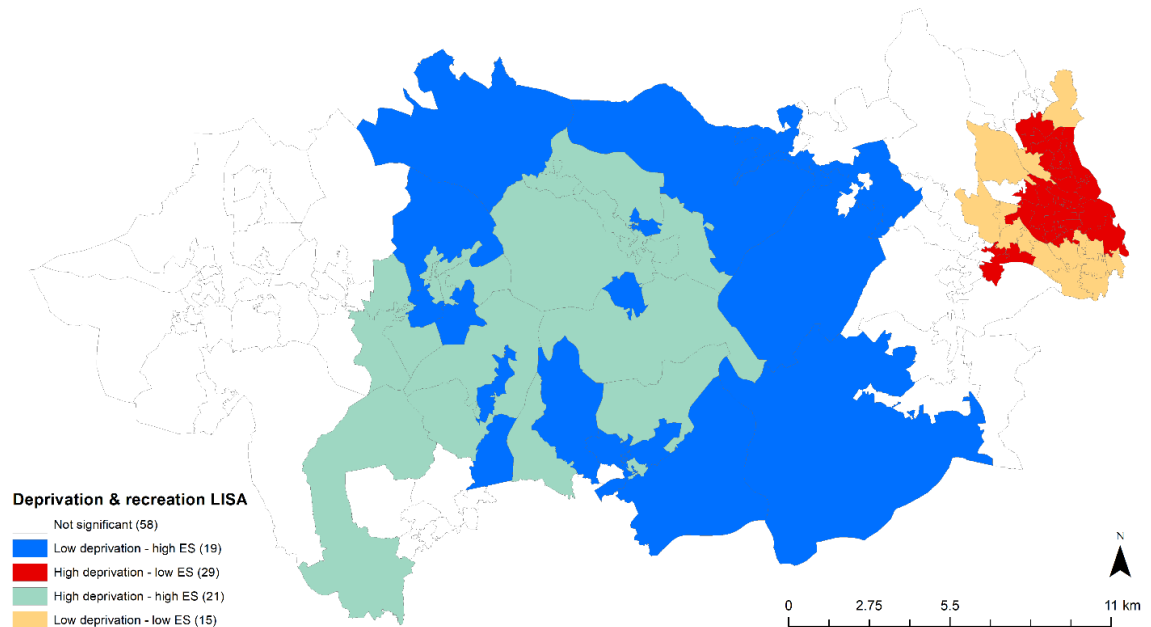


Figure 6.6 South Pennine LSOAs with significant local spatial associations between IMD ranks and recreation. Identifies clusters of areas where there are significant associations between ES indexes and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class.

Interestingly, these results indicate that access to public outdoor recreation spaces is lowest for those in the most deprived areas as well as those in the least deprived deciles. Since this was contrary to expectation, it was pertinent to further explore whether these areas also had similar private garden space. It was expected that the least deprived deciles have more expansive private spaces which in some ways compensate for lower public spaces. Whilst it's not feasible to produce a detailed mapping of garden area across the entirety of the case studies, there is an existing database (GLUD) which does give some indication of garden coverage (in 2005) which can be supported by an assessment of housing types (with detached and semi-detached housing typically providing greater garden area than terraced housing/apartments). Average garden area was calculated from GLUD 2005 and Census 2011 data as 'total garden area/total number of dwellings' and plotted as a boxplot against deprivation deciles (Figure 6.7).

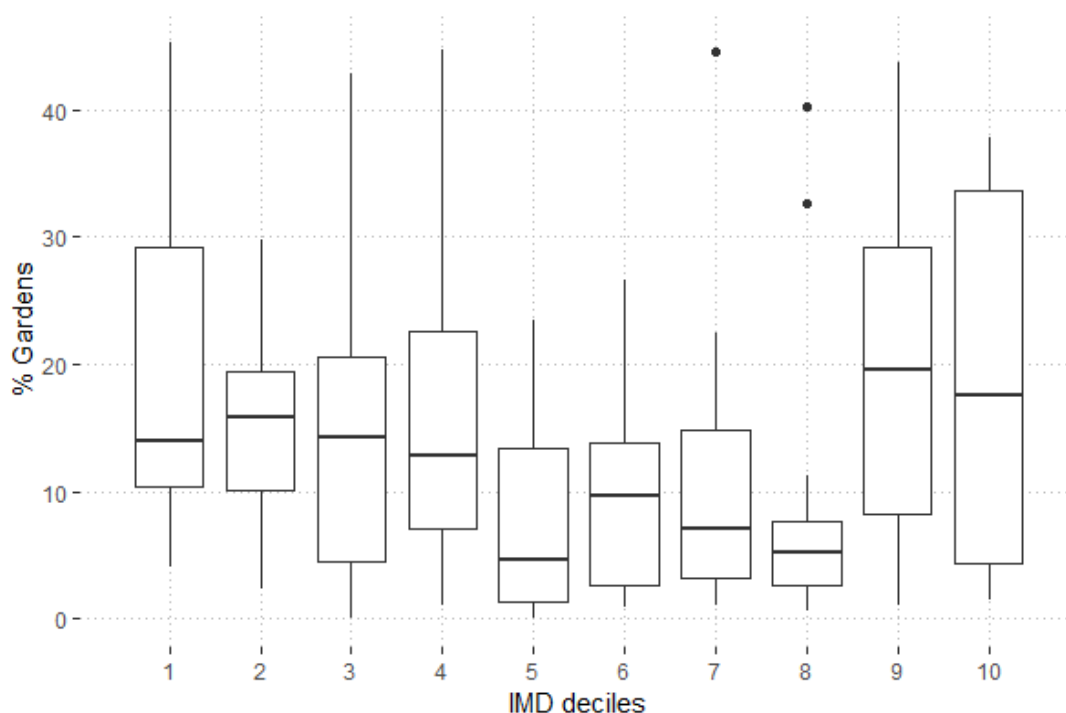


Figure 6.7 South Pennine LSOA garden area per population weighted IMD deciles. Values are percentage of total area covered by gardens as determined by GLUD 2005, data.go.uk.

As expected, this illustrates a lower average garden area for the four most deprived deciles, which then increases as deprivation decreases. Similarly, the mean and median proportion of detached dwellings steadily increase with decreasing deprivation; lowest for the most deprived decile (mean - 5%, median - 4%) and highest (mean - 47%, median - 46%) for the least deprived decile. There is a notable difference in the mean and median proportion of apartments for the most deprived decile (24% and 21% respectively) compared to all other deciles. These results confirm that least deprived areas have greater access to private garden space, which can fulfil several of the recreation based roles that public greenspace performs. Thus, the most deprived areas suffer from the lowest public and private recreation ES.

6.3.3. The spatial and social distribution of recreation in Northampton

There is a clear gradient in recreation across the Northampton case study region with lowest values in the northwest in the district of Daventry, and highest values in the southeast district of South Northamptonshire (Figure 6.8). There are a range of lower and higher values across Northampton town. Thus there is no clear distinction in recreation ES across rural and urban LSOAs. This is primarily due to the higher density of PROWs and of larger recreation areas (>100ha) to the south of the case study

region within 10km Euclidean distance of the more southern LSOAs. There are only two areas with an areas greater than 500ha and these are located to the south and south east of the study area.

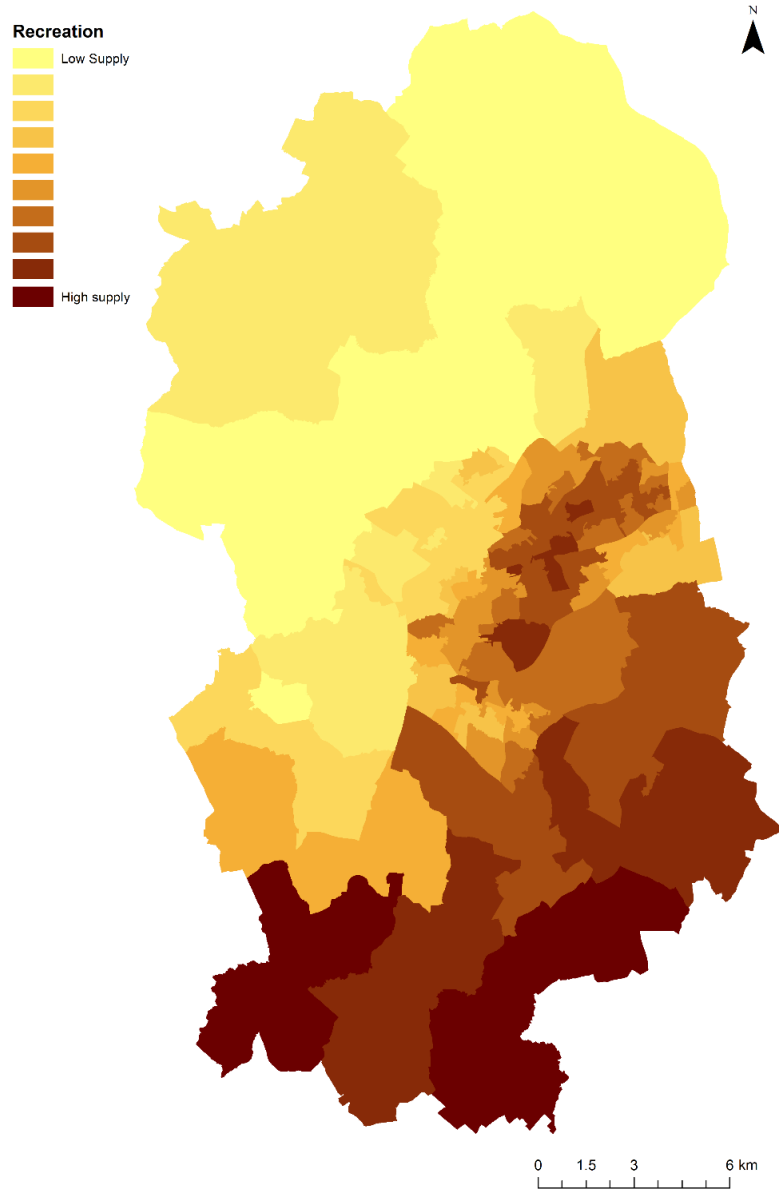


Figure 6.8 The spatial distribution of recreation ES aggregated to LSOAs Northampton case study region.

There is no significant correlation between recreation and IMD ranks (Table 6.2) whilst the concentration index lies close to 0 at -0.02 as further illustrated by the concentration curve which lies close to the line of equality and very slightly rises above this for some mid deciles (Figure 6.11). ANOVA tests confirm there is no significant difference in mean recreation ES indexes across deprivation deciles. Visualisation of

the distribution of recreation ES values using a boxplot indicates no consistent pattern (Figure 6.9); median values are lowest for the 'upper middle' deciles (5th, 7th, 8th and 9th) and the least deprived decile shares similarly high recreation values as the more deprived (1st-4th) deciles.

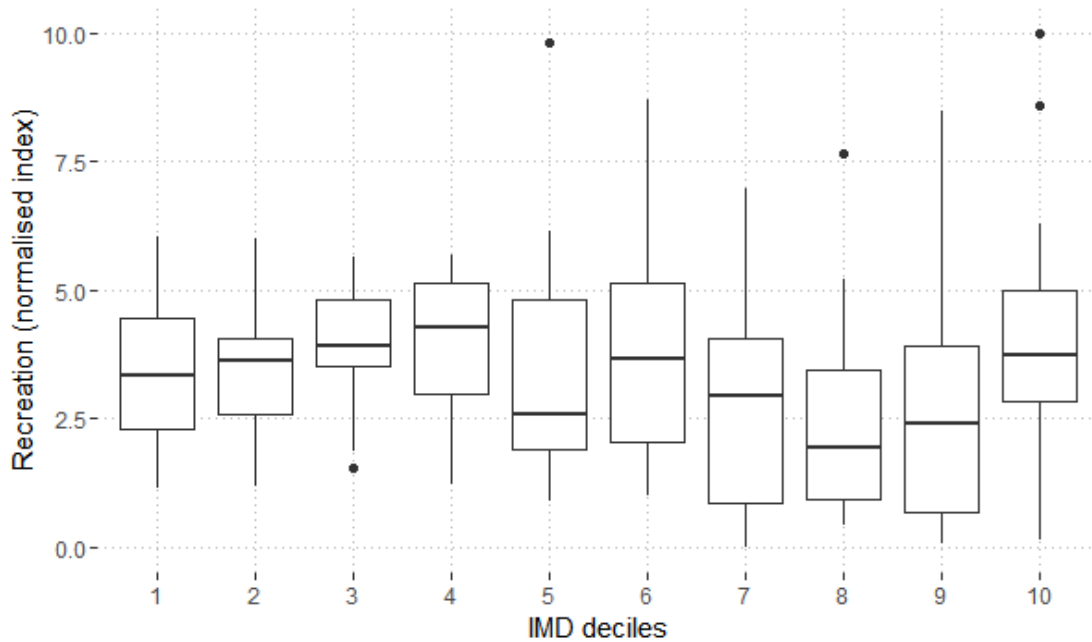


Figure 6.9 Recreation ES indexes per population-weighted IMD deciles for Northamptonshire study area. Deprivation decreases from decile 1 to 10.

Of those LSOAs with significant local spatial association between IMD ranks and recreation ES (Figure 6.10), only a few (14 out of 167 LSOAs) have significant association between high deprivation and low ES, and are concentrated in western areas of Northampton town. Conversely, the majority of significant local associations are areas of low deprivation with low ES, these are located in the northern rural areas of the study area – predominantly in the district of Daventry.

Once greenspaces and PROW are considered separately (Table 6.2), it is evident that their contrasting social distributions effectively 'cancel' each other out. There is an increase in greenspace availability with increasing deprivation yet a decrease in PROWs. Although this correlation is significant, it is weak. Of note, is that the interquartile ranges for the values in each decile tend to be larger for this service than for the other services in Northampton.

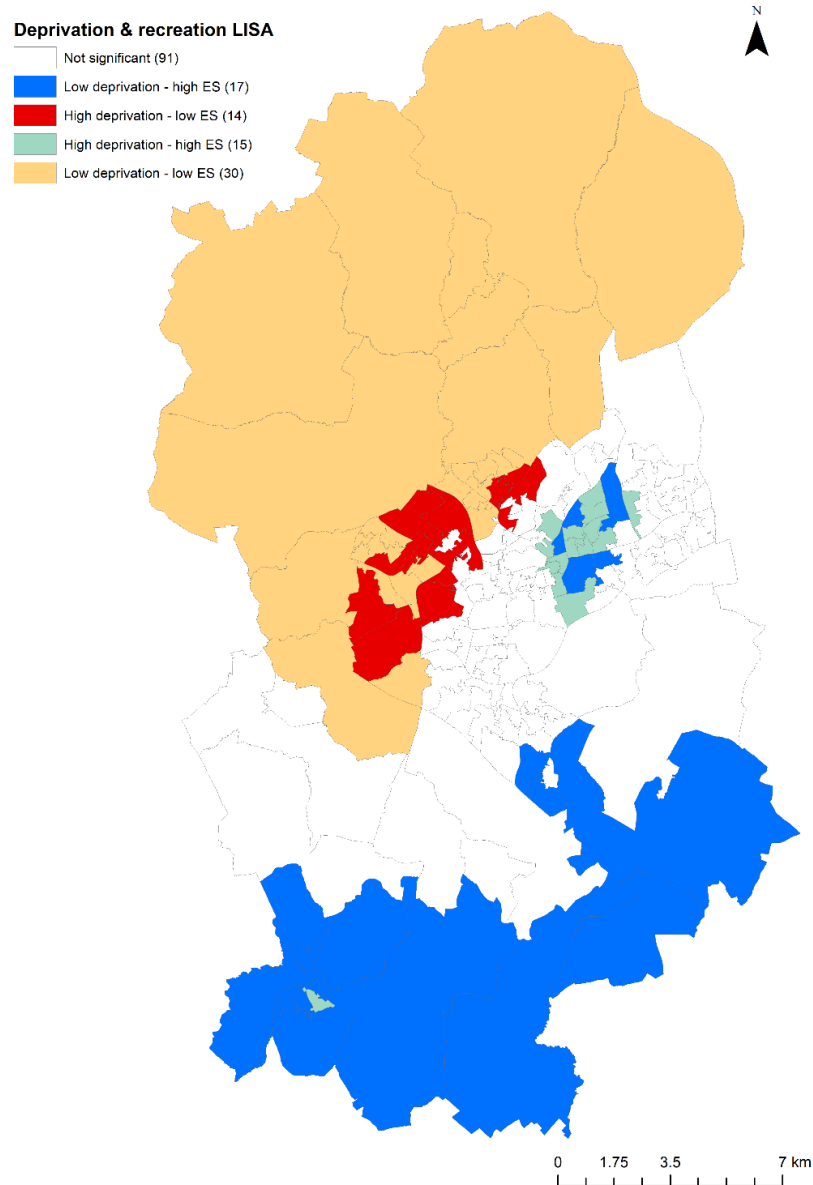


Figure 6.10 Northampton LSOAs with significant local spatial associations between IMD ranks and recreation. Identifies clusters of areas where there are significant associations between ES indexes and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class.

6.3.4. Sensitivity tests

Three variations in how recreation supply is calculated, and one in the way recreation demand is calculated, were applied and from these revised ES indexes produced. ES association with deprivation was then tested by re-computing correlation coefficients (Table 6.2) and via visualisation using boxplots (where this revealed further information this is discussed below, but boxplots are not presented).

For recreation supply, as explained in section 6.2.2, this involved calculating the amount of greenspace available within a distance of 1 mile and 2 miles of the LSOAs, and within the LSOAs only. There is no indication that more deprived areas have lower recreation opportunities than wealthier areas in Northampton and Leeds following ANGSt guidelines. This remains the case for Leeds for all variations in distances, and when only greenspace within the LSOAs is accounted for Northampton. Yet when greenspace within 1 and 2 miles are accounted for in Northampton, there is a moderately strong negative correlation with IMD ranks. The implication is that for Northampton more deprived areas actually have greater access to recreation at walkable distances but not necessary within their immediate area, nor access to the largest greenspaces at a further distance. Some of this sensitivity is likely attributable to the concentration of more deprived areas within Northampton town, which are close to urban parks.

There is a notable contrast with the results of the South Pennines case study region, where a significant positive correlation between recreation ES index and IMD ranks is consistently found, irrespective of how ES supply is quantified. Therefore it is evident that there is a persistent inequality whereby the most deprived areas suffer from the lowest accessibility of recreation.

Once demand is included, whereby areas with denser populations have a greater demand for recreation ES, the strength of association in all case areas increases. The direction of association in each case aligns with that established for the greenspaces not for the PROWs. For Leeds and the South Pennines, there is a decrease in ES as deprivation increases, meaning that for these areas demand is relatively high and supply is relatively low. In contrast, a moderate to strong negative correlation is revealed in Northamptonshire indicating that the more deprived areas have a relatively lower demand and higher supply.

To summarise, the social distribution of recreation is dependent on how recreation supply and demand is quantified. The effects observed are diverse across the study areas.

Table 6.2 Spearman rank correlation coefficients showing linear associations between recreation and IMD ranks.

a) Leeds

	IMD ranks	ES index	Greenspace index (ANGSt)	PROW density
IMD ranks	-	NS	0.155**	-0.322**
ES index	NS	-	0.618**	0.150**
Greenspace index (ANGSt)	0.155**	0.618**	-	-0.625**
PROW density	-0.322**	0.150**	-0.625**	-
Sensitivity	'LSOA only' ES index	NS		
	'1 mile radius' ES index	NS		
	'2 mile radius' ES index	NS		
	Population density	-0.388**		
	Population weighted ES index	0.263**		

b) South Pennines

	IMD ranks	ES index	Greenspace index (ANGSt)	PROW density
IMD ranks	-	0.207*	0.282**	NS
ES index	0.207*	-	0.978**	0.700**
Greenspace index (ANGSt)	0.282**	0.978**	-	0.552**
PROW density	NS	0.700**	0.552**	-
Sensitivity	'LSOA only' ES index	0.345**		
	'1 mile radius' ES index	0.367**		
	'2 mile radius' ES index	0.291**		
	Population density	-0.434**		
	Population weighted ES index	0.301**		

c) Northampton

	IMD ranks	ES index	Greenspace index (ANGSt)	PROW density
IMD ranks	-	NS	-0.180*	0.186*
ES index	NS	-	0.970**	0.337**
Greenspace index (ANGSt)	-0.180*	0.970**	-	NS
PROW density	0.186*	0.337**	NS	-
Sensitivity	'LSOA only' ES index	NS		
	'1 mile radius' ES index	-0.43**		
	'2 mile radius' ES index	-0.36**		
	Population density	-0.679**		
	Population weighted ES index	-0.416**		

IMD ranks increase as area deprivation decreases; a positive correlation indicates ES increases with decreasing deprivation.

Correlations between IMD and ES index and greenspaces and public rights of way PROWs separately are provided for initial calculations. For sensitivity tests, correlation with only IMD ranks are computed for variations in ES supply (LSOA only, 1-mile and 2-mile radius) and for population weighted demand.

NS- no significant correlation **Correlation significant at 99% confidence level

*Correlation significant at 95% confidence level.

6.4. The social distribution of recreation ES in the case study regions

For the Leeds and Northamptonshire case studies, there is no evidence that more deprived areas have lower recreation access than wealthier areas. Both are similarly large urban areas surrounded by countryside and small towns. For both there is a predominantly radial pattern of deprivation whereby most deprived areas are located in the centre of the study area, contrasting with a linear pattern of recreation ES.

Similarly, in both Leeds and Northamptonshire, there are significant correlations between deprivation and PROWs and greenspaces when assessed individually. However, these occur in different directions, thus effectively negating each other when combined to a single index. Interestingly, the direction of correlations are opposing for the different case study regions. More deprived areas in Northampton benefit from greater availability of greenspaces yet fewer PROWs. In contrast for Leeds, more deprived areas experience lower availability of greenspaces but have accessibility to PROWs. Further differences are observed in the sensitivity testing, where in Leeds, social distributions are not dependent on the distances applied, whilst for Northampton there is indication that for some definitions of recreation supply there is actually lower recreation access for wealthier areas.

Greater complexity in the associations between recreation and deprivation are found in the South Pennines. Whilst overall access to greenspace (not PROWs) is unequal and there is a decrease in access as deprivation increases, the most as well as the least deprived areas experience lower access. Further analysis did however reveal that for the least deprived this is somewhat compensated by having the greatest area of private gardens. Inequality is evident in the distribution of recreation in the South Pennines regardless of the distances used to estimate ES supply or if population density is used to indicate ES demand. Moreover for each of these sensitivity tests the strength of association becomes greater between decreasing recreation ES and increasing deprivation, in comparison to the initial calculation of recreation ES using ANGSt.

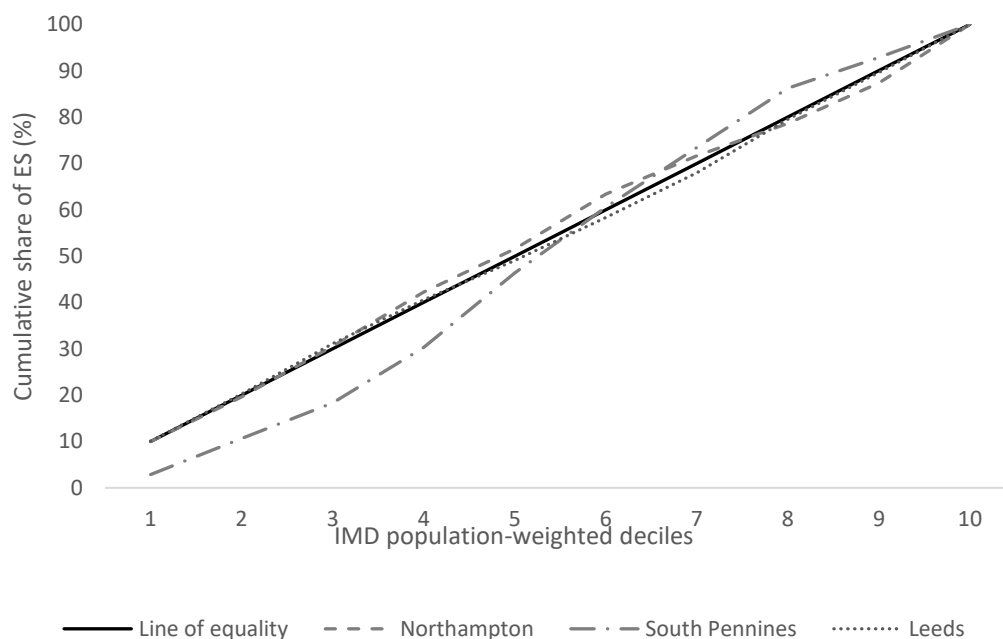


Figure 6.11 Recreation net ES index concentration curves.

IMD decile 1 represents most deprived population-weighted decile; IMD decile 10 represents least deprived population-weighted decile.

Line of equality shows theoretical equal distribution across all deciles. Concentration curves show actual distribution of recreation ES according to deprivation for each case study. Less than 'equal' share is indicated by the curve falling below the line of equality, a greater share is indicated by curve rising above the line of equality.

Overall, the social distribution of recreation is place specific (Figure 6.11). Inequalities are most prevalent in the South Pennines, largely due to the comparatively low availability of recreation spaces in the more deprived urban areas compared to the rural areas which are close to large areas of open access moorlands. However, when only rural areas are considered it is the wealthiest areas which have the lowest recreation accessibility. In all cases the social distribution of PROWs is converse to the social distribution of greenspaces, and some significant correlation is found. In addition, sensitivity testing is useful in some cases for developing a deeper understanding of the patterns of recreation for more and less deprived areas. Furthermore, for each case study region, areas with significant local association between high deprivation and low recreation have been identified; these are key areas when considering social outcomes of land management.

Chapter 7 The social distribution of surface water runoff reduction in regional case studies

This is the third and final chapter presenting analysis which corresponds to the second and third objectives of the thesis; assessing the social distribution of a specific ES within the case study regions and evaluating the sensitivity of this distribution to the way in which the ES is modelled. The ES considered in this chapter is surface water runoff reduction (SWRR), which describes the degree to which the natural environment reduces runoff that contributes to both pluvial and fluvial flood risk. Runoff is influenced by land cover characteristics that affect surface water infiltration, storage, interception and evapotranspiration (Weng, 2001; Whitford et al., 2001). Greater surface runoff, such as that associated with impermeable urban surfaces, may be realised as increases in discharge volume and peak flow (Whitford et al., 2001) which can exceed the capacity of manmade drainage systems (Każmierczak & Cavan, 2011). Therefore reduction of this runoff contributes to a reduction in flood risk (Whitford et al., 2001). Greater understanding of how equitably SWRR ES is distributed is important due to the social, health and economic implications of the associated flood risk, as highlighted in Chapter 3 section 3.2.2.1.

This chapter characterises areas of higher (and lower) runoff reduction, and how this surface water runoff attenuation is associated with more or less deprived areas. To achieve this, it is necessary to first identify spatial patterns in land cover and soil types associated with surface water runoff. Section 7.1 describes the data used for doing so. Subsequent sections discuss specific methods applied to mapping SWRR ES, building on the methodology outlined in Chapter 3 section 3.3.1.2. Section 7.2.1 commences with an explanation of how the ES flows and thus how the service provisioning areas (SPAs) are defined - specifically this addresses the delimitation of LSOA river catchments. This section continues by outlining how ES supply for the catchment areas is calculated through application of a soil curve number approach. Methodological consideration given in section 7.2.2 to the third thesis objective - sensitivity testing – where variations applied consider uncertainty and alternative conceptualisations in modelling of surface water reduction ES supply. Section 7.2.3 explains the approach for quantifying demand for SWRR in the LSOAs using flood hazard data and gives a brief description of its combination with ES supply to generate a net ES index.

Results addressing objective 2 (social distribution of ESs) are given for each case region individually (sections 7.3.1 - 7.3.3). For each, the spatial and social distributions of surface water runoff are described and visualised through a series of maps, boxplots and statistics. Section 7.3.4 examines, across all three case studies collectively, the

sensitivity of the revealed social distributions to uncertainties in quantifying ES supply and the alternate conceptualisations of supply and demand.

7.1 Data

The Land Cover Map of Great Britain 2007 (LCM2007: NERC CEH, 2011), released in 2011, forms the main basis of land cover information used to quantify SWRR supply. This is a 25m resolution raster dataset which provides continuous coverage of 20 land cover classes across Great Britain. Whilst an updated map for 2015 is now available, this was not the case when the analysis commenced. One limitation of the LCM2007 dataset is that the resolution does not reveal the detail of highly variable land cover in built-up areas. OS Mastermap Topographic vectors (2014) are therefore used to enhance the detail of the classification. The LCM2007 land cover class covering the highest percentage of total area of each OS Mastermap polygon is assigned to the respective polygon. A final land cover class is assigned according to which dataset provided the most information. For example, all buildings and roads delimited by OS Mastermap were classified as impervious land cover, and woodland defined by OS Mastermap was assigned as woodland for the final classification. For all areas defined as 'natural' or 'general surface' by OS Mastermap, the LCM classification was used. Land cover data was supplemented by information on soil types, obtained from the Hydrology of Soil Types (HOST) (Boorman et al., 1995) reclassification of National Soil Map of England and Wales; this was provided by the Centre for Ecology and Hydrology (CEH; www.ceh.ac.uk). The limitation of this dataset is its lower resolution (1km), however, it has national coverage and provides accessible information on how the soil may influence hydrological processes. This is based on the physical soils type and conceptual models of hydrological processes (Boorman et al., 1995). HOST data was matched to land cover data according to the HOST class with the highest coverage of each OS Mastermap land cover polygon.

As discussed in the next section, mapping ES supply also required delimitation of catchment areas of the case study LSOAs. As a minimum, this requires data depicting watercourses and elevation, both of which were obtained from the Centre for Ecology and Hydrology (www.ceh.ac.uk). This comprised a hydrologically corrected digital terrain model (IHDTM) (Morris & Flavin, 1990) at a 50m resolution and a 1:50,000 watercourse vector dataset, both derived from 1:50,000 OS Mapping.

Mapping of ES demand drew on the Environment Agency's Risk Band layer from their Risk of Flooding from Multiple Sources (RoFMS) datasets. This national map integrates individually produced sets of data; Risk of Flooding from Rivers and Sea and Risk of

Flooding from Surface Water (Environment Agency, 2017a). The Risk of Flooding from Rivers and Sea is a 50m resolution gridded dataset which is derived from local flood models and accounts for the influence of flood defences and the dependency between coastal and fluvial sources. This allocates each cell to a flood risk category denoting the likelihood of flooding in any given year (Environment Agency, 2017a). The Risk of Flooding from Surface Water datasets include three separate maps showing the surface water depth from rainfall with a 1 in 30, 1 in 100 and 1 in 1000 chance of occurring in any given year. From these a 2m gridded dataset is extrapolated to give a probability of flooding from surface water. To generate the Risk of Flooding from Multiple Sources, the probability maps for each source of flooding are added together, this therefore does not account for any dependencies between surface water flooding and fluvial/coastal flooding and may overestimate the combined chance of flooding (Environment Agency, 2017). Risk of flooding from groundwater, reservoirs or drainage/sewers is not accounted for. The resulting dataset is 2m gridded dataset with each cell assigned to four risk bands: 1- greater than 1 in 30, 2 - between 1 in 30 and 1 in 100, 3 - between 1 in 100 and 1000 and 4 - less than 1 in 1000 in any given year (Environment Agency, 2017). The dataset was accessed in 2017 from environment.data.gov.uk.

7.2 Methods

7.2.1 Mapping surface water runoff reduction supply and flows

For SWRR, ES flows from service providing areas (SPAs) to service benefitting areas (SBAs) are determined through physical, hydrological processes (Syrbe & Walz, 2012) and occur at the catchment scale. Since the LSOAs are the spatial unit required for analysing inequalities, the LSOA boundaries within the case study define the SBAs, consistent with analysis of the other ESs. The river catchment for each LSOA are then defined, these delineate the SPA extents. This identifies all the natural capital upstream from the LSOA which may reduce runoff and thus be of benefit to those downstream by reducing the risk of flooding (Syrbe & Walz, 2012). Figure 3.7 in Chapter 3 provides an illustration of these spatial linkages.

ArcGIS 10.3 Arc Hydro Tools (downloads.esri.com/archydro) were used to define catchment boundaries following the ESRI (2011) workflow. Model inputs were the IHDTM and watercourse data. Catchment extents were verified by comparison of a sample from each case study to extents presented in the Flood Estimation Handbook Web Service viewer (fehweb.ceh.ac.uk). Individual catchments were linked to their

corresponding LSOA via a unique Hydro ID. The combined extent of catchments was used to extract the relevant data required for quantifying ES supply (as described in section 7.2.1).

A storm-runoff coefficient (USDA-NRCS, 1986), also known as the soil curve number method (SCS CN method), is used to quantify ES supply. This estimates the amount of runoff from defined land cover/soil type combinations under a selected precipitation scenario and pre-condition (i.e. dry or wet soils). Originally devised by the United States Natural Resources Conservation Service (NRCS), the method is empirically based; derived from observations of rainfall, runoff, land use, soil permeability and antecedent wetness. Observations were made for US land cover and soils but it is now the most widely used runoff model (Zeng et al., 2017) and has been applied globally (e.g. Laterra et al., 2012; Fu et al., 2013; Sjöman & Gill, 2014; Zeng et al., 2017). In the UK, this method has been used to estimate the reduction in runoff by natural land cover in various urban areas including Edinburgh, Glasgow, Leicester, Oxford, Sheffield (Tratalos et al., 2007; Holt et al., 2015), Manchester (Gill, 2005), Leeds (Perry and Nawaz, 2008) and Merseyside (Whitford et al., 2001). Whilst typically the application of curve numbers has been to urban areas in the UK, it was originally developed in the US for agriculture watersheds (USDA-NRCS, 1986). A 2011 pilot study for the Scottish Environment Protection Agency (SEPA) set the precedent for its use in UK rural catchments, using the method as part of a more detailed assessment of potential natural flood management in Scotland (Halcrow, 2011). This was carried out for a dominantly upland catchment which features some agricultural land and smaller settlements downstream. A similar approach was then taken by Thomas & Nisbet (2016) for the Pickering Beck catchment in rural North Yorkshire.

There is however some caution required regarding the application of the curve number method in the UK, especially with respect to transfer of US values to UK land covers, particularly in rural areas, where it has been found to underestimate runoff (Halcrow, 2011). However, there are factors which lend itself to being an appropriate method for this study, not least that it allows for a relatively straightforward quantification of the role of natural land cover in runoff generation. Furthermore, our goal is to develop a spatial comparative analysis, rather than estimate runoff volumes for use in subsequent inundation analysis, which would require a higher degree of accuracy and more complex hydrological modelling (e.g. Nedkov & Burkhard, 2012). Such an intensive approach was not adopted given the context of the wider analysis concerned with mapping of multiple ESs in multiple regions. Thus in this analysis, where approximate runoff values are sufficient to derive the important relative spatial differences, estimated runoff volume is not given, and ES supply values are provided on a relative scale. As Thomas and Nisbet (2016) conclude, whilst the empirically based CN method does not

have the sophistication of physically based hydrological models, it does nevertheless offer a clear way of identifying locations of high and low runoff generation. The scope for error in estimates of the relative difference in ES identified using the CN method is tested for in the sensitivity analysis (section 7.2.2).

To implement the CN method, discharge (runoff depth) for a rainfall event is calculated as (from Holt et al., 2015):

$$Q = \begin{cases} \frac{(P-0.2S)^2}{P+0.8S}, \\ 0 \end{cases}$$

$$Q = \frac{(P-0.2S)^2}{P+0.8S} \text{ if } P > 0.2S, \text{ else } Q = 0 \dots \dots \dots \text{Equation 7.1}$$

$$S = \frac{2540}{CN} - 25.4 \dots \dots \dots \text{Equation 7.2}$$

Where:

Q = runoff depth

P = precipitation

CN = curve number

S = maximum potential retention of water.

CNs range from 0 to 100, where 100 indicates a completely impermeable surface where 100% rainfall becomes runoff. Natural grassland tends to have the lowest curve number due to it allowing infiltration into soils (Armson et al., 2013), although this is dependent on the permeability of the underlying soils. Some natural land covers may have high CN such as bare soil, which may be highly compacted and allow little infiltration, in addition to not intercepting rainfall or enabling evapotranspiration.

To apply equation 7.1, CNs are first assigned to each OS Mastermap land cover polygon based on the final land cover class and soil type (see Table 7.2). Typically, CNs are transferred from those calculated by the USDA-NRCS (1986) for US land covers to UK land covers based on their closest match. However the land cover classes used and CNs assigned to these in the UK have differed across studies. Classification schemes and associated CNs used by Whitford et al. (2001), Tratalos et al. (2007), Halcrow (2011) and Holt et al. (2015) in addition to those originally determined by USDA-NRCS (1986) were used to inform those selected in this study. The classification was determined as one which can be depicted from LCM2007 combined with OS Mastermap data. Whilst it represents a simplification of the UK Broad Habitat classification, it presents a similar level of detail to those used in other

studies (e.g. Holt et al., 2015). Furthermore, similar classes which have the same CNs are combined into a single class (e.g. broadleaved and coniferous woodland are assigned to a single 'woodland' class). Assigned CNs are based on mid values from across the studies referred to above. Suburban and urban LCM2007 classes are used where more detailed classification is not possible, since within this class actual land cover varies the CNs assigned based on a weighted average. This weighting is derived by calculating the percentage of land cover types within the 'urban' and 'suburban' classes for a sample area using the more detailed 5m resolution classification derived for computation of the air pollutant removal ES (see Chapter 5).

For each land cover CNs further depend on four soil hydrologic groups. Following Perry and Nawaz (2008) and Halcrow (2011), HOST soil types are assigned to the four groups used by USDA-NRCS (1986). Table 7.1 lists the soil groups, and Table 7.2 and Table 7.3 list the land cover classes and associated CNs.

Table 7.1 Conversion from HOST soil types to NRCS Hydrological Group based on Halcrow (2011)

Standard Percentage Runoff (HOST) (%)	NRCS Hydrological Group	HOST soil class
<10	A	1,2,4,11,13
10-20	B	3,5
20-40	C	6,9,10,14,16,17,18,24
>40	D	7,8,12,15,19,20,21,22,23,25,26,27,28,29

Table 7.2 Curve numbers assigned to land cover classes and underlying soil group for dry antecedent conditions.

Land cover class	Soil Hydrological Group			
	A	B	C	D
Impervious	98	98	98	98
Water	100	100	100	100
Bare soil	77	86	91	94
Arable	64	75	82	85
Improved grassland	49	69	79	84
Semi-natural grassland	35	60	73	79
Scrub	45	66	77	83
Marsh/bog	95	95	95	95
Woodland	36	60	73	79
Suburban	58	74	82	86
Urban	60	75	83	87

Curve numbers range from 0-100. Low numbers indicate high permeability, 100 indicates impermeable land.

Table 7.3 Curve numbers assigned to land cover and soil groups for wet antecedent conditions

Land cover class	Soil Hydrological Group			
	A	B	C	D
Impervious	99	99	99	99
Water	100	100	100	100
Bare soil	88	93	95	97
Arable	80	87	91	92
Improved grassland	68	83	89	92
Semi-natural grassland	55	77	86	89
Scrub	65	81	88	91
Marsh/bog	97	97	97	97
Woodland	56	77	86	89
Suburban	76	86	91	93
Urban	77	87	91	93

Curve numbers range from 0-100. Low numbers indicate high permeability, 100 indicates impermeable land.

To isolate the effect of natural land cover in reducing runoff, a hypothetical scenario where all land cover is impermeable is used (following Holt et al., 2015; Maragno et al., 2018). Runoff (Q) is calculated for actual land cover and for the hypothetical scenario based on a rainfall depth of 1.2 cm and normal antecedent conditions (using CN values in Table 7.2) as considered by Holt et al., (2015) to be a heavy event for their study location of Sheffield. This is a reasonable scenario to transfer to this study given the geographical location of the case studies, which are typically wetter (South Pennines) and drier (Leeds and Northampton) than Sheffield. They are however within reasonable proximity to the south and north of Sheffield. A heavy rainfall event is used since it is in these instances that the ES has a greater importance. Subtraction of the two gives the runoff reduction made by natural land cover.

The antecedent ground condition, i.e. the saturation of soil prior to a rainfall event, influences how effective natural land cover is at reducing runoff. This may alter the spatial distribution of SWRR ES and therefore needs to be taken into account. CN values based on wet conditions (Table 7.3) are used to calculate runoff under wet conditions (Q_{wet}). As with the previous scenario, rainfall depths used by Holt et al. (2015) are applied. Thus, for the wet scenario an extreme event where rainfall occurs to a depth of 6 cm is used. Whilst such high rainfall is less likely in Northampton, this was exceeded for many parts of the South Pennines and Leeds regions during a 2015 storm which caused widespread flooding (Met Office, 2016), and is thus considered realistic. Again, the difference between the actual and hypothetical scenarios is calculated. Average runoff reduction from both normal and wet antecedent conditions are calculated (following Holt et al., 2015).

The final stage in the production of an ES supply index for SWRR is to aggregate the mean runoff reduction across each catchment. This is achieved by calculating the area weighted mean using the polygon intersection tool in Geospatial Modelling Environment software (www.spatialecology.com/gme). The results are rescaled between 0 and 10, where 0 represents the lowest supply and 10 represents the highest supply (section 3.3.1.2). This enables relative ES supply to be compared to relative ES demand.

7.2.2 Surface water runoff reduction sensitivity testing

Sensitivity tests address sources of uncertainty made in the quantification of surface water runoff ES supply. There are various potential sources of uncertainty in the computation of ES supply including the spatial and thematic resolution of land cover and soil classification data, and the CNs applied. The input datasets are widely available with detailed land cover given the areal coverage required and so have been applied in other studies adopting a similar approach (e.g. Tratalos et al., 2007; Holt et al., 2015). However, as noted above, the CNs applied have varied quite considerably across all such studies. This could impact upon the spatial distribution of modelled ES supply hence sensitivity tests were conducted to address variability in CNs selected.

Table 7.4 Curve numbers applied for 'high influence' ES supply scenario. Comparative to the 'best estimate' scenario curve numbers for vegetated land are reduced to represent a greater effectiveness of vegetation in reducing runoff. These are determined using the lowest curve numbers applied in the studies referenced in section 7.2.1.

Land cover class	Wet antecedent conditions				Dry antecedent conditions			
	Soil groups				Soil groups			
	A	B	C	D	A	B	C	D
Impervious	98	98	98	98	99	99	99	99
Water	100	100	100	100	100	100	100	100
Bare soil	72	82	87	89	86	91	94	95
Arable	51	67	76	80	71	82	88	90
Improved grassland	39	61	74	80	60	78	87	90
Semi-natural grassland	30	58	71	78	50	76	85	89
Marsh	85	85	85	85	93	93	93	93
Scrub	45	66	77	83	65	82	89	92
Woodland	30	55	70	77	50	74	84	89
Suburban	53	69	77	82	72	83	89	91
Urban	57	72	81	85	75	86	91	93

Table 7.5 Curve numbers applied for 'low influence' ES supply scenario. Comparative to the 'best estimate' scenario curve numbers for vegetated land are increased to represent a lower effectiveness of vegetation in reducing runoff. These are determined using the highest curve numbers applied in the studies referenced in 7.2.1.

Land cover class	Wet antecedent conditions				Dry antecedent conditions			
	Soil groups				Soil groups			
	A	B	C	D	A	B	C	D
Impervious	98	98	98	98	99	99	99	99
Water	100	100	100	100	100	100	100	100
Bare soil	77	86	90	91	88	93	95	97
Arable	76	85	90	93	88	92	95	97
Improved grassland	68	79	86	89	83	90	93	95
Semi-natural grassland	30	58	71	78	50	76	85	89
Marsh	95	95	95	95	98	98	98	98
Scrub	48	67	77	83	68	82	89	92
Woodland	45	66	77	83	65	82	89	92
Suburban	62	74	81	84	79	87	90	93
Urban	68	79	86	89	83	90	93	95

Two variants of CNs are tested (Table 7.4 and Table 7.5); firstly where vegetation is more effective than the baseline scenario at reducing runoff ('high influence' scenario)

and; secondly where vegetation is less effective at reducing runoff ('low influence' scenario). The CNs used in the sensitivity tests are the highest and lowest values assigned to respective land covers in the same UK studies the baseline CNs were established from. They are therefore still considered to be a realistic representation. These alternative CNs are applied to calculations of ES supply and ultimately net ES index.

7.2.3 Mapping surface water runoff demand

The demand for regulating ESs may be conceived as the "amount of regulation needed to meet predetermined conditions" (Villamagna et al., 2013). In their review of how studies have operationalised ES demand, Wolff et al. (2015: p163) find that demand for regulating services tends to be quantified in terms of "exposure and vulnerability of society and economy to potential changes of ecosystem conditions". Specifically, for flood regulation (and thus for SWRR) a common approach is to combine flood hazard with vulnerability of assets and/or population density, often termed 'flood risk' (e.g. Stürck et al., 2014). Thus the service is deemed to be 'needed' more if there are more people or higher potential for economic loss (e.g. Verhagen et al., 2016). There is also a large literature regarding social vulnerabilities, where certain social groups experience greater flood risk or have lower ability to recover from flooding (see Kuhlicke et al., 2011 for a review). There is thus potential to incorporate this research within the concept of demand for SWRR, which would involve shaping ES demand according to socio-economic characteristics (e.g. Nedkov & Burkhard, 2012; Maragno et al., 2018).

For this analysis, however, neither the value of assets or socio-economic factors are incorporated into the quantification of ES demand. Accounting for greater monetary values of assets (e.g. housing) would bias demand to wealthy areas, potentially concealing demand in poorer areas (where more vulnerable households often reside). Similarly consideration of socio-economic conditions which characterise vulnerability would coincide with the subsequent comparison to deprivation data and so risks introducing a 'double counting' effect in the social distribution analysis.

Therefore, in this study, ES demand in the first instance is determined using flood hazard only, in line with Villamagna et al.'s (2013) description of demand for regulating services. Therefore demand is greatest where flood hazard is greatest irrespective of whether a location has 1 or 100 people, or property worth £70,000 or £700,000. Sensitivity testing subsequently assesses ES demand with a consideration of the number of people exposed to the flood hazard. This enables social distributions to be

tested in accordance with different normative stances on what 'fair' means, as discussed in section 8.3.

The case study LSOAs define the extents of SBAs, i.e. the location of demand for SWRR ES. Syrbe & Walz (2012) limit the SWRR SBA as the 'built area within the floodplain', however in this analysis non-built up areas accessible for recreation purposes (as defined in Chapter 6) and agricultural land are included. This is because of potential social and economic impacts from flooding of this land in addition to urban land cover. To compute an ES demand for each LSOA, the area-weighted mean flood hazard, obtained from RoFMS Risk Band dataset (see section 7.1) is calculated using intersection tools in Geospatial Modelling Environment software. Since the highest risk in the RoFMS dataset is assigned as the lowest band, the values in the RoFMS dataset were reversed prior to calculation of mean hazard. The results are rescaled between 0 and 10 to give a final ES demand index which can then be compared to ES supply for creation of the net ES index (section 3.3.1.2). Areas at risk of flooding (>0.1% chance of flooding in any given year) as defined by the RoFMS dataset are included on maps of ES demand to emphasise the spatial extent of demand within each LSOA.

Finally, the net SWRR index is computed by subtraction of demand from supply, both of which have been aggregated to LSOAs and have values rescaled between 0-10. LSOAs with the lowest values are where there is greater flood risk combined with lower amount of runoff reduced by natural land covers across the whole catchment. The highest values indicate areas where there is lower flood risk but higher interception, infiltration and evapotranspiration of runoff by natural land covers and soil. It should be noted that a value of 0 does not necessarily mean that runoff reduction is such that it negates flood risk.

7.3 Results

7.3.1 The spatial and social distribution of surface water runoff reduction in Leeds

Highest SWRR ES supply is clustered in the north-east of the Leeds region (Figure 7.1), a largely rural area dominated by arable farmland and woodland. Supply is also moderately higher along the River Aire, which crosses the region northwest to southeast. Lower supply radiates from near the city centre to the edge of the city in several linear sections. Highest ES demand is also predominantly along the River Aire; particularly in the southeast of the study site, in the west of the city and around the periphery town of Otley in the very northwest of the region (Figure 7.2). Notably, there

is also moderately higher demand in the rural north-eastern LSOAs. Overall there is moderately strong and significant correlation between ES supply and ES demand (Table 7.6).

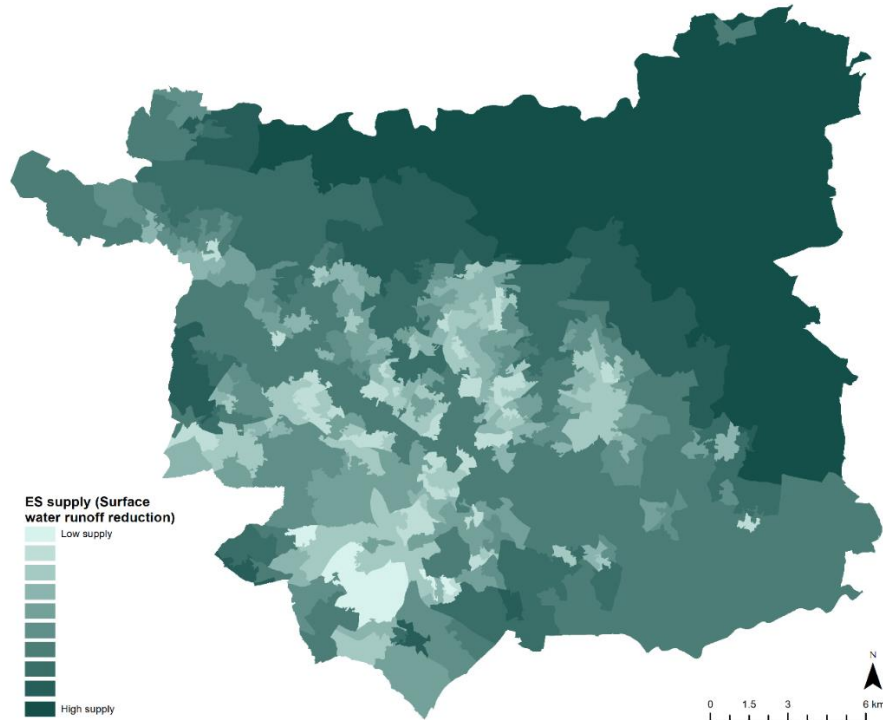


Figure 7.1 The spatial distribution of SWRR supply aggregated to LSOAs (using areal weighted mean) across Leeds.

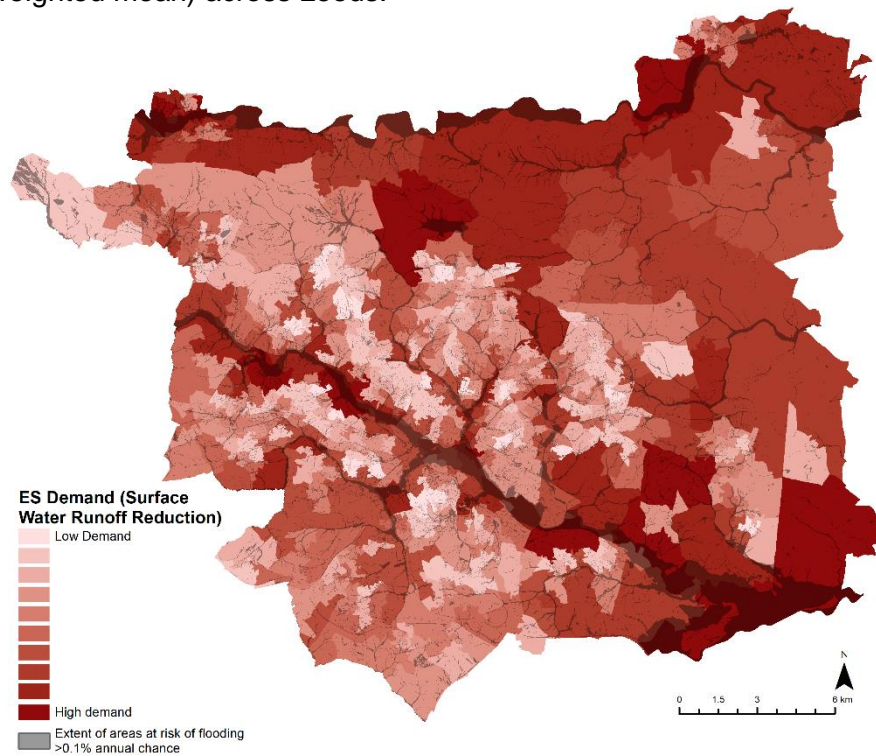


Figure 7.2 The spatial distribution of demand for SWRR aggregated to LSOAs (using areal weighted mean) across Leeds. Shaded areas indicate the extent of all areas at risk of flooding within the LSOAs.

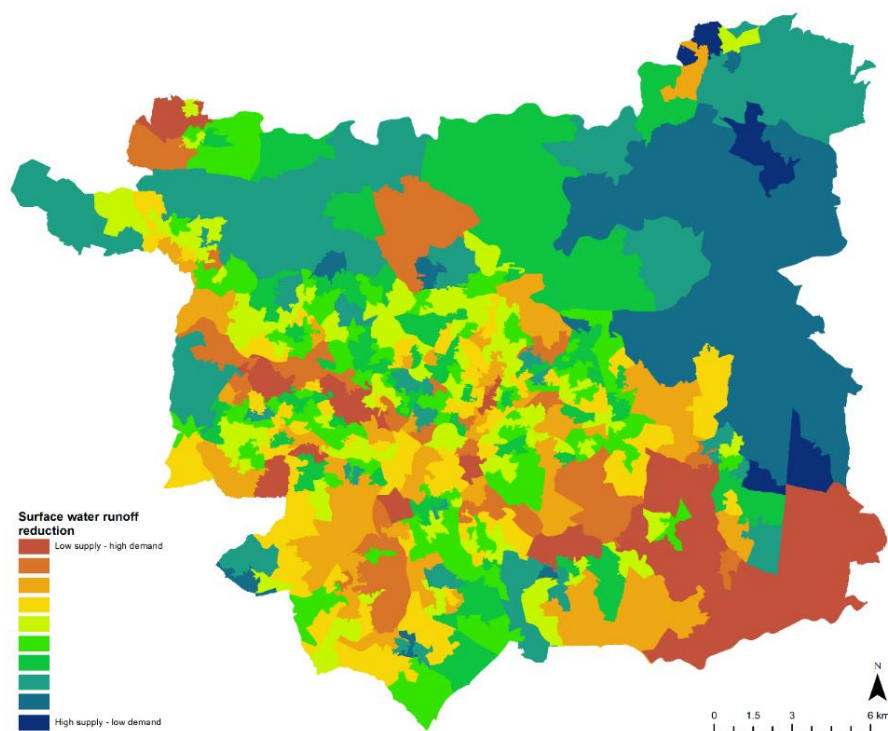


Figure 7.3 The spatial distribution of SWRR net ES index aggregated to LSOAs across Leeds.

The net ES index (supply less demand) is lowest in the northeast (but with pockets of high ES scattered across the district) and highest in the southeast, along further sections of the River Aire in Leeds city and in Otley (Figure 7.3). Spearman Rank correlation coefficients indicate that net ES index is more strongly associated with the ES demand ($\rho = -0.647$) compared to ES supply ($\rho = 0.275$) (Table 7.6). Overall for Leeds, SWRR ES is lower in more deprived LSOAs than in less deprived areas. Spearman Rank correlation coefficients between the ES and deprivation are given in Table 7.6 (below p. 198). These reveal a small but significant linear relationship between the net ES index and IMD ranks ($\rho = 0.226$), thus ES increases as deprivation decreases. This is primarily driven by the significant moderate positive correlation between ES supply and IMD ranks ($\rho = 0.386$). Correlation between ES demand and IMD ranks is positive and weak (significant at the 95% rather than 99% confidence level). The concentration index, computed using the net ES index, is close to zero (0.09), indicating a relatively equal distribution of surface water runoff ES across deprivation deciles.

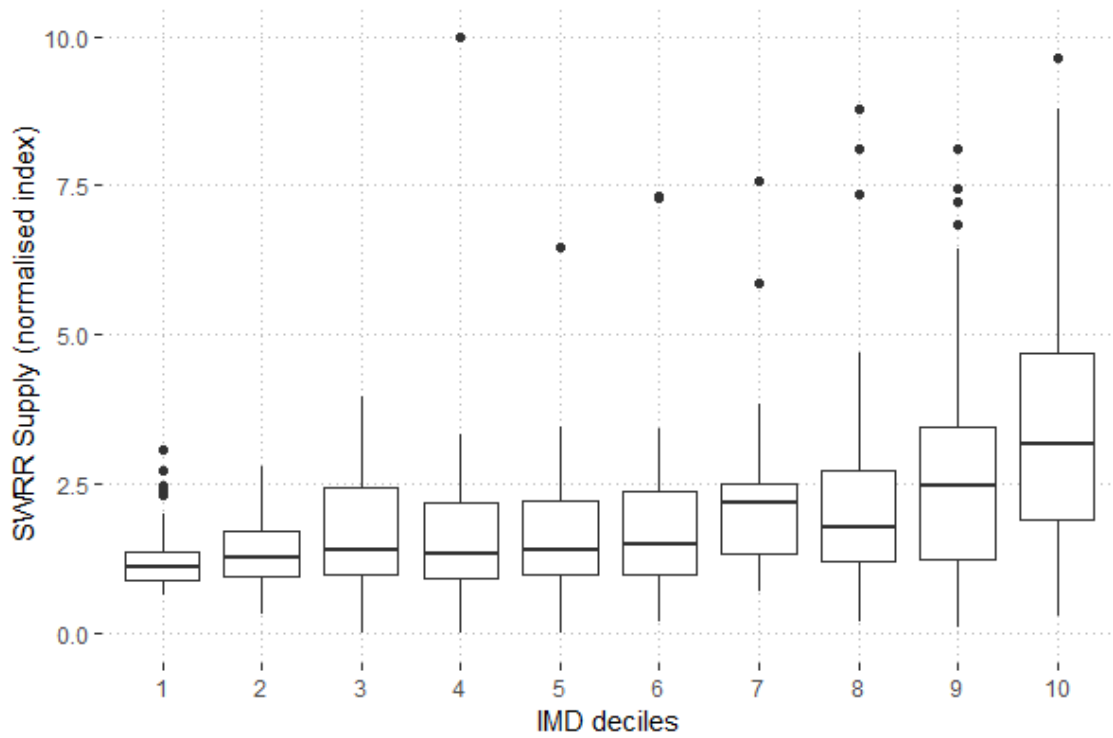


Figure 7.4 SWRR supply values per population weighted IMD deciles for Leeds. Deprivation decreases, decile 1 to 10.

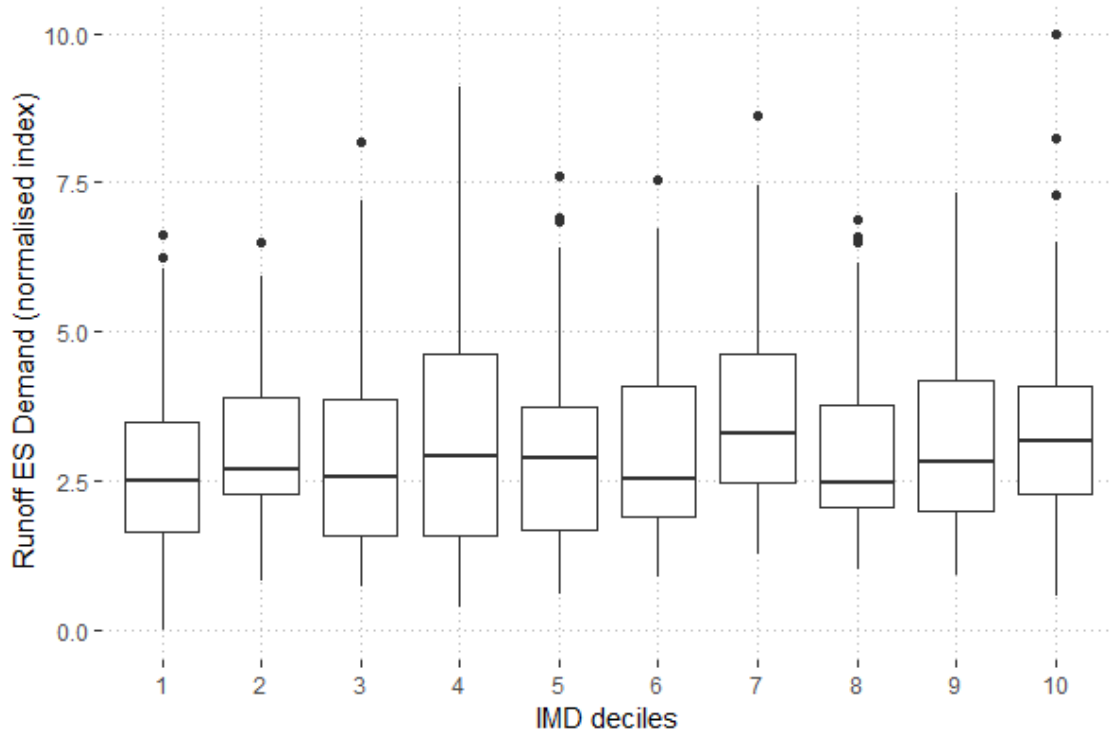


Figure 7.5 SWRR demand values per population weighted IMD deciles for Leeds. Deprivation decreases, decile 1 to 10.

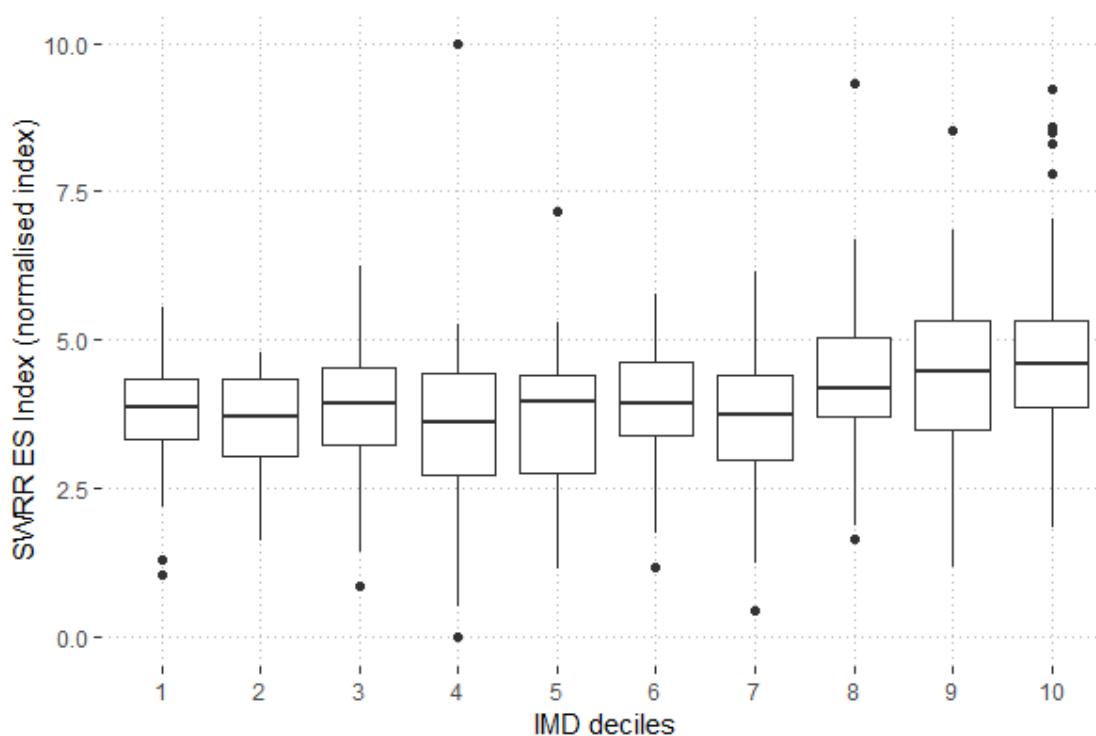


Figure 7.6 SWRR net ES index per population weighted IMD deciles for Leeds. Deprivation decreases, decile 1 to 10.

Boxplots provide a more detailed insight into how inequalities indicated by the correlation are manifest. Specifically, that net ES index tends to be slightly higher for the three least deprived deprivation deciles, but with no discernible pattern across the seven more deprived deciles (Figure 7.6). The distribution of ES supply across deprivation deciles is visualised as a boxplot in Figure 7.4. Supply is lowest for the two most deprived deciles and highest for the three least deprived deciles, reaching a maximum for the least deprived (10th) decile. There is no discernible change in ES demand across deprivation deciles (Figure 7.5).

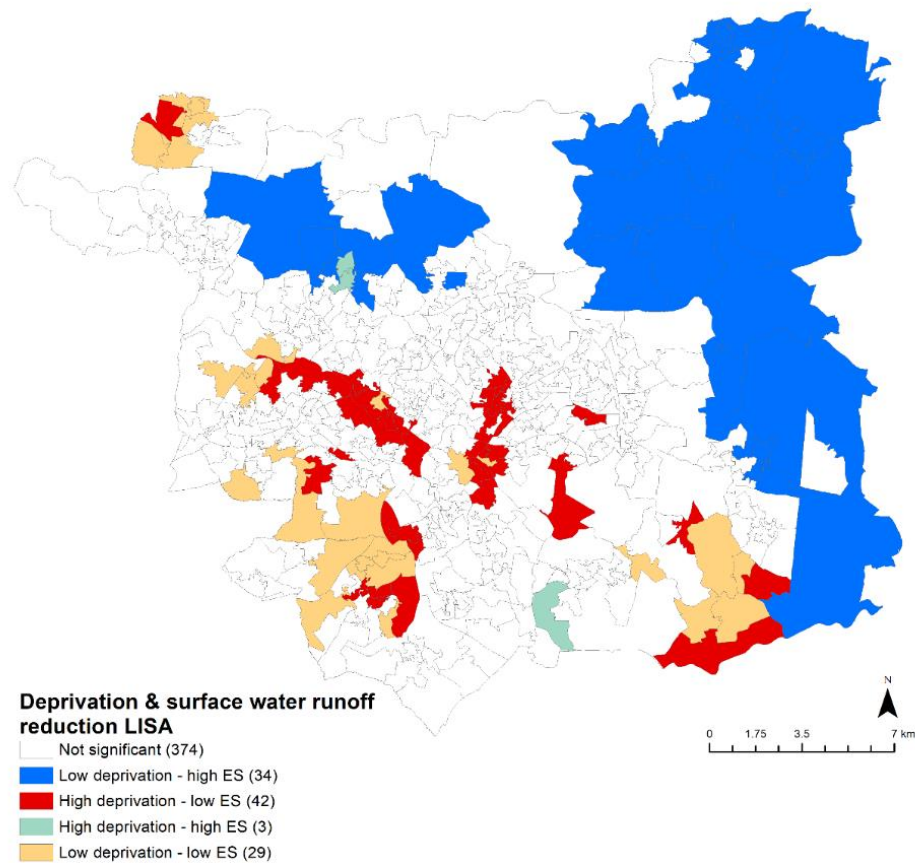


Figure 7.7 Leeds LSOAs with significant local spatial associations between IMD ranks and SWRR. Identifies clusters of areas where there are significant associations between ES indexes and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class.

Spatial correlations indicate that LSOAs where there is both low net ES index and high deprivation, are located in more built up areas. Specifically along a section of the River Aire northwest from Leeds city centre, to the east of the city centre and beyond the city boundaries around Allerton Bywater/Methley in the southeast of the region and Otley in the northwest (Figure 7.7). This constitutes the greatest number (42) of all LSOAs with significant local correlation between deprivation and net ES index. There are only three LSOAs where there is significant local correlation between high deprivation and high net ES index.

7.3.2 The spatial and social distribution of surface water runoff reduction in the South Pennines

Supply for SWRR in the South Pennines tends to be highest in the urban areas, notably around Halifax in the east of the study area (Figure 7.8). This appears counter intuitive but is likely due to the large proportion of total catchment area by HOST group

B soils (which allow for greater infiltration than the other soil groups present in the region). ES supply is lower in the rural areas, which have a greater coverage of HOST group D soils (the least permeable), whilst peatlands are also widespread. Peatlands have high CNs since the ground is considered to be permanently saturated and thus has less capacity to reduce runoff.

Highest demand for SWRR is concentrated along the River Calder, in the east of the study region running from Hebden Bridge in a southeast direction towards Sowerby Bridge (south of Halifax), and more centrally around Todmorden (Figure 7.9). LSOAs with lower demand are predominantly clustered in urban areas of Halifax north of the River Calder and in rural areas between the outskirts of the towns of Huddersfield and Rochdale in the south and stretching through the region northwards.

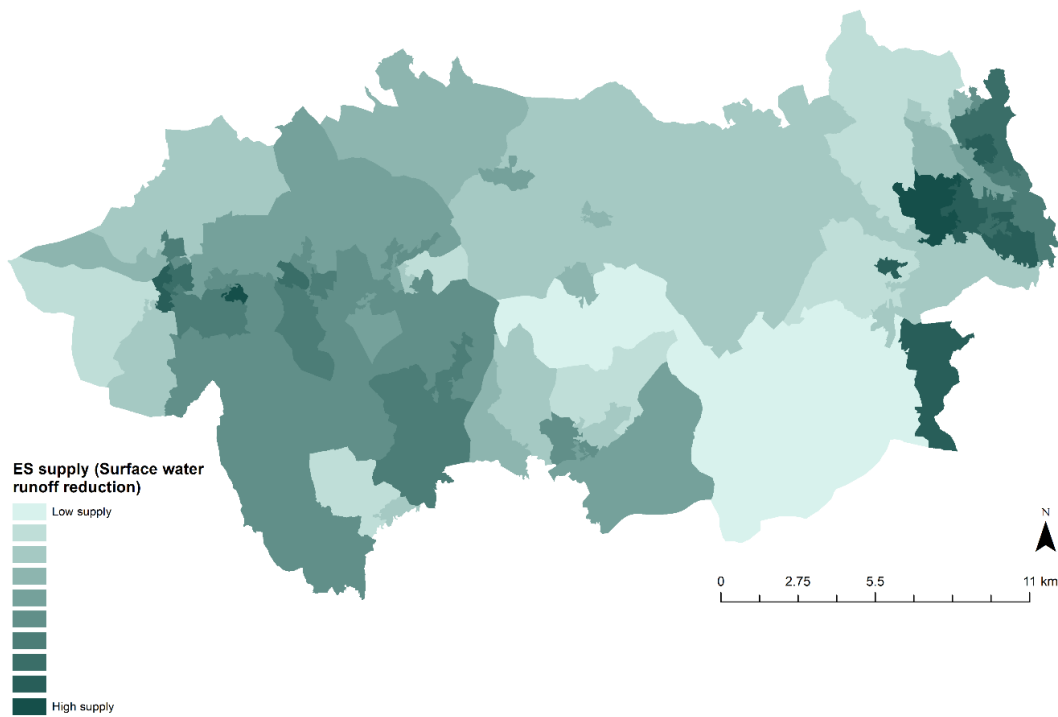


Figure 7.8 The spatial distribution of SWRR supply aggregated to LSOAs (using areal weighted mean) across the South Pennines.

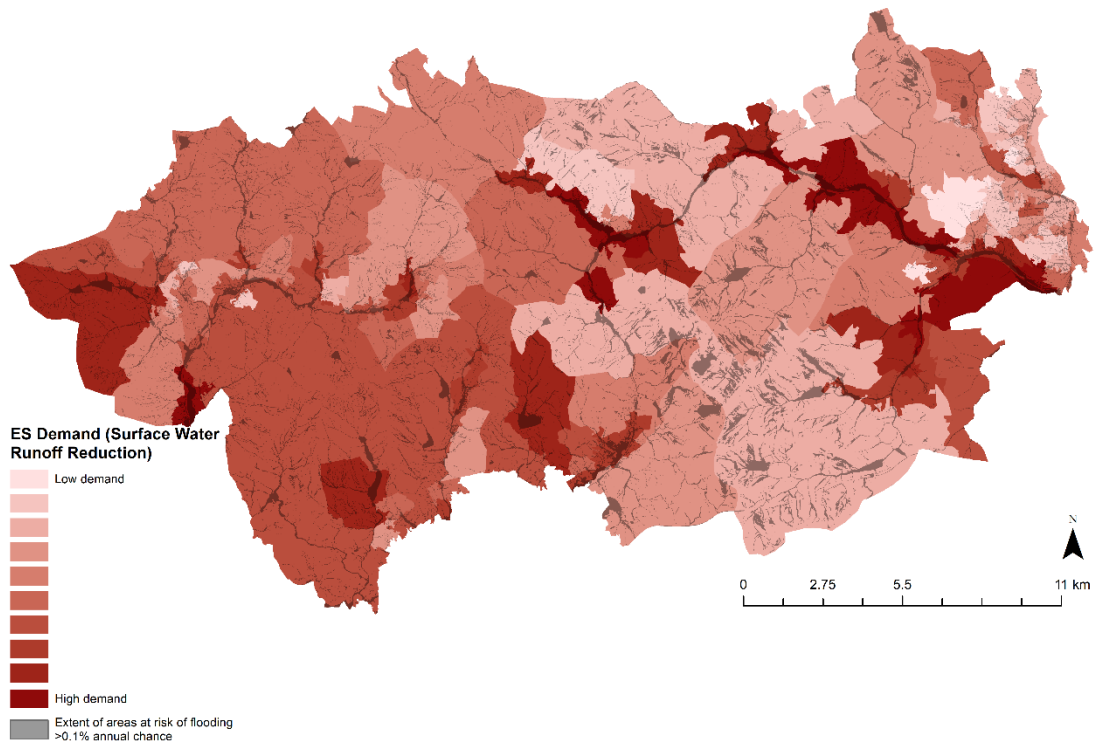


Figure 7.9 The spatial distribution of SWRR demand aggregated to LSOAs (using areal weighted mean) across the South Pennines. Shaded areas indicate the extent of all areas at risk of flooding within the LSOAs.

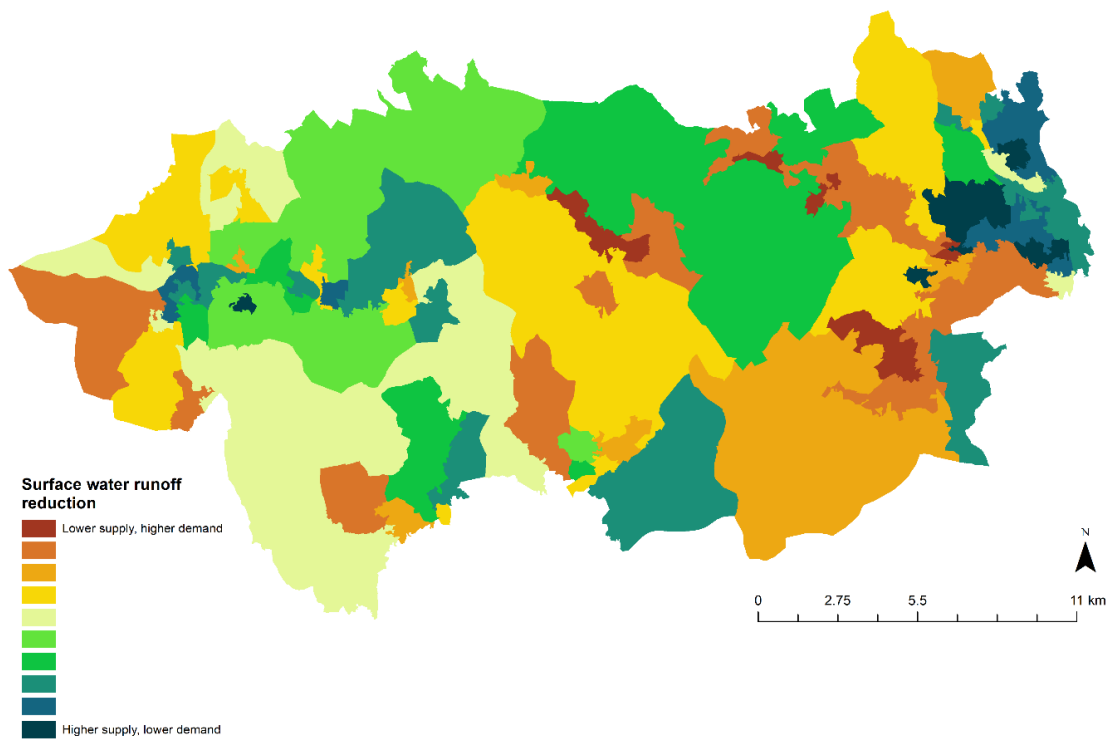


Figure 7.10 The spatial distribution of SWRR net ES index aggregated to LSOAs across the South Pennines.

There is spatial disparity between areas of high supply and high demand, as indicated by their strong significant negative correlation ($\rho = -0.521$). Thus ES supply does not seemingly satisfy demand, which is evident in the net ES index (Figure 7.10). Both supply and demand have very strong significant correlations with the net ES index. Therefore spatially, the net ES index is distributed as described above for supply and demand. It's can also be observed that in the rural LSOAs there is a north to south gradient of higher to lower net ES index.

A significant moderate negative correlation ($\rho = -0.320$) between IMD rank and net ES index coupled with a concentration index of -0.22 indicate ES decreases as deprivation decreases. This is driven by both supply and demand, whereby supply decreases and demand increases as deprivation decreases ($\rho = -0.316$ and $\rho = 0.275$ respectively) (Table 7.6). Thus the most deprived areas tend to benefit more from SWRR and have less need for this service. Boxplots show that the net ES index is highest for LSOAs in the most three deprived deciles, whilst the middle decile tends to have the lowest net index (Figure 7.1). Supply is highest for LSOAs in the two most deprived deciles, although there is also a large range of values for the 9th decile (second least deprived), including LSOAs with highest ES supply. Demand is lowest for LSOAs in three most deprived deciles; median demand peaks for the 5th decile but is generally higher for all of the middle to low deprivation deciles (Figure 7.12).

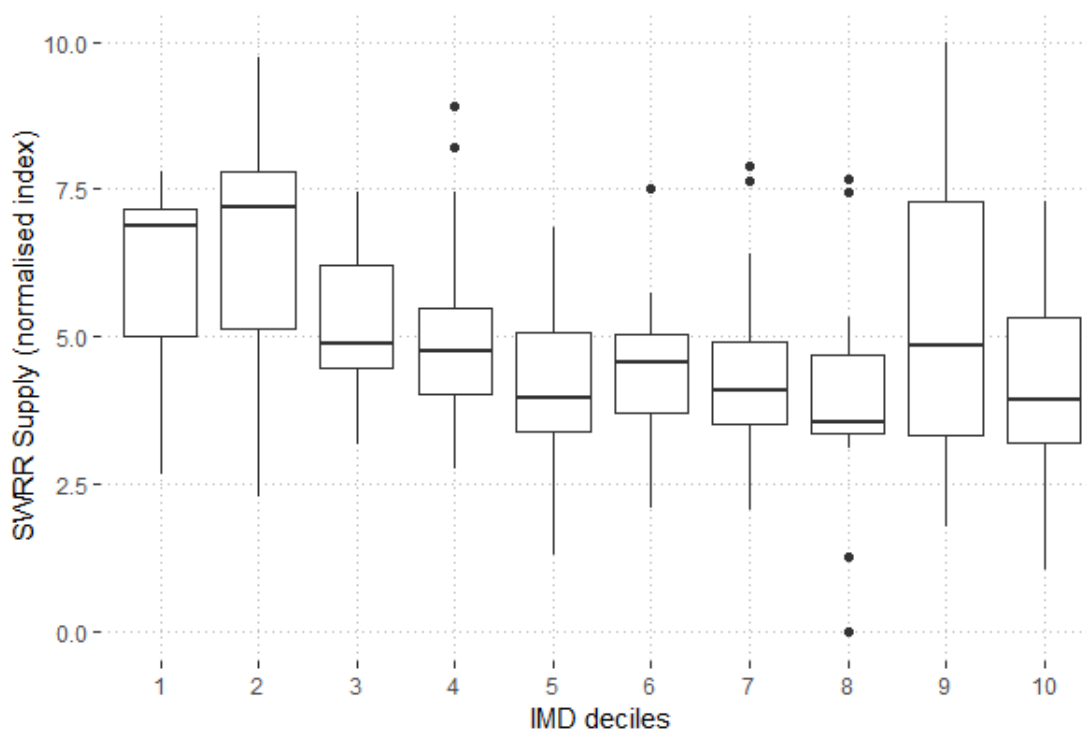


Figure 7.11 SWRR supply per population weighted IMD deciles for South Pennines. Deprivation decreases, decile 1 to 10.

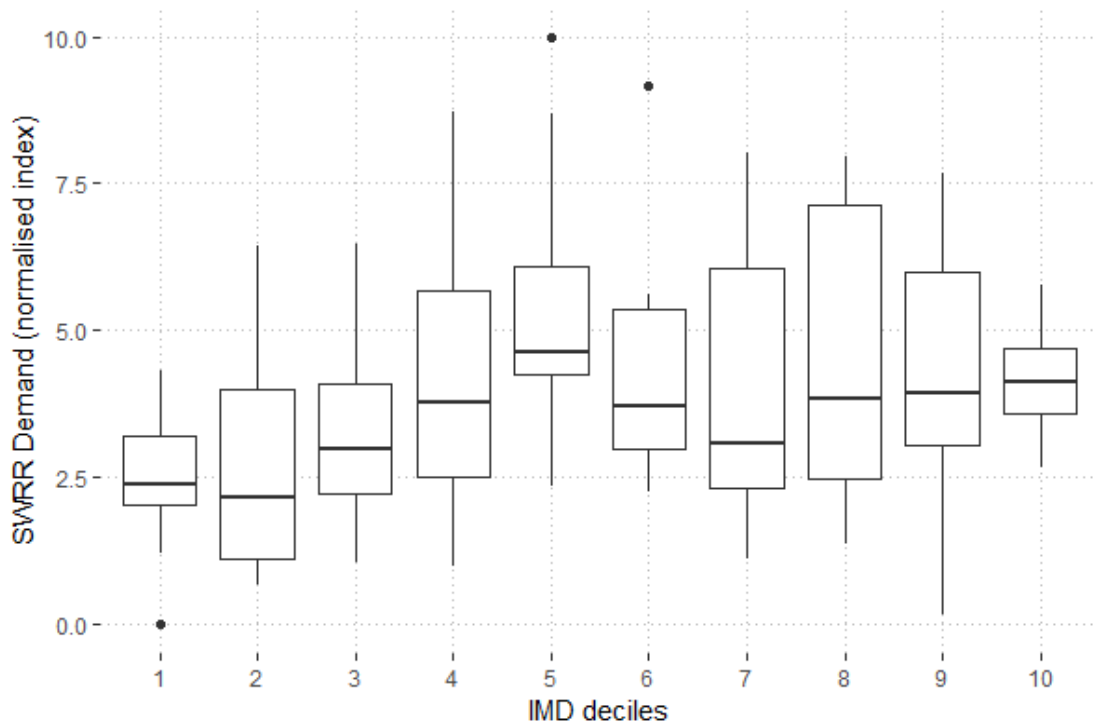


Figure 7.12 SWRR demand per population weighted IMD deciles for South Pennines. Deprivation decreases, decile 1 to 10.

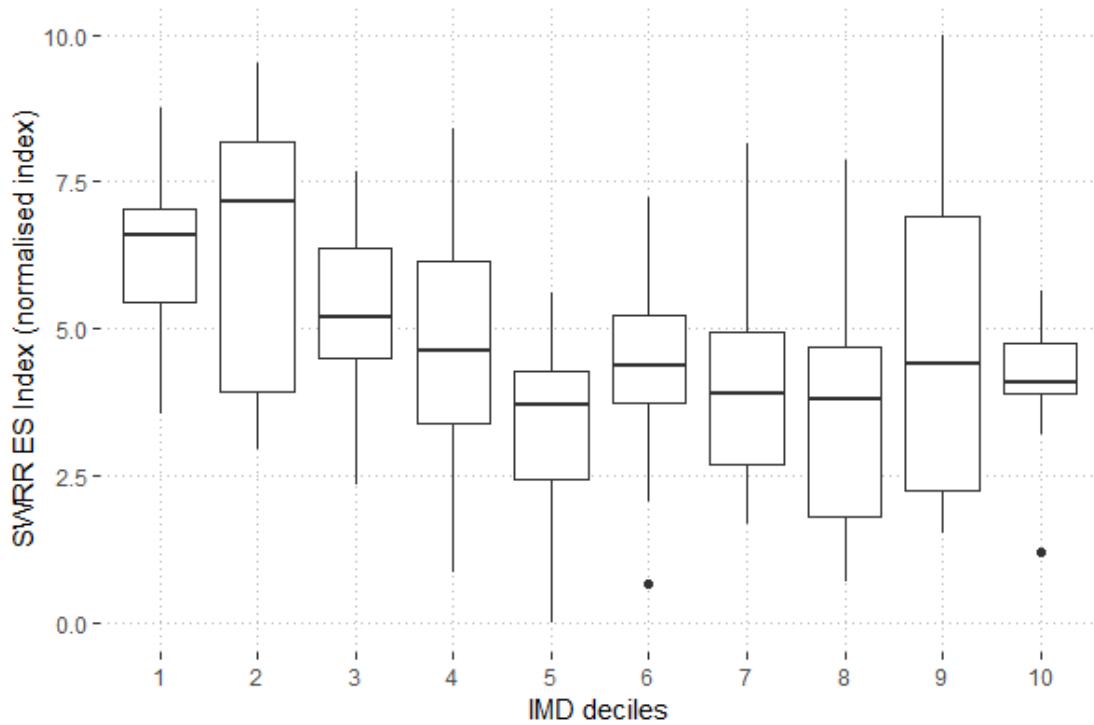


Figure 7.13 SWRR net ES indexes per population weighted IMD deciles for South Pennines. Deprivation decreases, decile 1 to 10.

Local statistics showing spatially dependent associations are, for the majority of LSOAs, not significant (Figure 7.12). Most notably there are very few areas where there is significant local association between high deprivation and low net ES index, this was also the case when tested separately for low ES supply and high ES demand. Where these do occur, they are located around Halifax.

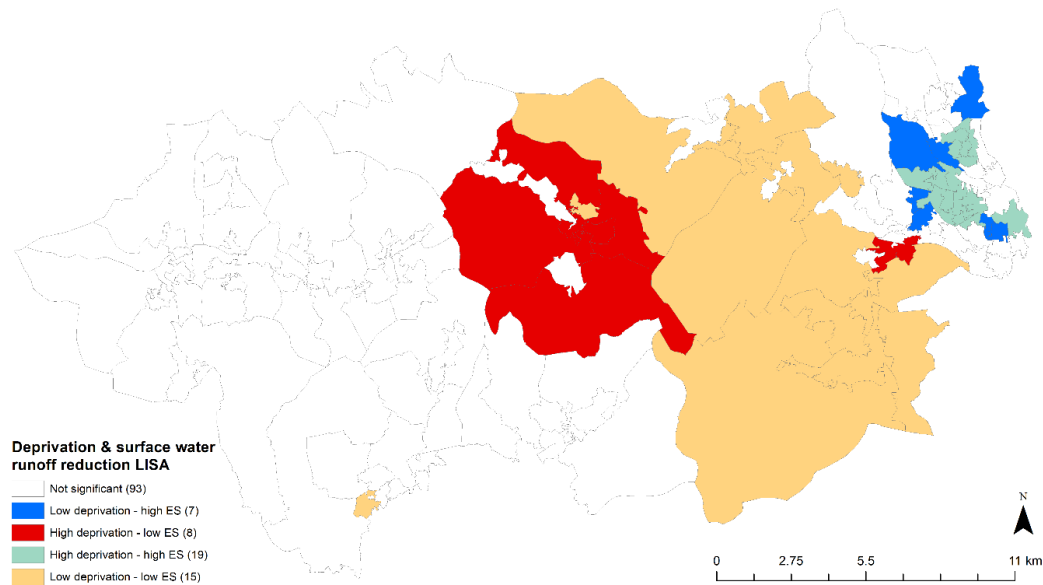


Figure 7.14 South Pennine LSOAs with significant local spatial associations between IMD ranks and SWRR. Identifies clusters of areas where there are significant associations between ES indexes and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend. Parenthesis indicate the number of LSOAs within each class.

7.3.3 The spatial and social distribution of surface water runoff reduction in Northampton

SWRR supply is spatially variable across the Northamptonshire study area (Figure 7.15). Generally, the catchments for Northamptonshire LSOAs are not extensive due to its location near the head of the River Nene and its relatively flat topography. Consequently ES supply is largely determined by the land cover and soil types within and close to the respective LSOAs. There is a single area which has much greater supply compared to all other LSOAs - this is located south of Northampton town in the east of the region and comprises both urban and rural LSOAs.

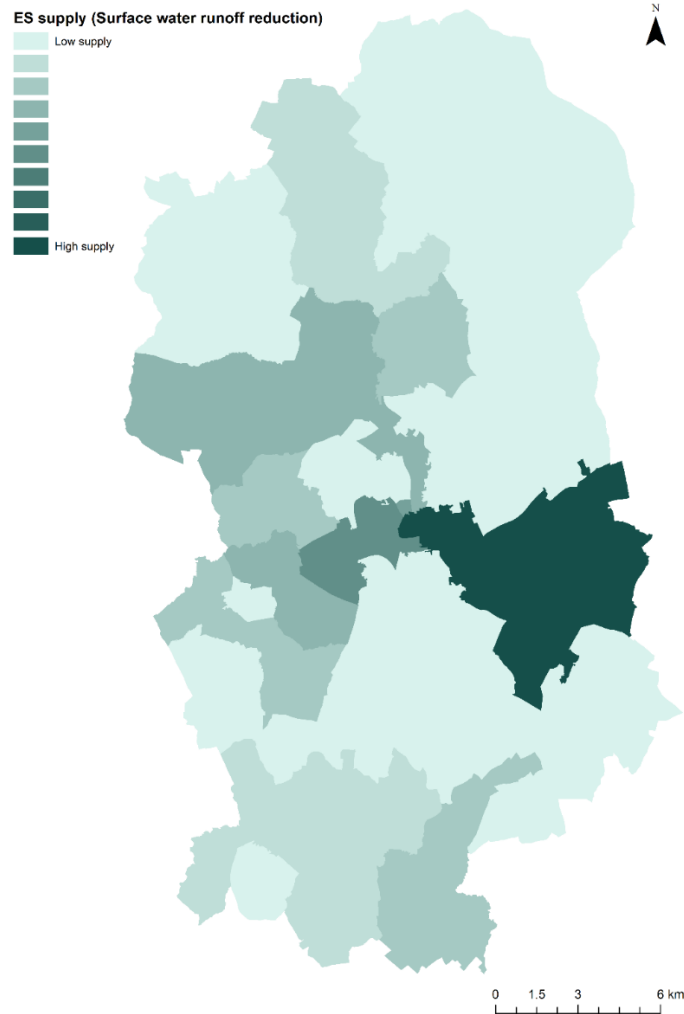


Figure 7.15 The spatial distribution of SWRR supply aggregated to LSOAs (using areal weighted mean) across the Northampton region.

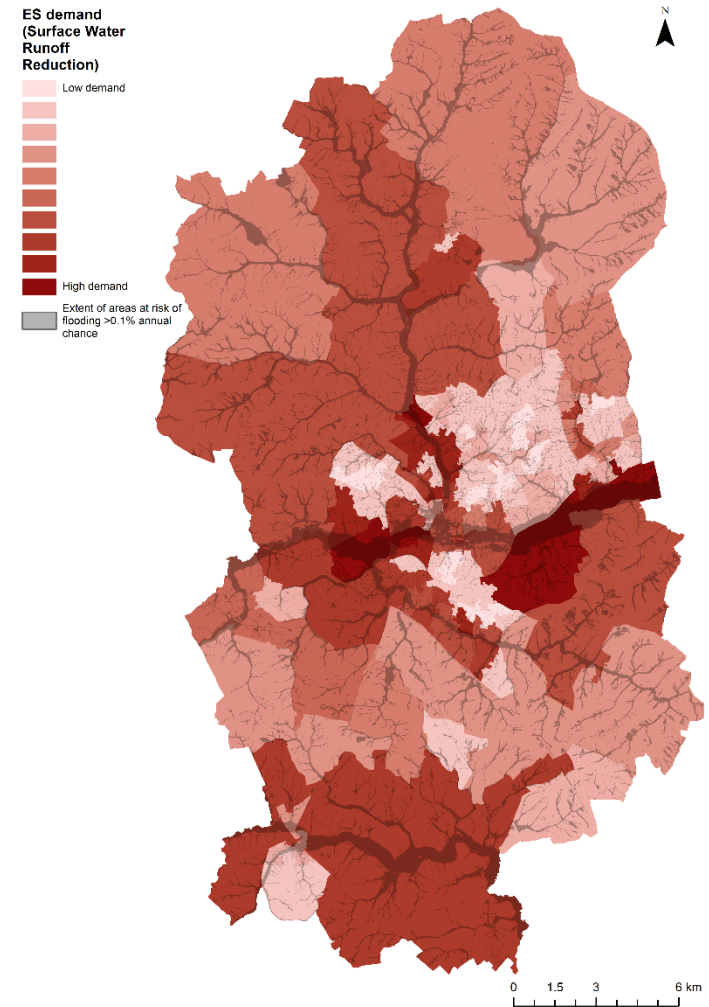


Figure 7.16 The spatial distribution of SWRR demand aggregated to LSOAs (using areal weighted mean) across the Northampton region. Shaded areas indicate the extent of all areas at risk of flooding within the LSOAs.

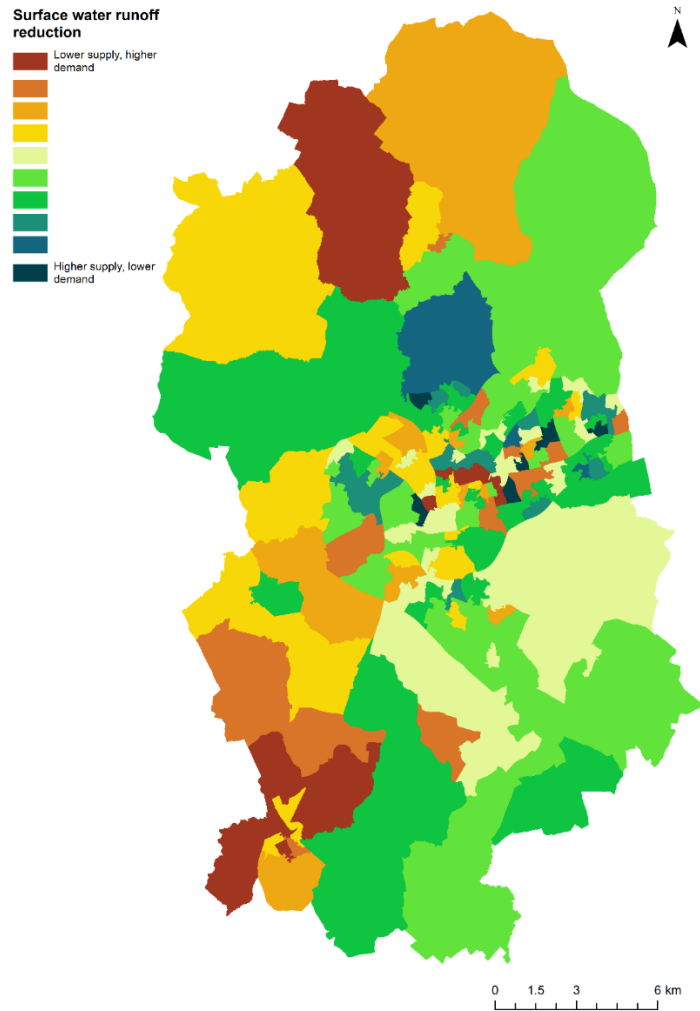


Figure 7.17 The spatial distribution of SWRR net ES index aggregated to LSOAs across the Northampton region.

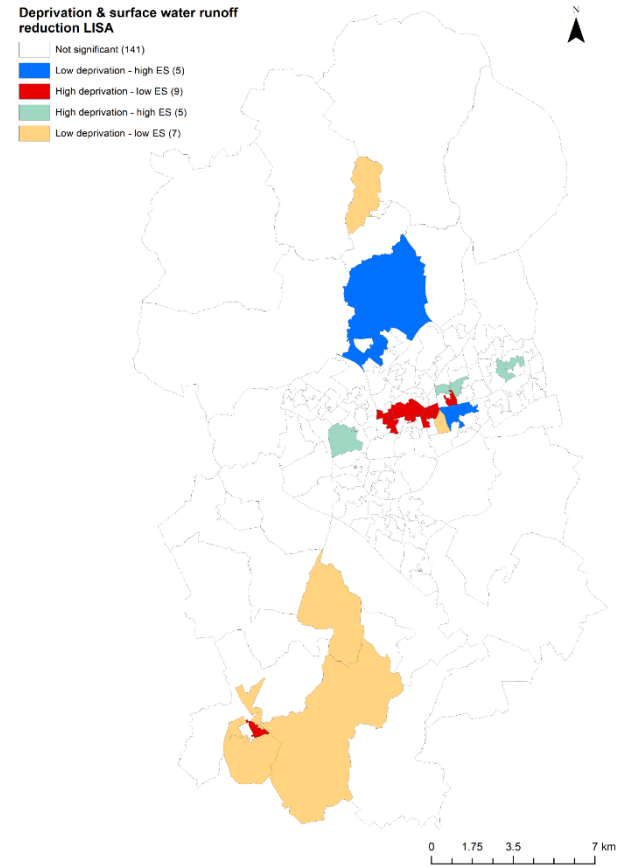


Figure 7.18 Northampton LSOAs with significant local spatial associations between IMD ranks and SWRR. Local indicators of spatial association (LISA) identify clusters of areas where there are significant associations between net ES index values and IMD ranks in the same direction (95% confidence level). The direction of associations are given in the map legend.

These LSOAs share the same catchment and higher supply is attributed to the greater coverage of woodland compared to other catchments. Otherwise rural areas to the west of Northampton town have a comparatively higher supply than elsewhere in the study region.

The spatial pattern of ES demand echoes that of supply, in particular, it is highest along the River Nene to the southeast of Northampton town (Figure 7.16). There is also high demand in South Northamptonshire's rural LSOAs located at the southern tip of the study region. Here, higher demand is attributed to the presence of Wootton Brook, a tributary of the River Nene.

Comparable spatial distributions of ES supply and demand is confirmed by their very strong significant correlation ($\rho = 0.872$). This produces a net ES index that is spatially scattered (Figure 7.17), since places of high supply and high demand effectively 'cancel' each other out (although note that they are based on relative and not absolute values). Whilst there is some clustering of similar net ES index values in rural areas, for example of lower supply and higher demand in the very north and southwest of the study area, there is no distinguishable pattern in ES index across Northampton town.

Deprived areas are spatially concentrated in Northampton, with only a few pockets of lower deprivation dotted around the rural areas in South Northamptonshire and Daventry. There is therefore low spatial coincidence with the SWRR net index across the majority of the region (Figure 7.18). The lack of association between net ES index and deprivation levels is confirmed by correlation coefficients (Table 7.6) and a concentration index close to zero (-0.04). These indicate no significant relationship between deprivation and net ES index, and no significant relationship between deprivation and ES supply and ES demand individually. The distribution of the net ES index across deprivation deciles is visualised in the boxplot in Figure 7.21. There are slightly higher net ES index values for the middle deciles and interquartile ranges indicate more LSOAs in the most deprived deciles have lower net ES. However, in general the boxplot confirms there are no major differences in net ES across deprivation deciles.

The result from the local analysis are shown in Figure 7.18. There are few LSOAs with significant spatial correlation between deprivation and ES (26 out of 167 LSOAs). Only one LSOA with spatial correlations indicating a significant association between high deprivation and low ES occurs outside of Northampton. Whilst the majority of less deprived areas with high deprivation occur in the south of the study region.

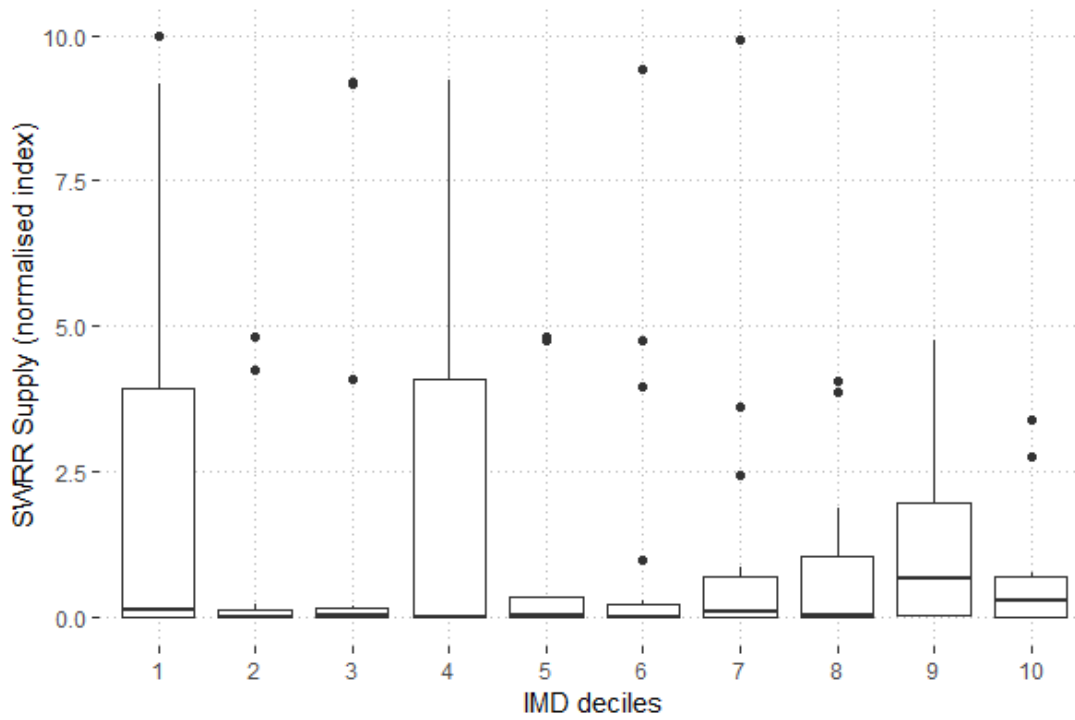


Figure 7.19 SWRR supply per population weighted IMD deciles for the Northampton region. Deprivation decreases from decile 1 to 10.

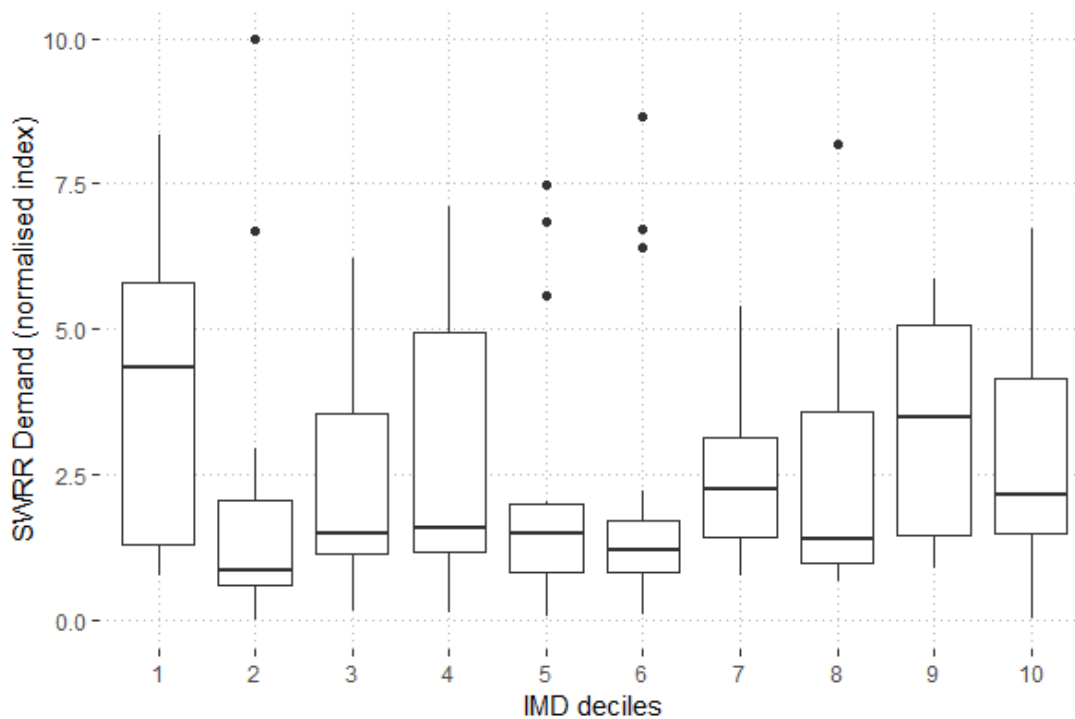


Figure 7.20 SWRR demand per population weighted IMD deciles for the Northampton region. Deprivation decreases from decile 1 to 10.

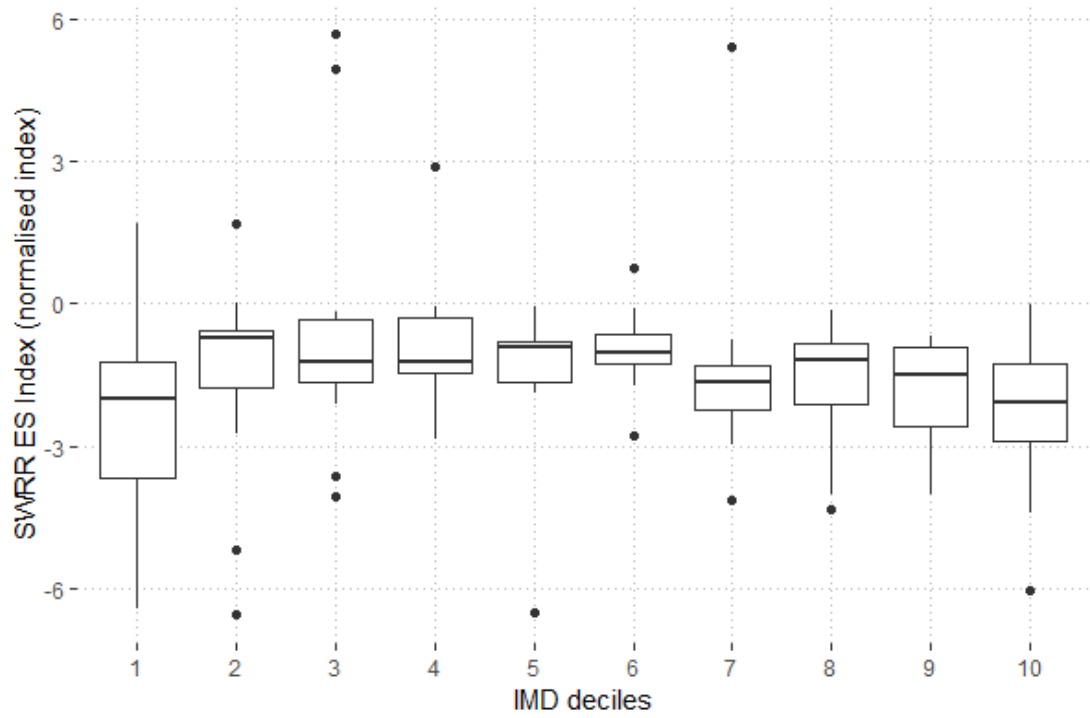


Figure 7.21 SWRR net ES indexes per population weighted IMD deciles for the Northampton region. Deprivation decreases from decile 1 to 10.

Table 7.6 Spearman rank correlation coefficients showing linear associations between SWRR ES and IMD ranks.

a) Leeds

		IMD ranks	ES supply	ES demand	ES index
	IMD ranks	-	0.386**	0.113*	0.226**
	ES supply	0.386**	-	0.454**	0.275**
	ES demand	0.113*	0.454**	-	-0.647**
	ES index	0.226**	0.275**	-0.647**	-
Sensitivity	'High influence' ES index	NS			
	'Low influence' ES index	-0.133**			
	Population weighted ES demand	-0.319			
	Population weighted ES index	0.435**			

b) South Pennines

		IMD ranks	ES supply	ES demand	ES index
	IMD ranks	-	-0.316**	0.275**	0.320**
	ES supply	-0.316**	-	-0.521	0.858**
	ES demand	0.275**	-0.521	-	-0.860
	ES index	-0.320**	0.858**	-0.860**	-
Sensitivity	'High influence' ES index				
	'Low influence' ES index				
	Population weighted ES demand	-0.269**			
	Population weighted ES index	NS			

c) Northampton

		IMD ranks	ES supply	ES demand	ES index
	IMD ranks	-	NS	NS	NS
	ES supply	NS	-	0.872**	-0.337**
	ES demand	NS	0.872**	-	-0.622**
	ES index	NS	-0.337**	-0.622**	-
Sensitivity	'High influence' ES index	0.169*			
	'Low influence' ES index	NS			
	Population weighted ES demand	-0.362**			
	Population weighted ES index	0.210**			

IMD ranks increase as area deprivation decreases; a positive correlation indicates net ES index increases with decreasing deprivation.

Correlations for baseline calculations are given for IMD and net ES index, and IMD and ES supply and demand separately.

For sensitivity tests, correlation with only IMD ranks are computed for sensitivity of net ES index given variations in ES supply (including a 'high influence' scenario where vegetated land is assigned a greater capacity to reduce runoff; a 'low influence' scenario where vegetated land is assigned a lower capacity to reduce runoff; and a scenario where the SPA is defined as the LSOA only, not for catchments – see section 7.2.1). Correlations with IMD ranks are given for variation in demand given a population weighting and the net ES index given this weighted demand.

NS no significant correlation **Correlation significant at 99% confidence level

*Correlation significant at 95% confidence level.

7.3.4 Sensitivity tests

The sensitivity of the social distribution of SWRR net ES index is tested with respect to assumptions in the conceptualisation of supply and demand and to the uncertainties in CNs assigned to describe the relative abilities of different land covers to reduce runoff.

Tests of the uncertainties in assigned CNs were conducted only for Northampton and Leeds, the first case studies analysed (Table 7.6), as the long processing times involved limited the opportunity to conduct sensitivity tests all the case studies. For Northampton, there was no change in the results given CNs that represent an enhanced influence of vegetated land cover on SWRR (the 'high influence' scenario – see Table 7.3). In this case there remained no significant correlation between the ES and deprivation. Similarly, under this scenario for Leeds no significant correlation was found, however, this contrasts with significant positive correlation between the ES index and IMD determined under the 'best estimate'.

For the 'low influence' scenario whereby vegetation has less impact on SWRR, a different social distribution is revealed from that determined under the 'best estimate' scenario. In Northampton, there is weak positive correlation (95% confidence level), indicating some inequality, with net ES index decreasing as deprivation increases. This compares to no significant relationship given a 'best estimate' of CNs. For Leeds, the SWRR ES spatial distribution is notably changed, with a higher net ES index in many of the suburban and urban areas. This is likely due to the much lower impact assigned to the agricultural land in the northern areas in reducing runoff. As a result, for the 'low influence' scenario the relationship between net ES index and IMD is reversed, indicating a weak association between lower deprivation and higher ES.

Overall, there is some sensitivity to the parameters assigned within the SWRR ES modelling which can result in a different interpretation of its social distribution. This indicates a need for further, more detailed analysis. Chapter 5 presented results of the correlation analysis of deprivation and percentage greenspace derived from the Generalised Land Use Database (GLUD). This provides some insight as to how well assessing inequalities using this easily accessible metric is representative of inequalities in some of the actual ESs generated by greenspace. Comparison to the correlations with deprivation for all variations of SWRR index reveals similar results only for the net ES index in Leeds and the net ES index for the South Pennines which incorporates population density within demand. Otherwise results are inconsistent, indicating proportion of greenspace to be a poor proxy to use in assessing the inequality of SWRR.

Sensitivity to the conceptualisation of ES demand is carried out consistent with previous analysis, i.e. where surface water regulation ES is demand not by an individual, but by an area's population. This involves reassessing ES demand as a function of the size of the benefiting population. This test was conducted for all case study regions. Crucially, this test shows greater consistency in the findings across the three regions, namely, a significant moderate negative correlation is found between ES demand and IMD ranks. Thus once population weighted demand is considered in conjunction with ES supply, there is an apparent increase in inequality in the distribution of net ES index compared to the ES index original calculations. This results in a stronger degree of inequality for Leeds ($p = 0.435$ compared to $p = 0.226$) and indication of some inequality for Northampton ($p = 0.210$ compared to non-significant results). Whilst for the South Pennines, no significant correlation is found but this compares to a previous finding of negative correlation (a higher ES index for more deprived areas). The implication of this sensitivity test is that reducing ES demand (i.e. the risk of flooding) for the more deprived areas would also meet the needs of the many, rather than the needs of the few.

7.4. The social distribution of surface water runoff reduction in the case study regions

The results described above reveal that the social distribution of SWRR is different for each case region, in addition the findings should be used with caution since they have . Figure 7.22 provides a visual comparison of the three regions using concentration curves, associated with the concentration indexes, where increasing divergence of a curve from the line of perfect equality, indicates rising inequality. These findings,

The concentration curves reinforce the finding of inequality, whereby more deprived areas benefit least from SWRR, only occurs within the Leeds region. This association was consistent regardless of the changes in estimates of net ES index as applied in the sensitivity tests. However, the concentration curve for Leeds lies closest to the line of equality compared to those for the other regions. For Northampton there was no strong indication of inequality, although under two scenarios (when ES demand is population weighted, and when the role of vegetation in reducing runoff is increased) a decrease in net ES index with increasing deprivation was detected.

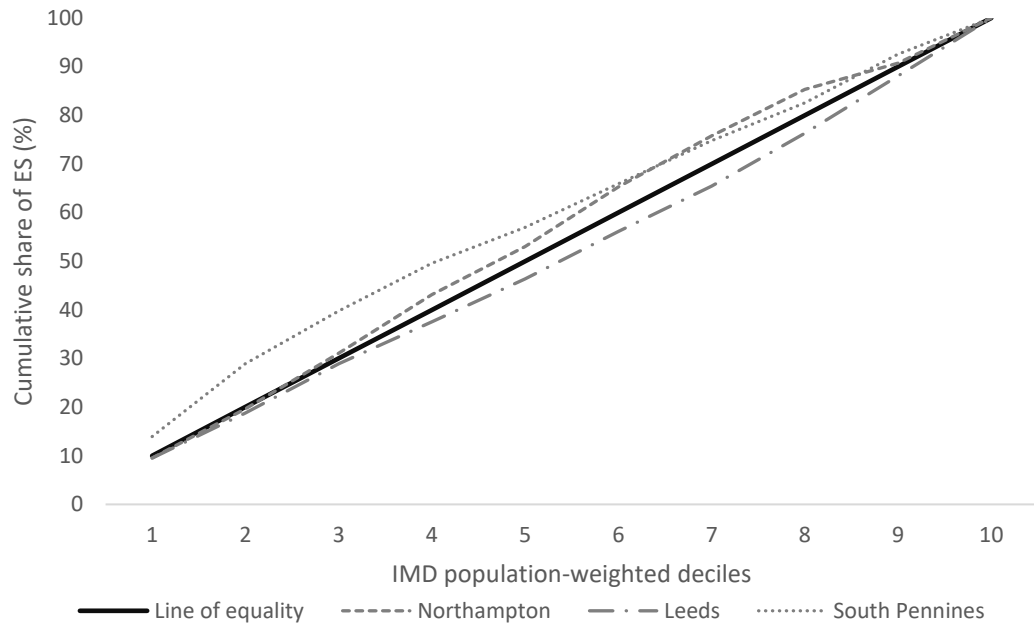


Figure 7.22 SWRR concentration curves for net ES index.

IMD decile 1 represents most deprived population-weighted decile; IMD decile 10 represents least deprived population-weighted decile.

Line of equality shows theoretical equal distribution across all deciles. Concentration curves show distribution of SWRR net ES index according to deprivation for each case study. Less than 'equal' share is indicated by the curve falling below the line of equality, a greater share is indicated by curve rising above the line of equality.

This is reflected in the concentration curve, which does not fall below the line of equality but does show the less deprived share a greater proportion of the ES. A reverse association was revealed in the South Pennines, as shown by the greatest divergence of the concentration curve above the line of equality for the most deprived deciles. Although the sensitivity tests indicate that this finding varies given different model estimates of supply and under different conceptualisations of demand. When demand is based solely on flood hazard, it is greatest for the least deprived areas. However, under a utilitarian perspective (greatest benefit for most number of people), if demand is considered higher for areas which have a higher flood hazard and population density then it is greatest for the most deprived areas. It is noted however that once considered in conjunction with ES supply, there is no significant correlation with deprivation. Overall the results indicate that the social distribution of the ES of SWRR in England is place specific, but these findings should be used with caution given sensitivities to model inputs and due to issues of ecological fallacy arising from generalisation across LSOAs.

Chapter 8 Discussion

Preliminary insights into inequalities in the distribution of ecosystem goods and services were determined through the first national analysis of a comprehensive set of natural capital indicators and subsequent comparison to deprivation at the district level in England (Chapter 4). The characteristics explored can be considered as two of the 'states' underpinning the socio-ecological system illustrated by the research conceptual framework (Figure 3.6; Chapter 3). The analysis revealed that the most deprived communities in urban areas are characterised by an overall low extent and quality of natural capital, but that simultaneously there are rural deprived areas characterised by the highest quality and extent of natural capital. Deprivation is lowest in districts typically featuring the highest proportion of woodland coverage and publically accessible spaces and also in districts which are largely agricultural but with inaccessible natural environments. These findings demonstrate that the distribution of natural capital does vary by social deprivation, however, a lack of consistent pattern of inequality at the national level implies that equitable management of ecosystems should be addressed at a sub-national level. This emphasises the need for a more in-depth analysis at a more local level as presented by the case study analyses in Chapters 5-7.

The case studies of the Leeds, Northampton and South Pennine regions were used to examine the distribution of ES by deprivation. Natural capital classification carried out in Chapter 4 provided a broad insight into the different types and quality of natural capital characteristics of these districts, comprising of rural and urban areas with varied natural capital profiles and strong gradients in socio-economic deprivation. Distributions of two regulating ES – air pollutant removal and surface water runoff reduction, and one cultural ES – recreation were assessed (the selection of which were discussed in section 3.2). Each of these exhibit direct, quantifiable ES flows (of particular relevance to high income countries) from natural capital to beneficiaries on local to regional scales, with distributions which are affected by local and national land planning policy.

Variations in ES supply, demand and the net ES index (supply less demand) within the study regions were modelled in relative terms and aggregated to small area geographies

(LSOAs). This accounted for the flows of ES from more distant natural capital. Typically the supply side has dominated the literature, but increasingly, there is greater recognition for ES research to better consider the beneficiaries and their demand for ES. This is critical to the concept of ES which is based upon the benefits of ES to human health and wellbeing, and in particular for quantifying and addressing potential inequalities. In this case, the beneficiaries were defined as the population living within the case study LSOAs, with the Index of Multiple Deprivation (IMD) used to differentiate their socio-economic conditions. Demand for the ESs was determined as the level of need (i.e. exposure to a hazard) for the selected ESs. This need criteria was applied because it is considered that in the absence of need (e.g. no flood risk) then there is no benefit from a service (e.g. which reduces flood risk) supplied to that area.

Results from the analyses, which address objectives 2 (characterising the social distribution of ES) and 3 (sensitivity tests), were detailed for each ES individually in Chapters 5-7, with further reflection on these results detailed below. Sections 8.1 and 8.2 review results on a case study basis and per ES respectively, thus considering the wider implications of results. Any inequalities observed may be perceived as fair or unfair hence these different perspectives, which are important for understanding how inequalities may (or may not) be addressed, are discussed in section 8.3. The discussion then broadens to consider opportunities for policy and practice to address ESs within an inequalities framework. This discussion first addresses better accounting for distributional justice with respect to ESs, through for example national policy and guidance (section 8.4.1), and in practice through ES assessments and tools (section 8.4.2), taking methodological insights from the assessment carried out in this research. Section 8.5 then considers how information regarding the social distributions of ESs can then be considered in decision making for sustainable land management and planning. The final section (8.6) concludes by considering some limitations of the analysis and opportunities to refine and build upon the research.

8.1 How equally are the selected ESs socially distributed in three case study regions?

8.1.1 Leeds

ES inequality was found to be most prevalent in Leeds compared to the other case study regions. Both air pollutant removal and surface water runoff reduction ES decline with increasing deprivation (with respect to net ES indexes). Notably, the strongest association between deprivation and air pollutant removal is revealed for Leeds and it is the only case study whereby surface water runoff reduction is lowest for most deprived areas. No significant relationship is found between recreation net ES index and deprivation: as deprivation increases availability of public greenspace declines, but this effect is offset by increasing provision of public rights of way.

Supply is the key factor in the association between surface water runoff reduction and deprivation in Leeds. Leeds is also the only case study whereby air pollutant removal supply decreases with increasing deprivation. Considering patterns across all ESs jointly, Figure 8.1 shows the proportions of total supply/total demand in Leeds for deprivation quintiles. This illustrates overall ES supply is lowest for LSOAs containing the most deprived 20% of the population, whilst simultaneously demand is highest. By contrast the least deprived 20% population have the greatest share of benefits but lowest demand. Although values for supply are relative to overall supply across the region and similarly, values of demand are relative to overall demand across the region, they are not relative to each other (i.e. even if supply and demand have the same value this does not mean that the ES supply fulfils demand). Regardless, it remains evident that there is a disproportionate distribution of ESs in Leeds.

Overall the economy in Leeds is strong, but for considerable time Leeds City Council has been concerned by its two-speed economy whereby robust economic growth overall has not benefitted most inner city residents, who live in areas of severe deprivation (Boyle & Alvanides, 2004). Indications of an unequal economy persist, for example, Leeds has an above average employment rate and the 8th greatest increase in private sector jobs 2016-2017 (Centre for Cities, 2019). However in 2015, Leeds was also identified as the third most unequal city on the basis of Job Seeker's Allowance claimant rates (Centre for Cities,

2015) and 22% of Leeds LSOAs are in the 10% most deprived neighbourhoods nationally (LCC, 2016).

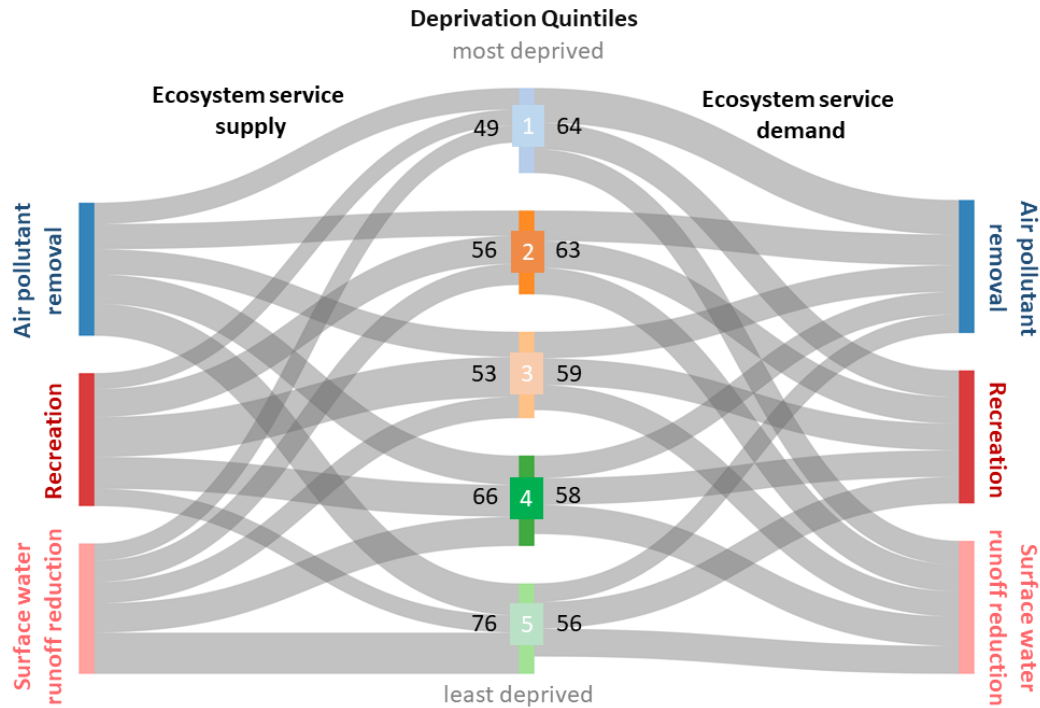


Figure 8.1 Flows of ES supply and demand stratified per population weighted deprivation quintiles in Leeds

Flows of ES supply and demand as percentage of totals of each service going to each deprivation quintile (based on IMD). Thickness of grey lines denotes share of flow. Deprivation is represented by the central coloured nodes, and decreases towards the bottom. The numbers in black show that for the three ES combined, overall supply increases as deprivation decreases, whilst demand decreases. See Appendix A4 for further guidance on interpretation of this diagram.

In comparison to other city local authority district boundaries, the Leeds district extends beyond the city boundaries and incorporates more woodland than typical of other cities. Furthermore it is located close to the Yorkshire Dales National Park and is considered to have good coverage of greenspace within the city. Despite this, district level analysis showed that natural capital extent and quality is close to the national average (Chapter 5,

section 5.2). In comparison to the town of Northampton as another densely built up area, several of the natural capital indicators in Leeds (e.g. woodland) are slightly higher but natural capital across the whole Northampton case study region is higher overall (Chapter 5, Figure 5.8).

Thus in comparison to our other case study regions Leeds suffers from greater inequalities in the assessed ESs and lower overall natural capital within the district boundaries, indicating a greater need for both net, and more equitable, increases in natural capital and ESs. The need to improve access to environmental benefits for all within the district is asserted within the Green and Blue (GBI) Infrastructure Strategy (West Yorkshire Combined Authority, 2017) and the Health and Wellbeing Strategy (Leeds Health & Wellbeing Board, 2016). In particular equal access to the benefits of recreation, flood risk reduction and clean air are noted. Specifically, the strategy aims for everybody to be “within easy reach (1km) of an outstanding, diverse, well used GBI network” (West Yorkshire Combined Authority, 2017 p.2). It also highlights the need for more integrated flood management using natural flood management approaches, explicitly recognising the importance of these benefits flowing from areas beyond the district boundaries. Despite these assertions, there has been little research regarding the distribution of greenspace within Leeds, and none which examines the social distribution of the ESs. It is therefore considered that the knowledge generated from this analysis is of particular relevance and potential use to the current challenges faced within the district.

8.1.2 Northampton

For Northampton, the case study region was extended beyond the town’s district boundaries to encompass some of the rural, less deprived LSOAs within surrounding districts. However compared to Leeds, which is bordered by uplands, woodland and agriculture, Northampton is a smaller urban area and surrounded by districts where agriculture dominates. Throughout the study region there is also a higher than national average extent and quality of inland water (Figure 5.8), which is significant in terms of recreation ES. As with Leeds, deprivation is concentrated within the denser urban areas, but in comparison to Leeds wealthier areas located outside Northampton town are characterised by overall lower natural capital and are also proximal to areas with lower

natural capital from which less ESs can flow from. It is likely that this contributes to the lower inequality in ESs observed in this region. This observation is supported by analysis of the distribution of greenspace recorded in the Generalised Land Use Database (GLUD) which shows a similar weak positive correlation with IMD (0.239 & 0.277 for Leeds and Northampton respectively). This shows that the different distributions of ESs are driven by natural capital beyond the case study boundaries, not just within them.

For Northampton, a lower social gradient in net ES index is found compared to both the Leeds and South Pennines case studies. Notably, Figure 8.2 illustrates that supply for all the ESs assessed is actually greater for the most deprived areas although demand is also greater.

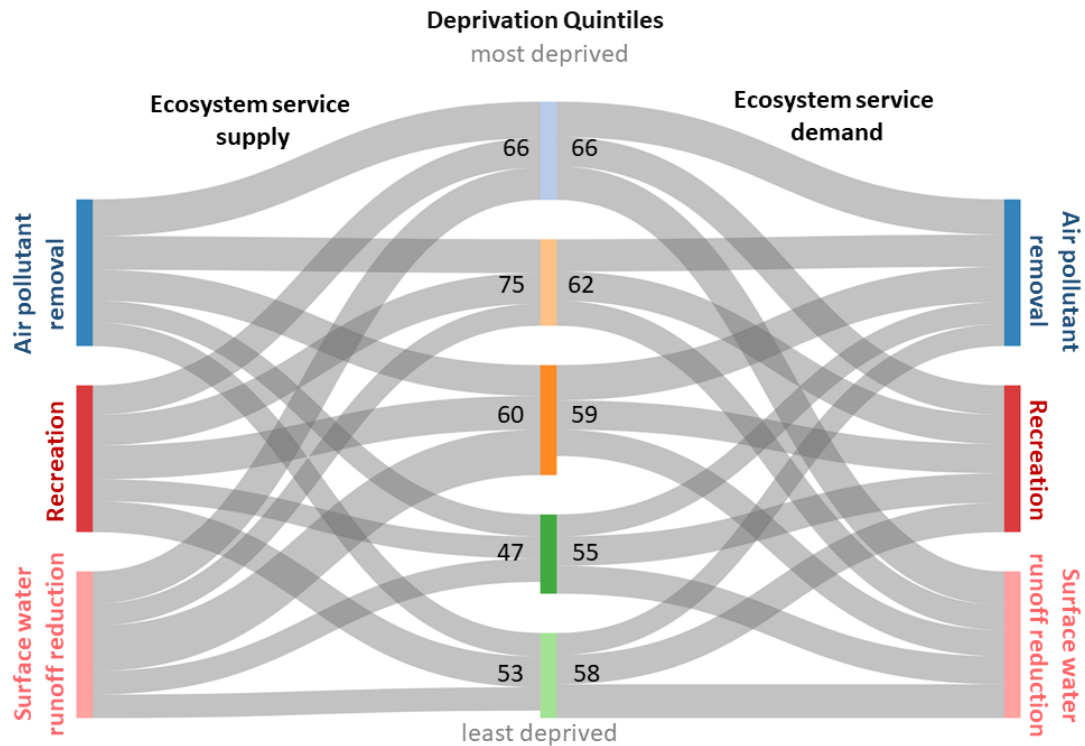


Figure 8.2 Flows of ES supply and demand stratified per population weighted deprivation quintiles in Northampton. Flows of ES supply and demand as percentage of totals of each service across Northampton going to each deprivation quintile (based on IMD). Thickness of grey lines increase as service supply or demand increases. Deprivation is represented by the central nodes, decreasing towards the bottom. Overall supply decreases and demand decreases as deprivation decreases. See Appendix A4 for further guidance on interpretation of this diagram.

Across the case study regions, air pollutant removal net ES index is consistently lower in more deprived areas, however association is weakest in Northampton. Only 9% of LSOAs exhibit significant local association between high deprivation and low air pollutant removal ES (compared to 15% and 16% for the South Pennines and Leeds respectively). However, metrics for air quality related health concerns overall in Northampton fall below English averages (Northamptonshire Local Nature Partnership, n.d.), thus the presence of any inequality in air pollutant removal ES remains pertinent for this region. With regards to recreation, as with Leeds, there are no significant differences across deprivation levels in the Northampton case study. However, converse to Leeds and the South Pennines, more deprived areas have greater access to open greenspaces and lower access to public right of ways. The creation of the large Nene Valley Nature Improvement Area (NIA) (Collingwood Environmental Planning, 2015), which increased availability of accessible greenspace has likely had some impact upon the social distribution of ES given its proximity to Northampton. Longitudinal analysis would be required to confirm this. The distribution of recreation becomes more favourable for more deprived areas when assessed at distances of 1.6km and 3.2km. Therefore it can be asserted that there is a higher presence of accessible greenspace within the town itself and the smaller spatial extent of the town compared to Leeds, leading to lower travel distances to recreational spaces on the edges of the town. This supports the finding from Ala-Hulkko et al. (2016) that opportunities for cultural ESs tend to be greater near urban areas. However, this difference may be less significant if the many historic houses and grounds around Northampton are accounted for. These have been excluded since they charge entry fees, but this may not be a barrier to access for those with higher incomes. These locations are also likely to offer high quality natural environments and provide other cultural services such as cultural heritage, inspiration for art and appreciated scenery.

In contrast to Leeds and the South Pennines, there is no significant association between deprivation and surface water runoff reduction net ES index, supply or demand in Northampton. Interestingly woodland cover and semi-natural grassland which, based on SCS curve numbers (USDA-NRCS, 1986; see Chapter 7, section 7.2.1) are the most effective land covers in reducing runoff, have low coverage in the Northampton region. This indicates that the relatively equal distribution observed, could potentially be attributed

to low variability in runoff reduction across much of the region which has low coverage of the natural capital important for this service. In addition the size of catchment areas are more consistent, reducing the likelihood of differences in ES supply. That said, careful consideration should be given to the interpretation of results regarding the social distribution of surface water runoff reduction as there is some indication of sensitivity to uncertainty in ES estimates. Specifically, when the relative influence of vegetation in reducing runoff is enhanced to align with less conservative estimates of curve numbers (see section 7.2.2), net ES index is lower for more deprived areas.

The analysis for Northampton is not the first which has examined ESs in Northamptonshire (see Rouquette, 2016), however, previous work did not account for the spatial flows of ES nor the implications of ES spatial distribution for inequalities. Overall it is evident that notable inequalities in the social distribution of the three ESs evaluated in the Northampton case study region are not present; and wealthier areas benefit less from recreation ES. This however is in the context of an area where overall many aspects of natural capital tend to be lower (excepting agricultural land and blue spaces). Therefore, if investment is made to increase the extent and quality of certain aspects of natural capital which are lower (see Figure 5.8), such as trees and hedgerows, this would ideally be achieved in a way which does not introduce inequalities.

There are also more nuanced findings showing the presence of some inequalities in some contexts and these should be recognised nonetheless, and potentially addressed dependent on how fairness is perceived (see section 8.3). Perhaps the principle concern of the ESs examined for Northampton case study is air quality which overall is poor, and where the most deprived neighbourhoods have greater exposure and could therefore benefit the most from increase air pollutant removal ES.

8.1.3 South Pennines

In contrast to Leeds and Northampton, the South Pennines case study region is predominantly rural. The natural capital classification undertaken in Chapter 4 indicates the districts included in the case study are amongst the richest in natural capital across England, with the exception of high quality agricultural land. Nonetheless, the South Pennines also incorporates towns where the most deprived areas are concentrated,

although these lie close to areas of high natural capital. In contrast to the other case studies higher deprivation is also observed across several rural areas. Given these spatial patterns it was expected that inequalities would be lower in the South Pennines.

Analysis reveals that in some respects this hypothesis holds true. For example, Figure 8.3 shows that supply of the three ESs is overall greater for the more deprived areas. Similar to Northampton demand for the three ESs considered is also greater in more deprived areas.

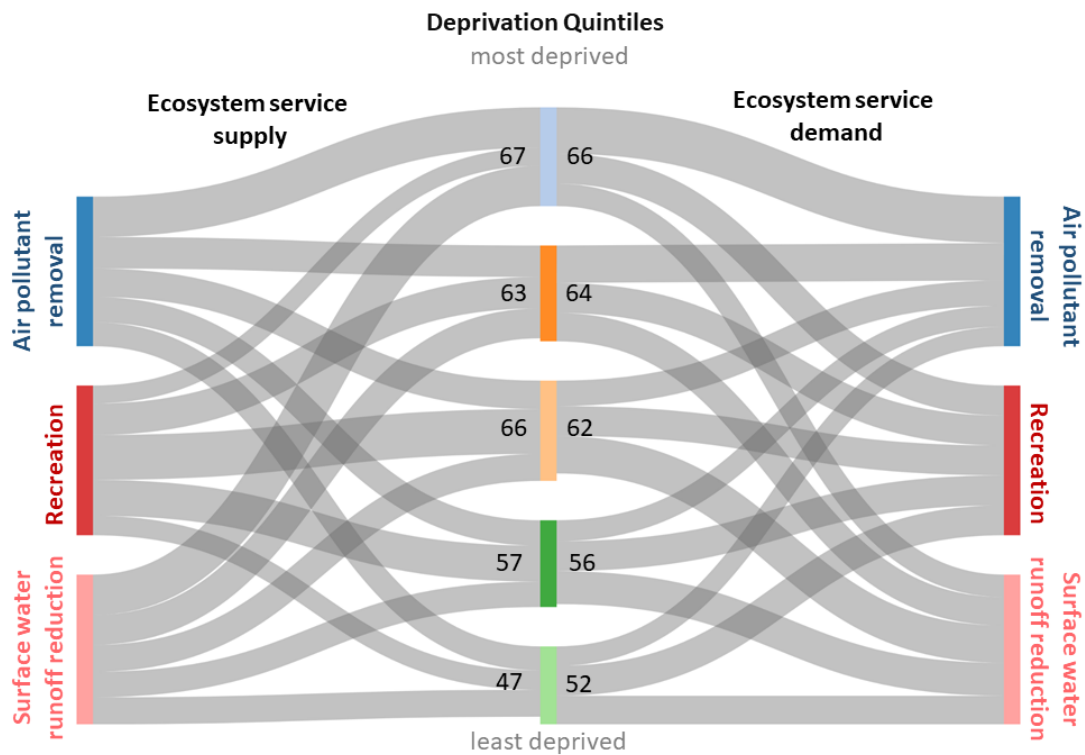


Figure 8.3 Flows of ES supply and demand stratified per population weighted deprivation quintiles in South Pennines.

Flows of ES supply and demand as percentage of totals of each service across the South Pennines going to each deprivation quintile (based on IMD). Thickness of grey lines increase as service supply or demand increases. Deprivation is represented by the central nodes, decreasing towards the bottom. Overall supply decreases and demand decreases as deprivation decrease. See Appendix A4 for further guidance on interpretation of this diagram.

With respect to air pollutant removal, net ES index is lower for more deprived areas, aligning with the results for Leeds and Northampton. Similar to Northampton, this is driven by demand but is not lessened by the distribution of ES supply, which shows no significant difference across levels of deprivation. Overall concerns regarding air pollution are lower across this predominantly rural landscape compared to the dense urban areas of Leeds and Northampton. However, air pollution in rural areas is of growing concern, given pollution from multiple sources such as agriculture and household burning of fuels (DEFRA, 2019). Moreover, the quality of natural capital in the uplands may be adversely impacted by low air quality (Caporn & Emmett, 2009). These factors may have some distributional impacts not accounted for in this analysis.

The South Pennines is the only area where recreation is lower in more deprived areas, and this is emphasised once the number of people requiring the service is accounted for. This social distribution is driven solely by the distribution of greenspaces (not public rights of way). Spatial correlations do indicate that areas with significant association between low recreation and high deprivation are located in the largest town in the case region, Halifax. Therefore, the inequalities present are largely driven by a rural-urban gradient, with much greater access concentrated in rural areas due to the availability of large openly accessible upland nearby. This contrasts with the district of Leeds whereby low recreation and high deprivation was found only for areas outside the city boundaries. It also contrasts with other studies which have found recreation supply increases in more urbanised areas compared to rural areas (e.g. Manchester, UK, Radford & James, 2013). Thus natural capital in the South Pennines overall is more accessible compared to other regions (Chapter 5 section 5.2 shows that the South Pennines have higher than average publically accessible green and blue spaces, whilst Leeds and Northampton have lower than average), but this is concentrated in wealthier areas.

This inequality may also be confounded if the 'open' uplands which although may be within 'reasonable' travel distance (i.e. 10km) are perceived as less accessible by those from lower socio-economic backgrounds; the notion of the English countryside as 'exclusive' for the wealthier was promoted by artists in the Romantic period such as the poet William Wordsworth and there are indications, for example from questionnaire responses, that this has left a legacy in today's society (Suckall et al., 2009). In addition, through greater access, higher socioeconomic groups may also benefit from other potential benefits from

upland natural capital, such as other cultural benefits (e.g. appreciated scenery) and general health and wellbeing benefits (e.g. Maas et al., 2006). A social gradient can also be considered exacerbated by the smaller extents of private gardens found in more deprived communities (see Figure 6.7). Conversely, not all recreation sites are substitutable (De Valck et al., 2017) and opportunities in rural areas, provided primarily by open countryside, do not necessarily fulfil all recreation needs. For example, for people with limited physical ability (see Seeland & Nicolé, 2006 for insight into recreational needs of disabled users) or for carrying out a particular activity (De Valck et al., 2017), or for socialising (Jennings and Bamkole, 2019). Further analysis of preferences for those from different socioeconomic, demographic, ethnic and cultural backgrounds in the local area would be beneficial for teasing apart these desires and needs (e.g. Riechers et al., 2016).

In contrast surface water runoff reduction net ES index is lowest for the least deprived areas in the South Pennines, driven by both ES supply and demand. It has previously been shown in England that residential areas with higher flood risk from rivers tend to be wealthier (Walker et al., 2006). Results for South Pennines ES demand align with this, potentially driven by the general desirability of waterside locations and appeal of culturally desirable towns within the region such as Hebden Bridge (Smith & Philips, 2001) which are at higher risk of flooding given their location in the upland valleys. However, vulnerabilities to flooding tend to be lower for the wealthier who have greater resources and are more able to take out adequate insurance (Walker & Burningham, 2011).

Accounting for these vulnerabilities within ES demand could potentially reduce the strength of, or reverse the direction of association between SWRR net ES index and decreasing deprivation. Moreover, once the number of people affected is accounted for within ES demand there is no significant gradient across social groups.

The South Pennines case study covers two Local Authority Districts, however, joined up approaches for improving the socio-economic and environmental conditions across the South Pennines landscape are facilitated through organisations such as the South Pennine Local Nature Partnership and Pennine Prospects. Multiple projects coordinated by these organisations such as the 'Woodland Heritage' project (www.celebrate-our-woodland.co.uk/) seek to connect more communities from a range of demographic, ethnic and socioeconomic backgrounds with their natural landscape. However, such initiatives require a strong evidence base to help obtain funding. The patterns of inequality in ESs

are mixed in the South Pennines, thus understanding the locations or services of greatest concern can contribute to this evidence base (Pennine Prospects, pers. comms).

Overall, analysis within the case study regions shows the importance of natural capital beyond the immediate boundaries of an area. They also demonstrate that where they may be greater inequalities there may simultaneously be greater natural capital within or near to the region overall (e.g. Leeds compared to Northampton).

8.2 Are there consistent patterns of inequality in air pollutant removal, recreation and surface water runoff reduction in England?

8.2.1 Air pollutant removal

There is substantial evidence that air pollution in England is greater for those of lower socio-economic status (Mitchell et al., 2015). The analysis presented in Chapter 5 adds further weight to these findings for different areas, and shows that demand for air pollutant removal increases as deprivation increases in the Northampton, South Pennine and Leeds regions. Although, air pollutant concentration is estimated using a simple distance decay approach accounting for proximity to roads weighted by traffic flow, the consistency of results with studies based on more accurate modelling gives confidence that the observed patterns are reliable.

The analysis presented in this thesis does, however, extend these existing studies by examining how inequalities in the distribution of air pollutants may be mitigated by natural capital. Across the three case studies, it is found that the distribution of natural capital does not contribute substantively to the inequality in distribution of net ES, but neither does it help compensate for inequalities in ES demand. Moreover, woodland cover was found to be socially distributed at the district level nationally, suggesting that at this scale air pollutant removal ES supply is greater for wealthier areas (in addition to other benefits associated with woodland) although this remains to be fully tested. This presents an opportunity for tree planting for multiple benefits, to be targeted to more deprived areas as part of a holistic approach to reducing social inequalities in air quality, both nationally and

locally. This aligns with criteria for Government funding for planting initiatives in England (Forestry Commission, 2019).

Sensitivity testing for a greater (or lesser) effectiveness of vegetation in removing pollutants has no notable effect on the ES social distribution, indicating results are robust. However, sensitivity testing was not carried out to test the reliability of vegetation mapping from remotely sensed imagery. Evidently, the distribution of trees is critical to the social distribution of this service and greater spatial accuracy of tree distribution and knowledge of tree species and structure could provide more accurate assessment of the distribution of this ES (e.g. Escobedo & Nowak, 2009), and of other ES.

Supply of air pollutant removal is found to be low in rural areas, this may be surprising but can be attributed to two factors. Proximity to roads is a factor in ES supply since effectiveness of vegetation in removing air pollutants increases in areas with higher concentrations of ambient pollutants. Tree planting in urban or suburban areas often occurs along streets, thus creating an enhanced supply of ES compared to trees planted along field boundaries, for example. This echoes arguments that it is essential that the right trees or hedgerows are planted where they can be most effective (Vogt et al., 2017) considering the ES they are intended to supply. As a local ES, whereby the greatest supply is provided by trees near to road networks there is also potential that the distribution of air pollutant removal supply correlates with noise reduction ES, although separate analysis would be required to confirm this.

8.2.2 Recreation

In general across the three case study regions, there is only some evidence of inequality in the distribution of recreation ES, being lower for more deprived areas in the South Pennines only. This contrasts with the common assertion that wealthier areas tend to have more public greenspace (Boone et al., 2009; Dai, 2011), but does support other findings in the UK (e.g. in Bristol - Jones et al., 2009). There is strong evidence from national surveys in England and Denmark that those from lower socio-economic backgrounds and ethnic minorities visit natural environments less often (e.g. Schipperijn et al., 2010; Natural England, 2018a). Jones et al. (2009) and Boyd et al. (2018) find that accessibility is not the primary reason for this, which the results for this analysis in Leeds and Northampton support. For example, the MENE headline report 2016-2018 states only 5% of

respondents quoted 'accessibility' as the principle reason for not visiting the natural environment more often (Natural England, 2018a). However, 34% of respondents also state time limitations as a reason, thus greenspace which can be accessed within a short timeframe and sufficient public rights of way evidently remains important. This is supported by Dallimer et al.'s (2014) assessment for Sheffield, England, where higher use of greenspace is associated with shorter travel times. Overall, physical accessibility is a prerequisite for ensuring access to recreation for all, but strategies for increasing access by lower socio-economic and ethnic minority groups will require a more nuanced approach considering many features of the greenspace (e.g. facilities) and other social factors which influence visitation (Boyd et al., 2018).

The minimum requirements set out by ANGSt do not necessarily account for the time or distance people are willing to spend travelling for recreational purposes (Ala Hukko et al., 2016), thus examining patterns of inequality at different travel distances is important. Schule et al. (2017) found relatively consistent patterns of association between deprivation and recreation at different travel distances. Macedo and Haddad (2015) and Ferguson et al. (2018) also reached similar conclusions under sensitivity testing of the distances applied to model recreation, although they observe that applying distance buffers can overestimate the service supply. In contrast, this analysis revealed different patterns of inequality are revealed depending on location and the way in which recreation is measured, consistent with findings by Wen et al. (2013). In Northampton, when accessible greenspaces and footpaths are considered within walking distances (up to 3.2 km) only compared to use of the ANGSt criteria more deprived areas have greater access. Conversely, for the South Pennines, shorter travel distances increase the gradient of inequality already observed, emphasising that more deprived areas have less access to recreation opportunities within 'walking distance'.

In addition, the association between public rights of way and deprivation differs to that between greenspaces and deprivation. Most studies of accessible greenspace are confined to urban areas and often neglect the role of public rights of way (a review of relevant literature is given in Chapter 2 sections 2.2 and 2.4). The focus upon distribution of open greenspace recognises their support of a wide array of activities for different users. However the role of public rights of way is also important since they improve access to a range of different types of natural capital in areas where open greenspaces are not

viable. Notably, in agricultural production areas (for example in wealthier rural agricultural areas on the outskirts of Northampton) and alongside waterways (for example in Leeds footpaths along waterways provide access to natural capital for more deprived communities who have lower greenspaces). Greater understanding of how different socio-economic and ethnic groups use footpaths in rural and urban areas, and the impacts on health and wellbeing, is needed to fully inform equitable planning of access to recreation.

In this analysis it is also assumed that recreation is a non-rival service, however there is a threshold at which the use by a certain number of users impacts upon the use of the space by others (Norton et al., 2016). Sister et al. (2010) demonstrated a GIS based method for assessing social gradients in potential park congestion, finding for Los Angeles that parks are more likely to be congested in ethnic minority and low income communities. However thresholds will likely depend on the potential user, their perceptions, and the type of space and potential use of space (Wilkerson et al, 2018). For example, the threshold will likely be lower for a space perceived as more 'natural' with less landscaping, maintenance and human infrastructure, since the expectation is that the experience of such spaces should be tranquil (Fischer et al., 2018), although this in itself will depend on the user and their experiences. In contrast to ANGSt guidelines, other guidelines such as those applied in Berlin (Larondelle & Lauf, 2016) incorporate the idea of rivalry since they are based on greenspace provision per capita (Macedo and Haddad, 2015).

Recreation differs to the other ESs assessed in this study as it is only beneficial if a decision is actively made to access the service; surface water runoff reduction and air pollutant removal are passive services. Therefore, the implications of the different social distributions for public rights of way compared to greenspaces and for different distances are dependent upon the needs or preferences of the potential beneficiaries (Jones et al., 2009). As explained in Chapter 6 section 6.2, other characteristics such as the quality, safety and facilities are also important for the perception of greenspace accessibility and the likelihood that the spaces will actually be used (Beichler, 2015). Whilst there is some evidence of a greater likelihood that these characteristics are less favourable in more deprived areas (Wilkerson et al., 2018), this is not universal (Li & Liu, 2016; Ferguson et al., 2018). The case study of Northampton in some ways contrasts with these findings since recreation is as equally accessible physically by more deprived as less deprived communities. However, this does not account for the quality of greenspaces which may

remain lower for more deprived communities. More detailed analysis, involving stakeholders is needed to establish whether recreation provision fulfils the range of needs of local residents across socio-economic groups.

8.2.3 Surface water runoff reduction

Inequalities in surface water runoff reduction are location dependent, with different results for each case study region and are sensitive to uncertainties in models of service supply. There are however some commonalities, firstly, for Leeds and the South Pennines demand increases as deprivation decreases. This aligns with findings more widely across England (Walker et al., 2006). Secondly, there are indications of some inequality in Northampton, in addition to those more evident for Leeds. Specifically, in Northampton once the influence of vegetation in reducing runoff is increased (within upper estimates in the literature) then weak correlation is observed whereby there is some reduction in ES with increasing deprivation.

These findings should be treated with caution in consideration of the ecological fallacy, modelling method and sensitivity to curve numbers applied. The supply and demand for surface water runoff reduction ES are based upon area weighted averages for whole LSOAs, however not all locations within the LSOAs are at risk of flooding and analysis at a finer resolution for example which focusses on households at risk of flooding (e.g. Fairburn et al., 2009) or which uses social metrics available at finer spatial scales could improve the accuracy of results. In addition, the approach taken here does not involve quantifying the actual reduction in flood risk through more comprehensive hydrological modelling, for example as used to map changes in ESs by Eigenbrod et al. (2011) and Warhurst et al. (2014). The change in social distribution in Northampton and Leeds under sensitivity testing suggests that hydrological modelling would help to clarify the social distribution of surface water runoff reduction ES across the case study regions, and any potential impacts on this from development and changes in flood management. Given the efficiency of the curve number approach for assessing surface water runoff reduction ES, it would be beneficial to further evaluate the uncertainties arising. Ideally this would entail clarification of most suitable curve numbers to apply to UK land cover and soil data, given the range of values which have been applied (Tratalos et al., 2007, Holt et al., 2015).

Natural flood management (NFM) whilst not a new concept, has received increasing recognition for its potential to build greater resilience to flooding whilst providing multiple additional benefits (Environment Agency, 2017b; HMG, 2018). The Environment Agency in their management of river catchments, also accept social responsibilities, of which one is “Addressing environmental inequalities” (Environment Agency, 2004). In line with these commitments, implementation of NFM schemes through investment in natural capital presents an opportunity to more widely examine and, if required, address inequalities in surface water runoff reduction ES and concurrently other ESs. This is of particular relevance to Leeds where inequalities have been demonstrated and there is a strong interest in NFM (West Yorkshire Combined Authority, 2017).

8.3 What is a fair distribution of ESs?

Justice may be conceived with respect to participation, recognition and/or the distribution of outcomes (see Chapter 2, section 2.2.1; Agyeman, 2002). For some, the implication is that distributions of ESs observed are not as important as the means through which they arose (Cutter, 1995). For example under libertarianism, economic based mechanisms and processes guide environmental management and decision making, and the distribution of benefits is of no concern. However, the natural and socioeconomic processes which have shaped the urban and rural landscapes and thus the distribution of ESs are complex and historical. Byrne (2012) & Wolch et al. (2014) highlight park design philosophy, historic land development, changing cultures of recreation and histories of minority and socio-economic class oppression as some causal mechanisms explaining inequality in urban greenspace. Another, mechanism often observed within environmental justice literature is that of house pricing and residential mobility which is greater for high-income households who may choose to locate in greener areas. Simultaneously they also tend to benefit from conservation policy and have greater collective resistance to development (Lovell & Taylor, 2013; Wilkerson et al., 2018). Insight from longitudinal analysis is needed to better understand causality (Mitchell et al., 2015), although is constrained by data availability. Understanding these mechanisms is a particular challenge for these case study regions given their size, the inclusion of rural-urban landscapes and the diverse set of natural capital which contribute to the three ESs analysed.

Notwithstanding the influence of historic and modern processes and procedures, the current distribution of ESs is relevant for understanding how to manage ecosystems in an equitable manner, for the future and with respect to fair procedures, participation and outcomes. Judgement of what constitutes a fair distribution of ESs is dependent on the ethical stance taken, whether utilitarian, egalitarian or contractarian. In a recent review Lehmann et al. (2018) reflect on how these normative positions underline decisions of ES trade-offs and determines what constitutes a fair distribution of ES benefits.

Under utilitarianism, ESs would ideally be maximised for the greatest number of people. The concern is to maximise ES benefits, so the decision is dependent on location and ES of concern, rather than equality considerations. As outlined in Chapter 3, this is likely to inform an economic based argument for a particular intervention. Population count or density is therefore also often used to model ES demand (Wolff, 2015). On this basis sensitivity analysis, which conceived demand weighted to population density was undertaken (since population was not accounted for within initial calculation of demand). For the three ESs assessed in all regions there is significant correlation between population weighted demand and IMD, whereby demand decreases with decreasing deprivation. The resulting change to the social distribution of net ES index is variable. Therefore in some instances, areas with more people who have greater need for a service but lower supply are also the most deprived (air pollutant removal ES - Northampton, South Pennines; surface water runoff reduction – Northampton, Leeds; recreation – Leeds, South Pennines), but others are less deprived (air pollutant removal – Leeds, surface water runoff reduction – South Pennines, recreation - Northampton).

The consequence of this is that under utilitarianism, action to increase net ES may be directed to areas which are more or less deprived. This does depend on how 'greatest benefit' is defined since greater potential health and social benefits may be gained from increasing ESs for lower socio-economic groups who tend to be more vulnerable and have a lower baseline wellbeing (e.g. for recreation, Wilkerson et al., 2018). Greater vulnerability - the susceptibility to be harmed by and the ability to cope with environmental stresses – is a function of multiple factors, including socio-economic status (Cutter & Finch, 2008). Mechanisms which lead to an increased vulnerability of lower socio-economic groups can be illustrated in the context of recovery from flooding. They include a lower financial capacity to aid recovery, fewer opportunities for alternative employment or housing, less

access to assistance due to power dynamics, social relations and structural organisation and a lower ability to have prepared and mitigated against potential impacts prior to the event (Rufat et al., 2015). Post-flood impacts on health and wellbeing are wide ranging and may be physical but are often psychological in the form of anxiety and stress in response to the event and ability to recover. Ultimately, these impacts on health and wellbeing are likely to be greater for those with higher social vulnerability (Tapsell et al., 2002). On this basis it may therefore be argued that under utilitarianism the greatest benefit is achieved through focussing on increasing the ES benefits to the most deprived communities since this will improve resilience to environmental stresses.

Under utilitarianism compensation mechanisms such as payments for ecosystem services may be used to ensure delivery of the ESs to benefit the most people, whilst adjusting for the potential loss to a minority. Commonly this would involve awarding payment from beneficiaries to stewards of land for production of services, to compensate for loss of earnings from the land from other types of production (Sikor, 2013). To implement these schemes identification of a suitable 'buyer' of services is required (DEFRA, 2016). In the case of directing to services to poorer areas, depending on the ES and local situation, buyers may be difficult to find and financing projects may rely upon Local Authorities or other State actors.

An egalitarian justice perspective embraces the most literal concept of an equal distribution. Since both ES supply and demand considered together reveal the benefits important for health and wellbeing, an egalitarian perspective would advocate an equal distribution of both across all levels of deprivation. Although it should be recalled that in analysis such as this, which uses relative values, a net ES index of zero does not mean that supply fulfils demand. For non-rival ES, including air pollutant removal, surface water runoff reduction and recreation (up to a threshold), demand under this perspective should not be shaped by population counts but is equal regardless of where that person resides. Thus the initial concepts of demand used in this study, which are risk based for the regulation services or assumed uniform for recreation, follow an egalitarian approach.

On this basis, the distribution of ESs in Leeds may be considered the least fair of all case studies as the greatest difference between the least and most deprived quintiles occurs. This would suggest that land planning and management in Leeds should have a strong focus on addressing inequalities. More widely, in national policy, priorities for addressing

inequalities on an individual service basis are more difficult to establish when jointly considering supply and demand, and the wider ecological implications, compared to considering the distribution of a hazard or benefit alone. Regarding ES supply, no single ES is found to be unequally distributed across every study region. However since demand for air pollutant removal is unequally distributed in each case study region, indications are that this poses the greatest concern from an egalitarian perspective despite the supply of this service being equally distributed.

Under a social contractarianism conception of justice a fair distribution would be based on an enforceable minimum standard applied to all. Statutory requirement for protection of natural capital tends to focus on conservation of particular habitats and species (Wildlife & Countryside Act 1981) or landscapes and natural beauty (National Parks and Access to the Countryside Act 1949). Although not designed with respect to ESs, these offer some protection of natural capital, and hence will have an impact on ES benefits and their social distribution. However, with respect to ESs, minimum standards may be impractical given challenges of measurement and enforcement, they may also not be appropriate.

There are minimum standards in the UK relevant to the specific ESs evaluated in the research. With respect to the regulation services, the statutory focus of the relevant Acts is upon the reduction of the hazard; there is, nor can there be, no minimum requirement for the degree to which natural capital is used to contribute to a reduction in a hazard since this is dependent on a wide range of local factors and may not always be the most appropriate mechanism. For instance, mandatory standards exist for air quality, with maximum ambient pollutant concentration levels set under the Clean Air Act (2010), and whilst natural capital can contribute towards improved air quality, the principle means for meeting these standards remains through management of emissions (DEFRA, 2019). As another example, the Flood and Water Management Act 2010 is relevant to SWRR ES, in this case sustainable strategies which promote the use of natural capital and its services are a key element of effective flood management required by the Act, but these strategies need to be locally appropriate.

With respect to recreation, guidance on appropriate minimum standards related to proximity and some physical aspects are relatively simple, but minimum standards for quality or facilities given the possible range of functions and user perceptions are not feasible (Wolch et al., 2014). Access standards (ANGSt) formed the basis for defining the

ES supply in this analysis (as detailed in Chapter 6 section 6.2.1), but these do not have statutory basis. As Ekkel and De Vries (2016) highlight, minimum greenspace standards would ideally be developed on the basis of what is sufficient for health and wellbeing, but despite significant research regarding the mechanisms between greenspace availability and health (James et al., 2015), these remain undefined. Statutory requirements on greenspace may also be difficult to meet retrospectively in dense built-up areas, and whilst there are many examples of creative solutions for greening dense cities, for example the extensive planting of vegetation on buildings in Singapore (Yok Tan et al., 2013), open spaces are still needed to fulfil a range of recreational activities. Minimum standards for cultural services may also be considered inappropriate, potentially negating the diverse social and cultural needs and place-specificity (Wilkerson et al., 2018). Overall a statutory perspective which may be considered a suitable approach to ensuring distribution of an environmental hazard, is more complex and less suited for ensuring fair management of natural capital assets and an equitable distribution of the ESs that arise from it.

In summary, evaluation of the fairness of ES distribution is especially complex given the direct and indirect contributions of ESs to human health and wellbeing, the synergies and trade-offs between ESs which could potentially arise from the same natural capital asset, and the importance of natural capital assets beyond their services contributing to human health and wellbeing. Further difficulties arise through the interlinkages between ES supply and demand thus adding a layer of complexity to the traditional concerns of environmental justice with hazards. For example, is there inequality where deprived areas are characterised by lower supply coupled with lower demand, or where supply is higher but demand is also higher? Or is it the absolute supply and demand values which would be important in these cases? Does the interpretation of fairness also depend on the ES, and whether for example more deprived populations are also more vulnerable? Overall, there is no single conclusion regarding what constitutes a fair distribution of ESs. However, empirical evidence of the social distribution of ESs, combined with knowledge regarding the different philosophical interpretations of fairness, can facilitate more explicit consideration of how inequalities in ESs can be and should be accounted for in decision making (Lehmann et al., 2018).

8.4 Integrating ESs and distributional justice in policy and practice

8.4.1 National policy and guidance

The social distribution of ESs can potentially be modified via numerous mechanisms at national and local level including legislation, incentives, voluntary action and changing technologies and attitudes. Frameworks for management of ecosystems and/or their services are set out through international and national policies. Increasingly within these policies is a growing focus on interconnectedness and the need to recognise the whole socio-ecological system and several address the importance of equity.

Internationally, this is exemplified by the Ecosystem Approach set out by Biodiversity 2020, the UN's SDGs and the catchment based approach set out by the WFD. Notably, the UK, as part of the WHO European Region, is committed to the Ostrava Declaration and thus to “consider equity, social inclusion and gender equality in our policies on the environment and health, also with respect to access to natural resources and to the benefits of ecosystems” (UNECE, 2017 p.2). For England, DEFRA's 25 Year Environment Plan (25YEP) (HMG, 2018), informs the strategic approach and focus for managing England's natural environment. Centred on the concept of natural capital it is of particular relevance to how these issues of distributive (and other forms) of justice acknowledged within international policies are integrated within current approaches to managing the natural environment.

A key focus of the 25YEP is the principle of ‘environmental net gain’; whilst this aim in itself has merit its focus neglects distributional issues. This can be exemplified by the UK Government's Equality Impact Assessment of a revised National Planning Policy Framework (NPPF) which is the basis for England's local planning and development. Within the NPPF it is clarified that improvements to the natural environment are not expected to have differential impacts, that “protecting and enhancing the natural environment should benefit all groups” (MHCLG, 2018 p.19). Socio-economic status is not a protected characteristic in England and therefore not required to be considered within this assessment, nonetheless the perception is that improvements in natural capital are not a distributional justice concern. This contrasts with knowledge, for example, that improvements in air quality between 2001 and 2011 were beneficial to all but that it was

poorer communities who benefitted least (Mitchell et al., 2015) and with acknowledgements made within the Ostrava Declaration.

Nevertheless, in addition to the principle of net gain, the 25 YEP further states “we want to ensure an equal distribution of environmental benefits, resources and opportunities” (HMG, 2018 p.16). One of the ten 25YEP goals incorporates issues of justice “Enhanced beauty, heritage & *engagement* with the natural environment” (HMG, 2018 p.10). Under this goal, one of the indicators developed to measure progress is “Engagement with the natural environment”. This is intended to establish whether people from a full range of socio-economic and demographic groups are spending time in natural environments (HMG, 2019) and thus addresses the importance of considering inequalities with respect to cultural ES.

However, there are additional 65 indicators developed to measure progress against all goals within the 25 YEP (HMG, 2019). None of these indicators address inequalities in the distribution of environmental hazards or in ESs which people benefit from passively. Such inequalities could be important for other 25YEP goals such as the sustainable use of resources and increased resilience to natural hazards. Although with respect to reducing risks from natural hazards, the 25YEP does incorporate commitments, as originally committed to through the Aarhus Convention, to access for everyone to information on the risks posed by flooding and coastal erosion. The 25YEP also echoes commitment to provide clean air in line with legally binding targets (HMG, 2018) but there is a lack of acknowledgement regarding the distributional issues which have been clearly demonstrated with respect to air quality.

Overall, there is a policy focus upon net benefits but some recognition of issues of inequality in access to environmental benefits. With respect to the 25YEP this is focused upon culturally derived benefits, with emphasis on addressing issues of engagement with nature through community involvement and in urban areas. Notwithstanding the importance of such engagement, this analysis demonstrated that, at least with respect to the regions and ES analysed, access to nature is not solely an urban concern (note for example that urban deprived areas in Northampton had greater access to public greenspace) and social inequalities in the distribution of regulating ESs are potentially a greater concern than of cultural ESs (acknowledging that analysis of recreation did not incorporate social preferences nor quality of greenspaces). Therefore there is a risk that a

wider scope of distributional concerns relevant to multiple health and wellbeing and ecological outcomes are being neglected in policy. This is perhaps to be expected given a well-developed research base which examines unequal access to greenspaces, but which similarly does not address a wider range of natural capital and ES. Based on insights from the analysis in this research, it is unlikely that there is a universal pattern of inequality with respect to the supply of ESs across England, but inequalities are present.

Regardless of whether there currently exists an unequal social distribution of ESs, measuring change in inequalities could still be considered essential for monitoring equitable management in natural capital and its services and for meeting sustainability goals. For example, it may be desirable to increase ES for more deprived communities or as a minimum not reduce ES in the most deprived communities. Thus it can be argued that it is appropriate to incorporate an environmental inequalities measure(s) with a wider scope than people's engagement with nature within the 25YEP set of indicators. This is consistent with previous recommendations from the Environmental Audit Committee who advised DEFRA to reverse the decision to remove 'environmental equality' from its Sustainable Development Indicators (it was included previously 2007-2009) (EAC, 2012) and more recently with the recommendations arising from the 2017 Ministerial Conference on Environment and Health (which culminated in the Ostrava Declaration; UNECE, 2017).

This thesis has demonstrated an approach for measuring both regulating and cultural ES at different scales, however a cruder means of evaluating changes in environmental inequalities would likely be needed for inclusion in a set of national indicators. Since it has been shown that analysis of inequalities in environmental benefits using simple greenspace coverage metrics does not adequately reflect the social distribution of different ESs, the limitations of any simpler approach would need to be made explicit. Additionally, the scale dependencies of environmental inequalities (Baden et al., 2007) make establishing a single metric difficult.

There are other opportunities to better acknowledge impacts on inequalities in ESs. For example, the 25YEP proposes that progress against the plan is also assessed through continuation of the UKNEA (2011) on an approximate 10 year cycle (HMG, 2019). As an update to previous work, and given the more extensive scale of research involved this would be an ideal opportunity to better incorporate aspects of distributional justice within the ES evaluations. An increased academic research base, which develops a clearer

picture of how ESs are socially distributed across the country, and development of the tools which would facilitate this, would provide a stronger basis for such assessment. Although, within the environment sector there is also general lack of awareness of distributional concerns with respect to a full range of environmental hazards and benefits, and as such, a first step of such research may involve raising awareness of these issues and their importance through communication and collaboration.

With respect to regional and local decision making processes which impact upon the distribution of ESs, there are several mechanisms which could be used to better examine and address social distributional impacts of the benefits generated. This includes through local plans, the use of equity impact assessment alongside other decision making tools and through development of approaches to ES tools and assessments.

With respect to spatial planning in England, responsibility is devolved to Local Authorities and thus is a localised process (MHCLG, 2019). Given the diversity in the social distribution of natural capital and ESs established by this analysis, a local approach is likely to be better suited to land planning which generates equitable outcomes with respect to access to natural capital and ES. This would also be facilitated through further regional assessments of inequalities in ES such as that undertaken in this thesis, in collaboration with local authorities, which would increase awareness, provide empirical evidence of which inequalities are most pertinent in the local region and could utilise local knowledge to inform more nuanced assessments.

It is important to recognise, however, that local planning is carried out in line with the National Planning Policy Framework (NPPF). The NPPF sets a social objective for achieving sustainable development which includes “accessible services and open spaces that reflect current and future needs and support communities’ health, social and cultural well-being” (MHCLG, 2019 p5) and requires provision of sufficient greenspace. Akin to approaches in the 25YEP, this addresses participatory aspects of justice and access to greenspace but neglects the importance of the distribution of a wider range of natural capital and the services this provides. A key aspect of the NPPF with respect to natural capital is it’s requirement for preservation of the ‘green belt’ i.e. open land surrounding towns and cities which limits development unless required given the lack of other possibilities and given certain conditions (MHCLG, 2019). Whilst this preserves rural landscapes, it promotes densification of urban areas and thus has implications for the

distribution of ESs, environmental hazards and social equity (see Echenique et al., 2012 for assessment of different patterns of urban growth upon sustainability outcomes). Thus, there is opportunity for spatial planning to address issues of distributional justice in ES, however there is a lack of a driver in national policy to do so with respect to a range of natural capital and ESs and policy constraints may also preclude the ability of local planning to do so.

For development projects and investments, social and environmental impact assessments present an opportunity to ensure the distributional effects of decisions which change natural capital and which ultimately alter the supply (or demand) of ES are evaluated. Walker et al. (2005) and Walker et al. (2007) review 16 different impact assessments of relevance to environmental justice which may be carried out in the UK, of these three are statutory: Environmental Impact Assessment (EIA), Strategic Environmental Assessment (SEA) and Sustainability Appraisal (SA), but these do not give much attention to distributive issues. Conversely those which give much stronger consideration to distributive issues are not statutory. This includes those related to health (e.g., Health Impact Assessment) and Equality Impact Assessment, which aligning with the Equality Act defines protected characteristics as beliefs, race, age, sexual orientation but not socio-economic status (Walker, 2010). Of these only the Health Impact Assessment forms official policy but given the indirect links between ESs and health outcomes, inclusion of ESs may be difficult. Overall Walker et al. (2005, 2007) and Walker (2010) conclude that there is a 'distributional deficit' in impact appraisals in the UK. Specifically, Walker (2010) observed a lack of systematic evaluation of distributional concerns within key environmental decision making. Nevertheless, these do illustrate that there is guidance, but its use could be enhanced in particular with respect to environmental benefits. Notably, the wealth of methods and tools now available given the growth of interest in ESs, means that mapping distributions of environmental benefits is now more accessible. The Treasury Green Book (HM Treasury, 2018) which provides guidance for publicly funded bodies to evaluate investments in projects also includes guidance on socio-economic distributive impact appraisal. Notably in the 2018 update more emphasis is given to environmental appraisal including impacts on natural capital and ESs. This illustrates that some guidance is available to promote and support the distributional impact assessment aspect of ESs.

8.4.2 Research and practice

Policies provide the stimulus for addressing issues of equitable management of natural capital and ESs, but the responsibilities to implement in practice are devolved to others. This is echoed by analysis at both national and regional scale undertaken for this research. This section examines the scope for better integration of EJ distributional concerns and natural capital and ES assessments within practice and academic research, and uses the regional analysis of inequalities in ESs to inform suggestions.

8.4.2.1. Environmental justice distributional assessments

As highlighted in Chapters 2 and 3, assessments of inequalities in environmental benefits tend to focus on distribution of greenspaces or coverage of vegetation. This thesis has demonstrated that application of the ES framework produces a more nuanced understanding of environmental inequalities through clearer acknowledgement of the flows which deliver ES from their supply areas to benefitting areas. Specifically it was revealed that use of simple greenspace metrics in analysis of inequality does not represent the distribution of different types of ESs in all locations. Although there are benefits of these spaces which may not be conveyed within the ES framework. This echoes findings by Escobedo & Nowak (2009) who found air pollutant removal in Santiago, Chile was greater in low income areas as it was modelled as a function of ambient pollutant concentration, but tree cover itself was more extensive in higher income areas. For the case study regions in this analysis greenspace coverage (as a percent of total area using the GLUD dataset) was found to be lower in more deprived areas for all case study regions. This is consistent with the direction of association between air pollutant removal and the Index of Multiple Deprivation for all case studies, although the strength of association is underestimated using GLUD data for Leeds and Northampton. It is also reflective of the distribution of three ESs assessed in Leeds, but not in Northampton and the South Pennines. This suggests that studies finding inequalities in greenspace coverage may overestimate the inequalities in individual benefits derived from those spaces. This is particular pertinent since many assessments of health inequalities and their linkages to greenspace tend to be based on the proportion of neighbourhood greenspace (e.g. Astell-Burt et al., 2014). Accounting for the different spatial scales of ES flows also helps to address the issue of examining environmental processes within administrative boundaries.

8.4.2.2 Ecosystem Service Assessments

Issues of participation have been the focus of justice concerns within ES discourse (in high-income countries). This thesis has provided an example of how distributional justice can be incorporated within ES assessments and the importance of the philosophical construction of ESs (which drive how they are modelled) for EJ.

For quantitative spatial assessments of ESs across larger scales, justice issues may appear largely disconnected from the quantitative, biophysical methods applied to modelling ESs. Whilst the ethical stance relating to the concept of ESs itself (i.e. valuing nature on the basis of its contribution to humans as opposed to its intrinsic value) is contested, less attention has been given to value based construction of ES measurement in non-economic empirical ES analysis (Lehmann et al., 2018). Ideally, greater consideration should be given to how different conceptualisations of ES are based on normative positions and the consequences of this for fairness and equality. This is of particular relevance to how ES demand is considered.

Increasingly attention has been given to the need to quantify ES demand in relation to supply, however, there is inconsistency in the way in which demand is conceptualised and quantified (Wolff et al., 2015; Harrison et al., 2018), often aligning with the different categories of ESs. As observed in section 3.3.1.2, demand is often conceptualised in terms of the number of people who desire or need an ES. This aligns with a utilitarian perspective, in contrast to assuming an equal demand for each person aligns which is an egalitarian perspective. Other studies have examined demand of regulation services with respect to the amount of ES required to meet particular standards or guidelines (e.g. Baró et al., 2016) which more closely aligns with the social contractarian perspective. In addition, there is a growing literature regarding vulnerabilities and resilience to environmental hazards (Cutter, 2009), and as Fisher et al. (2013) observe there is much scope for vulnerability and resilience to be incorporated within ES demand models. More comprehensive and consistent integration of vulnerabilities within the conceptualisation of ES demand could enhance consideration of justice issues. The sensitivity tests applied in this research demonstrate clear implications for ES justice of adopting these different constructs (i.e. utilitarian versus egalitarian conception).

With respect to explicit assessments of social distributions of ESs, as carried out in this research, those undertaken tend to be limited to a single service, with emphasis on

recreation and other cultural ESs (see Chapter 2 section 2.3). This analysis has demonstrated that greater attention should be given to the importance of all types of natural capital and the social distribution of a range of ESs arising from this. In England, in addition to cultural ESs, regulating ESs were identified as particular direct relevance for health and wellbeing. Unlike cultural ESs, regulating services are passive hence require no active engagement by the user to access, and as they are driven by biophysical processes, the value of these services may be less recognised or understood by the general public (Brown et al., 2012). This is especially likely with ESs provided by vegetation in locations distant from the beneficiaries.

However, as illustrated for Leeds, the distribution of regulating ESs may be of greater concern for inequalities than cultural ESs (recreation), although there are uncertainties associated with this finding. Nevertheless distributional analysis of regulating ESs, as undertaken here, can play an important role complementing participatory methods and stakeholder engagement. ES distribution maps, in particular where spatially dependent correlations are used to identify areas of low ES within highly deprived areas, can be used to identify target areas for stakeholder engagement and are an effective communication and engagement tool (Norton et al. 2016). ES distribution maps could also be integrated within other participatory tools such as Participatory GIS (PGIS) (e.g. PGIS tool developed by Natural England, ADAS and theresearchbox; http://web1.adas.co.uk/pgis_algol/). An alternative approach, could use stakeholder engagement to identify the environmental issues of greatest concern for which assessments of inequalities in ESs could subsequently be carried out in the most appropriate way, including spatial distributive ES analysis.

Practically, the modelling of ESs is supported by a diverse set of tools and guidance (Bagstad et al., 2013) which offer the means to facilitate inequality assessment. ORVAL (Day et al., 2018) is one example of a tool which explicitly disaggregates benefit by social group. In line with existing analysis of inequality of greenspaces, this is focused on cultural ESs only. Sites can be explored through ORVAL, existing or proposed, based on their welfare contribution to each socio-economic group. On the one hand this emphasises the diverse needs of different users and distributional issues, on the other hand it is problematic because it generates values based on existing patterns of usage which are potentially influenced by current provision of accessible spaces.

Other tools which already account for the flows of ESs from natural capital and consequently the spatial distribution of benefits of multiple ESs such as Aries (Villa et al., 2014) offer the greatest potential to facilitate more comprehensive analysis of inequalities. These could for example be extended to enable generation of some global statistics (e.g. boxplots and correlation coefficients) or even local statistics (e.g. maps of local associations) as carried out in this analysis which compare the ESs to social or demographic data. Some tools including Aries and Ecoserv incorporate socio-economic data and population density into estimates of demand for some ESs, thus are already based on the idea that the most deprived and populated areas have the greatest need for ESs. As discussed, this is one of several perspectives and tools that would benefit from enabling demand (or supply) to be adaptable by users. Enabling users to define demand differently would make the implications of a particular perspective explicit. This would also enable flexibility depending on the objectives of a particular intervention.

Other tools are more tailored to identifying areas for maximising ES supply and conserving biodiversity (e.g. Co\$ting Nature, LUCI, Natural Capital Planning Tool). Nonetheless outputs could be used to cross reference against maps which identify areas which supply ESs to the most deprived populations to identify any win-win opportunities. These would also likely be most useful for showing the trade-offs between social and ecological outcomes. Tools currently designed to produce global (not spatially explicit) outputs e.g. I-Tree, are being increasingly adapted to produce spatial outputs (Bottalico et al., 2017) which will ensure that they can be utilised in conjunction with other spatial information such as maps of deprivation.

Overall, then, we see that inequalities would be better accounted for within the ES discourse with greater awareness of these more implicit factors relating to measurement and interpretation, in addition to the more fundamental need to more routinely address distributive impact appraisal in ES analysis. There is also potential for better integration of who the beneficiaries of ESs are, and recognition or even explicit modelling of inequalities within existing ES modelling tools and guidance. This complements existing approaches actively undertaken such as stakeholder engagement and promotes the idea that social distribution of ESs should consider a wider range of ESs and not be restricted to cultural ESs.

8.5 Sustainable decision making: balancing social equity and ecological outcomes

Ultimately, the aim of assessing the social distributions of natural capital and its services is to facilitate their more equitable management. Wilkerson et al. (2018) suggest that whilst this is of some use to enhancing greenspace supply where it may be most needed, community based action is more effective. However, in the case of the social distribution of multiple ESs, little is known regarding how equally they are distributed, and thus in the first instance such assessments can indicate what concerns there may be. For example, mapping of more locations and at different scales may (or may not) reveal more consistent patterns of inequality in certain environments or for particular ES. For example, in their comparison of greenspace availability and distribution across multiple European cities, De Sousa Silva et al. (2018) found patterns according to city location on the continent. This is particularly relevant for equity considerations over larger areas. Walker (2010) further argues that distributional analysis can raise the profile of equity issues within decision making, and as part of an impact assessment can raise attention regarding the implications of a particular intervention and enable mitigation measures to be developed. However, even when it is known that unequal distributions of ESs in a particular location exist, and that these should be addressed, there are multiple socio-economic and ecological implications of planning and management decisions against which such consideration needs to be balanced.

Sustainable interventions in management of natural capital are ideally generated through the production of synergistic social and ecological outcomes (De Groot et al., 2010). One practical example of this may be drawn in relation to the Northampton case study region; the Nene Valley Nature Improvement Area (NIA) (over 41,000ha) to the southeast of Northampton. This is one of 12 NIAs established by the UK Government in their 2011 Natural Environment White Paper with the aim of improving local wildlife, community and economies through reconnecting natural areas and naturalised gravel pits along the River Nene. In particular wildlife in these areas was increasingly disturbed by growing visitor numbers whilst a 2013 assessment revealed a deficiency of recreation opportunities for Northampton and nearby towns where ANGSt standards were not fulfilled.

Evaluation of the Nene Valley NIA (Collingwood Environmental Planning, 2015) revealed ecological successes, including 4km of river restoration and the creation, restoration and

improved ecological connectivity of 148ha of habitat. Overall, through effective community engagement ('Community Panels') cultural and recreational use of the area by different users was facilitated whilst limiting disturbances to wildlife. Interestingly, a visitor's centre, facilities and accessible pathways were developed to encourage recreation in the Nene Wetlands adjacent to a shopping centre development. This provided a focal point for recreation and a familiar environment to encourage those from different cultural and social backgrounds who are more likely to perceive natural environments as less accessible. It should also be noted that improving access to nature is not only considered a benefit to human health and wellbeing but also a conduit for increasing environmentally friendly behaviours (Wilkerson et al., 2018). The spatial distribution of recreation in the Northampton case study region was revealed in this analysis for this thesis to be concentrated in the South, to which the development of the Nene NIA has contributed. However, the NIA lies within 10km of many of the more deprived areas of Northampton and thus is not considered to have adversely impacted upon inequalities in the social distribution of ES in Northampton, especially given the concentration of public greenspaces closer to more deprived areas.

There are however two key challenges to equitable management of natural capital highlighted in the literature. Firstly, synergistic outcomes may not always be possible, requiring trade-offs both between different ESs, and between conserving natural capital and addressing inequalities (Daw et al., 2011; Seppelt et al., 2017). Moreover, some outcomes may be considered by some as synergistic but as trade-offs by others, depending on the perspective they are viewed from. This has been a key debate in the EJ literature where arguments have been developed from both theoretical (Dobson, 2003; Agyeman, 2004) and empirical perspectives (Mitchell et al., 2015) regarding whether social justice and environmental sustainability are compatible objectives.

Trade-offs between equitable outcomes and increases to natural capital extent and quality likely arise from policies which are based on the principal of net gain (of biodiversity or natural capital), e.g. the 25YEP (HMG, 2018). It is likely that development will occur near urban areas where the need is greatest, this may incur local losses in natural capital and its services which, under the principal of net gain, may be 'offset' in a different location. This will generate clear 'winners' and 'losers' with respect to the benefits obtained from natural capital and thus has implications for inequalities. Although, it should be noted that

there may be potential for developments to improve local natural capital (Holt & Rouquette, 2018). Analysis nationally and regionally in this thesis provides further insight into where other potential conflicts may exist. For example, accessible greenspace has been found to be higher in the more deprived districts (e.g. uplands) or within travel distances of more deprived LSOAs. Increasing publically accessible spaces where it is lowest (districts and LSOAs dominated by agriculture) may therefore not be relevant from a distributional justice perspective and may conflict with food provision objectives. Similarly, planting trees to reduce areas of highest flood risk in the South Pennines would primarily benefit wealthier areas and therefore be limited in terms of addressing issues of equity, although additional benefits may also effect the distributions of other ESs.

Another example can illustrate where there may be more subtle conflicts across scales; Recent Government initiatives supporting the aim of the 25YEP plan to plant over 130000 trees, prioritises funding for planting in more socially deprived areas (Forestry Commission, 2019). This may help to address the inequalities in air pollutant removal ES observed for the three case study regions in addition to providing other local benefits to people and nature. However, Morse et al. (2011) suggest that deprivation impacts upon environmental degradation and lower socio-economic status has been determined as a factor in reduced effectiveness of woodland management for carbon sequestration (Soto et al., 2016). If this holds true in England, carbon sequestration could be maximised through planting in less deprived areas. This in turn comes with risks, as Ernstson (2013) highlights that higher land values and development pressure in desirable less deprived areas can be a threat to natural capital. These examples are intended to illustrate that whilst there are synergistic outcomes (e.g. carbon sequestration will occur wherever the tree planting takes place), there remain subtle trade-offs.

Secondly, where there are synergistic outcomes, positive actions can result in unforeseen consequences with ultimately adverse effects socioeconomically or ecologically. Wolch et al. (2014) give a comprehensive account of the issue of green gentrification, using the example of the New York Highline, whereby investment in developing natural capital which provided a locally deprived community with various ESs (recreation, aesthetic etc.), led to very substantial land value uplift with financial benefits flowing to private real estate from the green infrastructure investment made by the local community groups. Wolch et al. (2014) suggest that green gentrification could be avoided if areas are made 'just green

enough'. There is some concern regarding this approach as it ignores the social distribution of the multiple ESs of greenspaces which may also be delivered to distant locations, and may also limit potential increases in biodiversity. One alternative approach would be to increase ESs for all, but implications of such a strategy for inequalities are unclear, and likely unsatisfactory for some dependent on their ethical stance. Practically, this is difficult since management of the natural environment requires investment with associated prioritisation and protection, all of which have distributional outcomes. 'Just green enough' may also be unsatisfactory from an equity perspective if the implication is a less aspirational vision for more deprived areas. More equitable approaches could incorporate involvement of local residents to ensure the creation or management of greenspaces is tailored to their needs (Haase et al., 2017; Wilkerson et al., 2018).

Guidance which recognises these multiple considerations for sustainable decision making are given by the 12 Principles of the ecosystem approach established by the EU' Biodiversity Strategy in accordance with the Convention on Biological Diversity (2004). Although this focuses on addressing participatory justice and not distributional justice. Schröter's (2017) criteria for sustainable outcomes offer an alternative example which do give attention to distributional issues. These are underpinned by concepts of ecological limits, distributional and procedural justice (Schröter, 2017), A minimum requirement for a sustainable approach would hold that inequalities in ESs are not created nor worsened and/or would require ESs use to be within ecological limits (Schröter, 2017), accounting for the many different ESs including provisional and supporting. Thus essentially, a baseline acceptability of socio-economic, distributional and ecological impacts of different management or intervention options needs to be recognised. Overall, information regarding distributional outcomes is a necessary component of sustainable and equitable decision making and actions, but this knowledge needs to be considered alongside the nuanced links between equality, natural capital and sustainability (Haase et al., 2017).

8.6 Methodological limitations and scope for further analysis

The analysis presented in this thesis represents the first which assesses the social distribution of multiple types of natural capital and several ESs at multiple scales - accounting for ES spatial dependencies and for case study regions with markedly different

natural capital profiles. However, there are limitations to the work undertaken, alternative approaches which could be explored and ways in which the work could be developed or extended to generate further knowledge and insight. Evaluation of the main limitations of the approach taken for the national assessment of natural capital distribution were principally covered in Chapter 4, section 4.4, so here the issues of the local level analyses of ESs are reviewed.

With regards to the reliability of ES mapping, there are necessarily assumptions made and uncertainties introduced (Martínez-Harms & Balvanera, 2012), although sensitivity tests have examined the impacts of some of these assumptions and uncertainties. More advanced sensitivity testing techniques such as Monte Carlo analysis and Bayesian approaches could give further insight into sensitivities (see Hamel & Bryant (2017) for a review of sensitivity analysis in ES mapping) including spatially explicit confidence levels of ES estimates.

Model accuracies would likely be improved through use of more complex models such as full hydrological modelling, inclusion of quality indicators of greenspaces (e.g. Hoffiman et al., 2017) or network based computation of travel times (e.g. Comber et al., 2008). Alternative methods available through modelling tools such as ARIES are becoming more accessible and could offer more effective means of modelling ES complexities. That said, the value of carrying out more detailed analysis of ES should be balanced against the subsequent generalisation to census tracts. Establishing the differences between ES supply and demand in absolute terms could provide an insight useful for interpreting how fair a distribution is and in particular for comparisons between areas. To facilitate this, greater coverage of more refined datasets would be needed e.g. higher resolution ambient air pollutant concentrations beyond city boundaries or spatially explicit tree structure data. One of the advantages of the data used in this analysis is it is free to access for those working in academic and public funded organisations (many are free to all), with predominantly national coverage and thus increases opportunity for replication elsewhere. For more advanced modelling, the increasingly free-to-access availability and coverage of remotely sensed data such as LiDAR could be utilised (e.g. Bottalico et al., 2017). Non-traditional data sources such as social media offer alternative means of modelling cultural ESs (Richards & Friess, 2015) but need to ensure fair representation of different social, cultural and demographic groups

An alternative perspective for those seeking to evaluate inequalities in ESs given limited resources and time, would be that the methods used in this research for modelling ESs need to be simplified or made more efficient (e.g. Burkhard et al., 2014). One example is that ES supply maps differ for each service, whilst often ES analysis utilise a single land cover map (e.g. Baró et al., 2016). A tailored approach was taken given the variable importance of different aspects of land cover and natural capital for each service. For example recreation relied upon accessibility assessment, air pollutant removal upon an accurate representation of tree canopy cover and surface runoff reduction upon permeability of land covers and soil types. This provides more reliable results but is less efficient. Other datasets more recently made available e.g. OS Greenspace map (www.ordnancesurvey.co.uk), Natural England's natural capital maps (<https://eip.ceh.ac.uk/naturalengland-ncmaps>) offer newer means of quickly obtaining input data required.

With regards to assessments of inequalities, the main limitations of this analysis relate to a lack of direct linkage to health inequalities and a lack of temporal component. Development of the analysis to address these limitations could provide valuable information for effective policy and planning. Comparisons to health metrics could better tease apart some linkages (e.g. Pearce et al., 2010), between the social distribution of ESs and health inequalities, although determining actual impacts requires epidemiological studies which tend to be complex and costly (Mitchell & Walker, 2007). This analysis is a snapshot relating to the most recent data available at the time, however establishing the change in distribution over time can provide further insight into how ESs and health inequalities are related, and what may underlie observed associations. This can also aid evaluations of the fairness of distributions and policy successes and failures.

Although analysis was carried out at different scales, evaluating the social distribution of ESs relied upon aggregation of data to LSOAs. This suffers from the limitations commonly faced within distributional assessments, such as the modifiable areal unit problem and ecological fallacy as discussed in Chapter 3, section 3.3.2. The distributions established cannot be assumed to be reflective of distributions at different scales, nor for all locations and people within the LSOAs. Further work could assess inequalities using different spatial units, for example through the use of interpolation to generate socio-economic data at finer spatial resolutions (Schüle et al., 2017). Critically the boundary of case study regions can have a notable impact upon whether inequality is detected (Baden et al., 2007). In this

analysis, it was determined that going beyond the urban boundaries would provide insight into rural-urban gradients, thus providing new insight to existing studies of inequalities in greenspace coverage which are often limited to urban boundaries, and testing the assumption that ESs are lower in urban areas. Modification to the boundaries and analysis of different areas is needed to better make generalisations regarding inequalities beyond the specific boundaries selected for this analysis. The case studies were selected to encompass a range of natural and social environments, however, there were few consistent patterns revealed which could reasonably be assumed as reflective of the distributions more widely across England and the UK. Inequality analysis across other areas in England and in other countries is therefore necessary to develop a more comprehensive understanding of how ESs are socially distributed within high income countries. There also remain knowledge gaps regarding the social distributions of other ESs, which could be developed where appropriate (e.g. noise reduction, local climate regulation).

Currently, the outputs offer insights into inequalities in ESs for the particular case study regions. However as static maps and outputs these can be difficult to compare, whilst it cannot be assumed that users have the technical skills to examine maps within GIS software. For those delivering projects which manage natural capital and stakeholder engagements, a web-based tool to explore data could be useful. This could be particularly effective in making the links between ES supply and benefitting areas explicit and as a decision making tool for exploring trade-offs and synergies.

8.7 Summary

This chapter commenced with examination of results from analysis of the social distribution of ESs within the context of their respective case study regions building on a national distributive analysis of natural capital. It was determined that inequalities in ESs are not pervasive for all ESs across all areas, but are widespread in Leeds and are common to all case areas for the air pollutant removal ES.

Various initiatives and strategies, some though not all of which have been detailed above, will likely change such ES distributions. This highlights that, regardless of whether any

unequal distributions observed are fair, there is a need to address inequalities, a need that is increasingly evident within government policy and associated frameworks.

Nevertheless, there is also need and opportunity for closer connection between discourses of natural capital and distributional justice, particularly within policy and practice and in ES appraisal (i.e. outside of purely academic discourse). Increased awareness and knowledge of the social distribution of ESs is an important first stage, but in seeking to address inequalities, it must be recognised that the decision making process is complex. Balancing ecological and socio-economic needs is a cornerstone of sustainable decision-making, however win-win decisions are not always attainable.

Having concluded this chapter with some caveats and considerations regarding the limitations of the research (and how some of these may be addressed), the next and final chapter seeks to conclude the thesis by extracting the key findings and conclusions to be drawn from this discussion and the wider research, reflecting on the original research aims and secondary research questions.

Chapter 9 Conclusion

This thesis has provided a unique insight into social inequalities in natural capital and ecosystem service (ES) distributions. To the author's knowledge this is the first example of a multiscale spatial analysis in a high-income country which assesses how a range of natural capital assets and the services they provide are distributed across socio-economic groups. Findings indicate that with respect to a comprehensive range of natural capital nationally across England, and with respect to two regulating and one cultural ES across three case study regions, there exists some inequalities but these are dependent on context and model assumptions and are not unequivocal across all ESs and types of natural capital.

This final chapter firstly reflects upon the context of this research and provides a brief review of the thesis (9.1), before (section 9.2) identifying key contributions and implications from the analysis and discussion, responding to the research questions posed in Chapter 2 section 2.5. These provide a more nuanced response to the overarching research aim, to "*Determine the social distribution of natural capital and ecosystem services in England, United Kingdom*" (p.4). Section 9.3 extracts the principal contributions of the research to advancing knowledge relating to the environmental justice and ESs fields, then section 9.4 identifies implications for policy and practice. Section 9.5 then outlines opportunities to build upon the thesis in future work before the final section (9.6) presents a closing summary.

9.1 Research context and overview

The dominance of human activity is threatening the world's ecosystems upon which human health and wellbeing also depends (IPBES, 2019). This challenge has resulted in the proliferation of research exploring the human-environment relationship, including that which reframes the natural environment as natural capital and ecosystem goods and services (ESs) (Seppelt et al., 2011) and which explores how access to these benefits is shared across society (Agyeman et al., 2016). This thesis began by exploring the array of ESs and environmental justice (EJ) research, both of which consider the impact of changes in the natural environment upon human health and wellbeing. ES discourse is

primarily focused upon managing ecosystems in a way which conserves and enhances them and the services they provide to people. EJ discourse is focused upon disparities for lower socio-economic, ethnic minority and other marginal demographic groups with respect to their natural environment and meaningful participation in the processes which shape it. However, for sustainable outcomes from land planning and management, both discourses are key considerations, as exemplified by principles set by the internationally agreed UN Sustainable Development Goals (UN, 2015) and the Convention on Biological Diversity (CBD, 2010).

There are commonalities in ES and EJ research and a demonstrable need for their closer alignment, in particular with respect to empirical analysis and within high-income countries, where few spatial analyses address both concepts together. This gap in knowledge was confirmed through Chapter 2 and 3's review of the theoretical development of concepts, implementation and challenges in the spatial analysis of ESs and of inequalities in environmental hazards and benefits. Specifically it is established that ES analysis has concentrated on assessing ES supply, thus neglecting ES demand and the beneficiaries of ESs (Villa et al., 2014; Bennett et al., 2015). EJ distributional analysis of environmental benefits has been dominated by studies examining differences in urban greenspace coverage and access to recreation. Overall, this EJ research has indicated that there is lower urban greenspace extent and quality in poorer areas but that this is not a consistent finding across all studies. Thus it is asserted that knowledge of inequalities in environmental benefits could be developed using the clarity and spatial explicitness of the ES framework. This holds true for country specific research in England, which although quick to adopt the concepts and develop an extensive knowledge base in both disciplines, does not include evaluations of ES inequalities.

Conceptually, analysis of the social distribution of ESs can be considered as an extension to existing natural capital and ES frameworks which conceptualise the generation of 'ES benefits' from natural capital through socio-economic and ecological processes. By conveying ES benefits as 'disaggregated', the importance of the distribution of ESs can be brought to the fore.

Chapter 3 developed the understanding of the conceptual approach and how to operationalise this with respect to the objectives. In practice, spatial assessments of environmental inequalities and of ES face multiple challenges, as discussed in Chapters 2

and 3. Many of these challenges are common to both fields, including data availability, appropriate scales of analysis, oversimplification of socio-economic and ecological processes and related uncertainties. The research objectives (Chapter 2 section 2.5; Box 1), were established with the aim of undertaking analysis which best addresses these concerns. Thus the spatial analysis in this thesis is conducted for different areas and resolution, incorporates the spatial flows of ESs from natural capital where possible, and is tested for sensitivity to several model assumptions and uncertainties.

The analysis chapters (4-7) addressed the three research objectives (Box 1) with Chapter 4 dedicated to Objective 1 and Chapters 5-7 addressing objectives 2 and 3 with a single ES explored in each chapter which also conducts sensitivity testing. The discussion chapter (8) reviewed these results in greater detail before widening the discussion to link to theoretical conceptualisations of how social distributions may be interpreted, and the wider implications for integrating EJ distributional concerns and ESs into policy and practice, and sustainable decision making.

Objective 1: To assess the social distribution of natural capital across England.

Objective 2: To assess the social distribution of multiple ecosystem services for case study regions in England.

Objective 3: To evaluate the robustness of results to model assumptions and uncertainties.

Box 1 Research objectives

In addressing these objectives, a series of secondary research questions were identified (Chapter 2, section 2.5). In answering these questions, current knowledge of ESs and EJ in England and perspectives for research and policy which examine ES inequalities are developed. The next section addresses these questions, drawing upon the analysis and discussions presented across Chapters 3-8.

9.2 Summary of findings

9.2.1 Which ESs are appropriate for assessment from the perspective of equitable management of natural capital in England?

For England three ESs were selected for analysis - recreation, air pollutant removal and surface water runoff reduction. These ESs were identified as the most appropriate for analysis of social distributions given their likely contribution towards addressing key health challenges in the English context and also more widely. For example, non-communicable diseases and air pollution are two of the global threats in 2019 as identified by WHO (2019) (see Chapter 3, section 3.2.2).

It is also possible to map these ES. Primarily the feasibility of mapping relates to whether services have direct, spatial links between natural capital and beneficiaries. Three services were selected as an appropriate number to assess for three case study regions; together they provide insight into inequalities in both cultural and regulating (and therefore active and passive) ESs, into services which are important at different scales and which are driven by different characteristics of natural capital. The different social distributions revealed for each ES across the case studies demonstrates this selection was important for developing a wide-ranging knowledge of inequalities in environmental benefits. From this, broad assumptions can be made with regards to services at similar scales and based upon similar modelling approaches. For example noise regulation is likely to have a similar distribution to air pollutant removal. However, there remains a challenge that only a subset of services are examined in this thesis and there is a need to understand how issues of equity can be incorporated into management of natural capital that produces numerous services, with indirect flows to beneficiaries, and which is of value irrespective of these ESs. There are other services, which may be appropriate to map in particular areas (e.g. urban or coastal areas); in a high-income country context these may include noise regulation, local climate regulation, education and regulation of mass movement (e.g. of soils).

9.2.2 How can the different scales at which ESs are delivered from natural capital to beneficiaries be accounted for within analysis of their inequalities?

Scale dependency is a critical characteristic of ESs, and even more so when seeking to understand the distribution of services; this issue was introduced in Chapter 3, section 3.1.3. The analysis (Chapters 5, 6 & 7) demonstrated that for large areas (>600 km²) it is feasible to account for the diverse flows of ESs individual to small area administrative units (LSOAs) within the study areas. To achieve this each administrative unit is identified as a location of beneficiaries, the flows individual to each service are then mapped for each LSOA in order to identify their, often unique and distant, ES source areas. This identifies the areas of natural capital where changes are pertinent for services within the case study region. This approach accounts for greater spatial complexities than existing analysis of inequalities in greenspace distribution. Different correlations with deprivation are revealed for the ESs and for greenspace coverage within the LSOAs, demonstrating the importance of accounting for the natural capital beyond the immediate case study boundaries for reliably assessing inequalities. However, as previously noted, this approach is only appropriate to those services with direct spatial linkages.

Assessment of social distributions of ESs nationally is approximated in this thesis through analysis of natural capital; the main reason for this is consideration of analytical practicality including resources and time. If these can be addressed, the same methods used for case studies could be applied nationally, although these would need to consider natural capital beyond the country border which may be impacted by data availability.

In addition there is potential to develop and combine the spatially explicit outputs from this thesis. Specifically, the source areas of ES for areas identified as highly deprived with low ES (i.e. developing outputs from Local Indicators of Spatial Association) can be identified using the information generated during the analysis. Overlapping these source areas will identify the areas of natural capital important for addressing inequalities across the three ESs.

9.2.3 Are inequalities in natural capital and ESs present at different scales?

Scale is a key factor in this research. Observed inequalities in environmental hazards and benefits are known to be scale sensitive, i.e. they may be revealed at some scales and not at others (Baden, 2007). However Baden et al. (2007) argue that even if inequalities are only established at a single scale, these findings remain valid evidence of the existence of inequality.

For this research, a national overview of inequalities in natural capital was provided. This national analysis, implemented using the Local Authority Districts as the spatial unit, benefited from use of multiple indicators of natural capital quality and extent to provide a comprehensive insight. It was established that whilst the most deprived areas tended to have the lowest extent and quality of natural capital, this did not hold true for all types of natural capital. Moreover, deprived areas were also found to have the highest extents and quality of natural capital. This suggested that there is not a consistent social gradient in natural capital extent and quality nationally at a coarse scale.

A similar finding has been echoed through more detailed analysis of selected ESs in case study regions. Thus at national and regional scales there is evidence of inequalities in environmental benefits, but that this is not universal for all areas or across all types of natural capital or ESs. However, the finding of some inequalities remains pertinent, in particular considering Baden's (2007) assertion. This again suggests that a greater knowledge base is required. It further suggests that policies and actions need to be scale appropriate, i.e. a 'one size fits all' policy is not appropriate for national policy which instead should require tailored approaches at regional and other sub-national scales.

9.2.4 Are inequalities consistent across different ESs?

Discussion in Chapter 8, section 8.2, examines the differences and commonalities in the distribution of each ES across all three case study regions. Consistency was found across the three areas only for air pollutant removal. This emphasises the importance of examining inequalities in multiple locations and in different contexts for building a comprehensive picture of inequalities in ESs nationally and internationally. Moreover, contextualising the case studies, as achieved in this analysis through characterisation of

districts based on indicators of natural capital and deprivation, can provide some indication of what is driving particular patterns of social distribution in ESs in different areas. For example, deprivation is concentrated within the built up areas in Leeds and overall there is higher availability of natural capital at, and beyond Leeds district boundaries, compared to Northampton; this therefore generates greater inequality in ESs in Leeds. The lack of consistency in the social distribution of these ESs aligns with other findings that in some urban areas access to public greenspaces is higher in more deprived areas, e.g. for Bristol (Jones et al., 2009) and Sheffield (Barbosa et al., 2007; Mears et al., 2019) but is lower for more deprived areas in others (e.g. Bradford, Ferguson et al. 2018).

9.2.5 Is there a rural-urban gradient in inequalities?

As highlighted in Chapter 2 sections 2.2.2 and 2.4, the focus of research internationally and within England has been on examining inequalities in environmental benefits in urban areas. Notwithstanding the importance of this research, it remains an assumption that inequality in environmental benefit is largely an urban concern. The national analysis presented in this thesis (Chapter 4) supports this to an extent showing that overall urban deprived areas tend to have the lowest extent and quality of natural capital, whilst there is high natural capital in more deprived rural areas. There is a wide range of evidence of the health benefits of living in overall greener environments (Twohig-Bennett & Jones, 2018), however, it cannot be assumed that all ESs are higher in rural areas since these may be dependent on accessibility or upon more distant natural capital, as demonstrated by Radford et al. (2018). Analysis in the case study regions shows that for air pollutant reduction there is a rural-urban gradient, although this is to be expected given the higher demand for this ES within urban areas. There is no consistent urban-rural gradient in inequalities in ES for surface water runoff reduction and recreation across the case studies. However, it is notable that when considering access to open greenspaces (excluding public rights of way), recreation opportunities may be greater within urban and more deprived areas compared to rural agricultural land, as established for Northampton and surrounding areas.

9.2.6 Are findings robust to uncertainties and model assumptions?

There has been much criticism of early research of ESs and of environmental inequalities with respect to overlooking uncertainties inherent in their analysis (e.g. Mohai et al., 2009; Eigenbrod et al., 2010). More recent studies show how this issue can be addressed through testing for sensitivity to assumptions and uncertainties (e.g. Schulp et al., 2014; Ferguson et al., 2018). Therefore to ensure findings of this research are robust, sensitivity tests were conducted. With respect to the national analysis of natural capital, testing was carried out to assess sensitivity of results to methodological choices, such as the number of natural capital clusters created. This demonstrated that results were largely robust to the method decisions made.

Regarding analysis of the social distribution of the selected ESs, various sensitivity tests were undertaken, specific to key sources of uncertainty for each ES. It was established that the pattern of inequality demonstrated for air pollutant removal was consistent given variation in the effectiveness assigned to vegetation. However, changes to the social distributions occurred in response to variability in the assumed effectiveness of different land covers in reducing surface water runoff. This was observed with respect to the Northampton (slight change only) and Leeds case studies, but the extensive nature of this testing precluded testing for the South Pennines. Although the social distributions in the first instance were based upon 'best estimates', the sensitivity of results revealed in this testing indicates that findings should be treated with caution, and ideally used alongside additional evidence, which should also address issues of ecological fallacy.

Moreover, considerable changes occurred in response to different distances used in modelling access to recreational greenspace. Therefore sensitivity to different distances at which greenspace is considered 'accessible' has important implications for establishing equitable approaches to greenspace management. For example, shorter travel times which incorporate only the greenspaces within walking distance increase the strength of association observed between deprivation and recreation for the South Pennines and Northampton and Leeds case regions. This suggests that it is important to establish the distances most critical to the communities in these areas based on stakeholder engagement. There also remain further uncertainties inherent within the analysis, related to data sources, generalisations and measures of inequality.

Sensitivity tests were also carried out with respect to the conceptualisation of ES demand, which is considered an approach unique to this research. The many different ways in which demand is conceived are discussed in respect to the methods for modelling each ES (Chapters 5-7). Initial analyses were based on demand that ignores population density; this conceptualisation sees ES demand as non-rivalrous, and the same for every individual. However economically driven ES management decisions likely need to consider the number of people potentially affected. Therefore, by examining sensitivity of social distributions in ESs to population weighted demand, it is possible to determine how this new conceptualisation of ES demand modifies the social distribution of ES. It was established that this change is significant for some ESs in some locations (e.g. increasing recreation access in the South Pennines and Leeds for the more deprived areas also increases access for more people) but not in all instances (for example increasing air pollutant removal in Leeds for the more deprived areas does not coincide with improving it for the most number of people) (see section 8.3). This is critical for considering what are 'fair outcomes' and demonstrates how the conceptualisation of demand can modify our understanding of inequalities in environmental benefits.

9.2.7 Can opportunities for synergistic social and ecological outcomes be identified?

Targeted increases in ESs to more deprived areas which are lacking in particular services, or which have a greater need for them, may facilitate synergistic social and ecological outcomes. This aligns with the original conceptions of ESs in enabling sustainable decision making (Schröter, 2017). Information on where these areas are located and where the corresponding natural capital is which produces the ESs is essential for achieving this. In this analysis, local indicators of spatial association are used to generate output maps which pinpoint areas within each case study whereby there is significant local association between high deprivation and low ES (with respect to each ES assessed individually). Regardless of whether global measures of inequality indicate an unequal social distribution of ESs, such target areas were identified for all services in all case study regions. This demonstrates a practical means for establishing opportunities for synergistic outcomes, in particular once considered alongside the supply areas as defined by flows of ESs (calculated as part of the processing steps within this analysis).

However, as conveyed in discussion (Chapter 8 section 8.5) this information should be used in conjunction with a wider knowledge base which examines the synergies and trade-offs from increasing supply of a particular ES in a particular area against the impacts this has upon other benefits, their social distribution, and whether this is appropriate ecologically and culturally. Moreover, consideration of what fairness in the distribution of ESs constitutes is critical for judging outcomes. Thus whilst this analysis presents information which can facilitate such decision making within the specific case study regions, and illustrates an approach applicable elsewhere, such information forms only a part of the understanding required for delivering sustainable outcomes (Bennet et al., 2015).

9.2.8 Are there opportunities for closer integration of distributive justice concerns and ES assessments within land planning policy and management?

Current considerations of environmental justice embedded within land planning policy and practice are concerned primarily with participatory and procedural forms of justice as recognised by commitments to the Aarhus Convention (UNECE, 1998). Legal protection for equality is available with respect to certain social and demographic characteristics through the UK's Equality Act 2010 and legally binding air quality standards (EU Directive 2008/50/EC and The UK Air Quality Standards Regulation 2010) provide protection for all with respect to this specific hazard. However, as discussed with particular reference to the UK Government's 25 Year Environment Plan and of the National Planning Policy Framework, which set the current policies for managing natural capital in England (Chapter 8 section 8.4.1), distributional inequalities with respect to environmental benefits are less recognised beyond consideration of access to greenspace. Specifically, there is a lack of recognition of potential inequalities in ESs which do not require actively accessing greenspace, such as surface water runoff reduction or air pollutant removal. For example, greater emphasis could be given to interventions which increase provision of air pollutant removal ES in the more deprived areas which currently exceed legally binding air pollutant concentration limits. Thus policies could be developed to encompass a broader scope of concerns with respect to inequalities in ESs, and link these to relevant legislation where it exists.

The thesis highlights that some non-statutory forms of appraisal provide guidance for assessing distributional impacts of investments. The most notable of these with respect to environmental inequalities across socio-economic groups is HMG's Treasury's Green Book (2018). At the local and regional scales, other non-statutory impact assessments such as Equality Impact Assessments offer some guidance with regards to considering issues of inequalities within decision making processes. These do not specifically address differences across socioeconomic groups or with respect to impacts upon environmental benefits, but they could be enhanced to do so.

The development of existing ES assessment tools to facilitate analysis of inequalities in their distribution, and a more developed scholarly research base at all scales from which policy and practice can draw upon, are also critical. This thesis presents a starting point for such research but developments are needed to further establish how pervasive and severe inequalities in ESs are and how these can be assessed with limited resources. This is necessary to raise awareness and enable implementation of assessments of the social distribution of ESs within policy and practice.

9.3 Key contributions to ecosystem services and environmental justice research

This thesis has contributed new knowledge to the ecosystem services and environmental justice disciplines in several ways. In summary, the key contributions include:

- An understanding of how natural capital is socially distributed in England, the first such analysis for England, and the only national analysis conducted for a high income country where population dependency on natural capital is less obvious (direct, local) than for low income countries where natural capital - livelihoods dependency is more immediate.
- New knowledge on the spatial distribution of three key ecosystem services, addressing ES supply, flows and demand, derived from analysis of three English regions (Leeds, the South Pennines and Northampton). Whilst maps of multiple

ESs have previously been produced for Northampton (Rouquette, 2016), these do not account for the spatial flows between ES supply and benefitting areas.

- New knowledge of the social distribution of three ecosystem services in England. This extends previous environmental inequality assessments in England through the application of an ES framework which establishes spatial flows of benefits from both distant and nearby natural capital, and examines these benefits (ES supply) in the context of demand for them (e.g. air pollutant and flood hazards). The analysis also addresses both rural and urban areas, a broader scope than most assessments of inequalities in environmental benefits (i.e. greenspace) which focus upon urban areas. Moreover, no prior analysis exists of the social distribution of ESs addresses the services of air pollutant removal, surface water runoff reduction and recreation, for a high-income country. The thesis develops a method for assessing the social distribution of each of these ESs, which are critical globally. It also demonstrates that scales of ES flows are important for distributional justice.
- The development of spatial analysis techniques applicable to the ES framework and able to generate insights into environmental inequalities. Examining how ES assessments can incorporate an equity dimension, aligns with calls for developing ES research to better account for social impacts, specifically equity.
- An improved understanding of the importance of sensitivity testing in ES distributional analysis. Analysis shows that for some services the uncertainties in proxies can alter the association between the ES and deprivation. Moreover, the study provides empirical evidence of the sensitivity of distributional assessments to assumptions in modelling ESs, specifically some of the ES flows and approach to modelling ESs demand on ESs. The sensitivity to flows is important given the large body of literature which examines inequalities in environmental benefits using metrics of localised greenspace cover.

9.4 Key recommendations for policy and practice

There are several aspects of this research which are relevant to policies and practice of natural capital management - these were highlighted in the preceding section and discussed in detail in Chapter 8. There are however two overarching policy/practice messages which are derived from this thesis. These are the need to:

- Give added emphasis to equity considerations within decision making for natural capital and ES management.
 - This involves further raising awareness of the need to increase access to greenspaces for all, through recognition of the potential impacts of inequalities in the social distribution of passive, regulating services upon health and wellbeing and health inequalities that are generated by a wide range of natural capital.
 - To achieve this in practice requires an increased focus upon beneficiaries - who are the winners and losers of changes in ES flows? This will entail quantifying ES flows, including those from distant natural capital, as well as ES demand.

- Generate greater understanding of what an equitable management of natural capital is, through equity assessment, evaluation and visioning. This implies that explicit consideration is needed of the normative conceptualisations of ES – i.e. what is ‘fair’ in ES management?
 - This involves recognising equity as an important consideration within large scale ES assessments in addition to local, participatory based assessments. Specifically, a clearer understanding of how inequalities may be revealed dependent on how ESs are conceptualised and modelled should be developed. For example, social distributions of recreation may change if different distances to recreation sites are used. Thus if a desired outcome is improved access for all, clarification of what measure is appropriate and why needs to be made.

- Similarly, the conceptualisation of ES demand is based on ethical choices (justice conception subscribed to) which should be considered explicitly to ensure interventions align with the intended and sustainable outcomes.

9.5 Future research developments

There is much scope for closer integration of ESs and EJ research. Doing so will improve understanding of linkages between the two discourses important to the provision of information needed for sustainable policy and decision making. This thesis has explored the links between distributional EJ justice, natural capital and a selection of ESs, and how these can be integrated within spatial assessment. The limitations to this analysis (discussed in Chapter 8, section 8.6) align with existing critiques of environmental inequality and ES assessments, and the discussion also highlighted how future analysis may address such limitations. Drawing on these prior assertions it is possible to reflect more broadly on the potential of further research to advance social justice and environmental sustainability.

Primarily, the evidence base with regards to the social distribution of ESs needs to be developed, in England and internationally. At present, general conclusions cannot be drawn with regards to existence of inequalities in specific ESs, and how pervasive and severe they may be. Establishing this evidence base is important for understanding the potential contribution of ESs to health inequalities. Developing the evidence base will involve further empirical analysis, examining different ESs, different social and demographic sub-groups, and at different scales, incorporating data to evaluate impacts on health and wellbeing and using different methods. Notably, modelling historical changes in the social distributions of ESs and how they respond to future scenarios are important for establishing unpicking causality and judging fairness (Mitchell et al., 2015; Wilkerson et al., 2018); this cannot be achieved via cross sectional inequality analysis alone (see section 8.3 for discussion of what is a fair distribution). Efficient ways to achieve this may involve building on existing natural capital and ES data and analysis, and extending existing ES tools to facilitate distributional analysis. Such developments would also benefit from enabling users to define changes in modelling parameters to test for

sensitivities. Moreover all approaches to analysis should incorporate sensitivity testing to ensure the evidence base is robust. As discussed by Hamel & Bryant (2017) and exemplified in this analysis, sensitivity tests do not need to be complex to be revealing.

Multidisciplinary approaches are needed to improve understanding of how the complex and indirect linkages between natural capital, ESs and beneficiaries can be better accounted for within assessments of the social distributions of ESs. This could better integrate knowledge from other literature, for example vulnerability research (e.g. Boone, 2010). This aligns with recommendations of previous studies, and with the increasing acknowledgement within ES discourse of the need to account for the full socio-ecological system. Multidisciplinary teams of researchers and practitioners can also facilitate the integration of knowledge for increasing impact. For example distributional ES assessments could be used to enhance other forms of knowledge and stakeholder perceptions, consistent with requirements of the Aarhus Convention and Convention on Biological Diversity.

9.6 Concluding statement

The aim of this research was to establish whether inequalities in the distribution of ecosystem services in England exist. Equality is an important element of sustainable decision making which must address both social and ecological outcomes. Addressing inequalities and managing ecosystems in an equitable manner is thus embedded in international initiatives, such as the UN Sustainable Development Goals (UN 2015). The justification for this research is based on the premise that information regarding the social distribution of ecosystem services is necessary to facilitate equitable natural capital management. However, currently there is little research on high-income countries which provides such empirical evidence. This thesis has provided a unique example of an approach to integrating distributional justice concerns and the ecosystem service framework in the context of large scale spatial assessments across multiple areas and scales.

England was selected as the research focus, being a high-income country with strong research and policy interest in both ecosystem services and environmental justice, but no previous study of the social distribution of multiple ESs or natural capital. A broad national

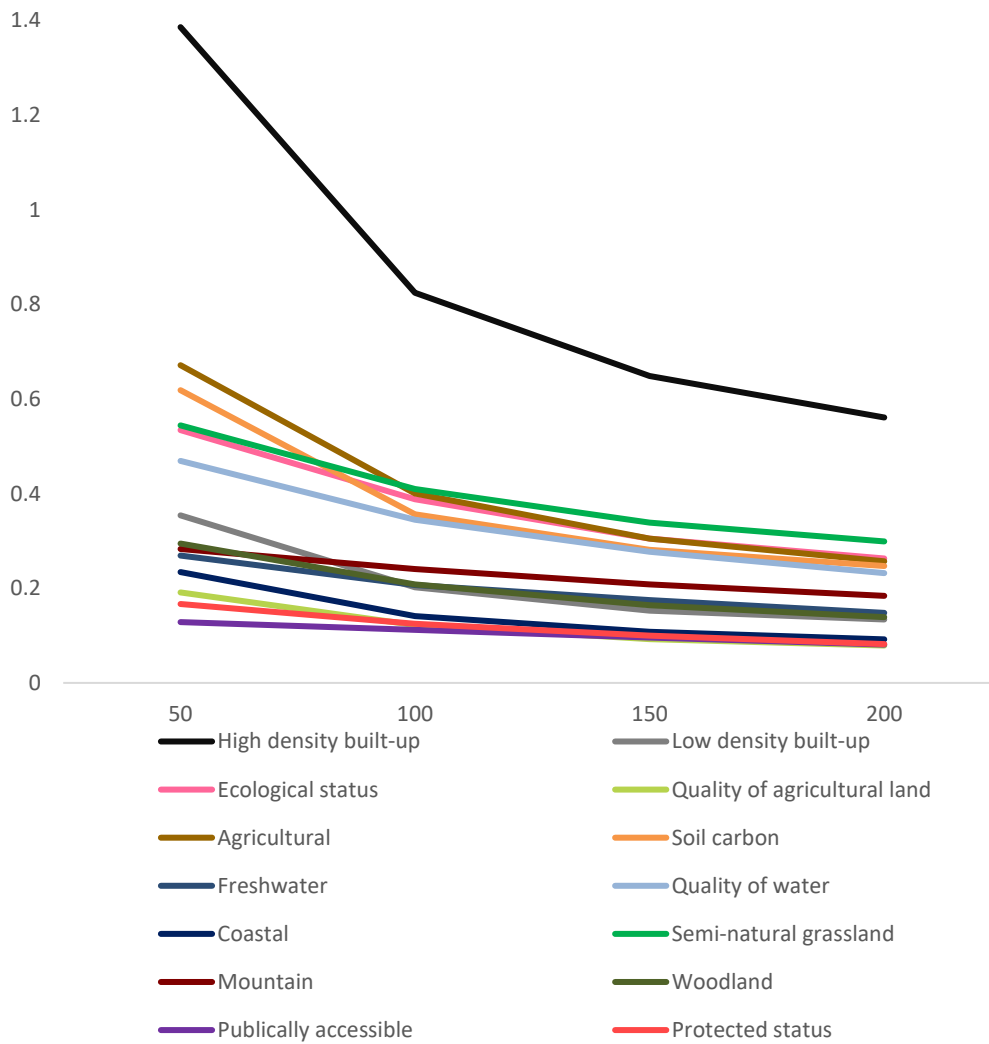
analysis of natural capital and deprivation, and detailed analysis of the social distribution of three ecosystem services in case study regions revealed that some inequality exists. Most notably it was established that inequalities are present across all assessed ecosystem services in the district of Leeds, encompassing a major English city. Inequalities were also consistently found with respect to the removal of air pollutants, and whilst driven by the demand for this ecosystem service - proximity to roads as a pollutant source – the distribution of vegetation was such that it does nothing to reduce these inequalities.

Thus overall, a mixed picture was revealed with the presence and severity of inequalities varying according to location, ecosystem service, natural capital and ES modelling assumptions. This aligns with prior findings on environmental inequalities more widely, which for some hazards and benefits (greenspace) are inconsistent. This demonstrates a need for greater understanding of links between ecosystem services and environmental justice in high income countries, developed via an increased evidence base across all ecosystem services and considering alternative conceptions of justice. Ultimately, whilst improvement in quality and extent of natural capital assets is a critical aim in itself, greater consideration of who is affected by natural capital change is needed.

Appendix A

A.1 Moran's I global statistic

Moran's I calculated with 50, 100, 150 and 200km threshold ($p < 0.001$)

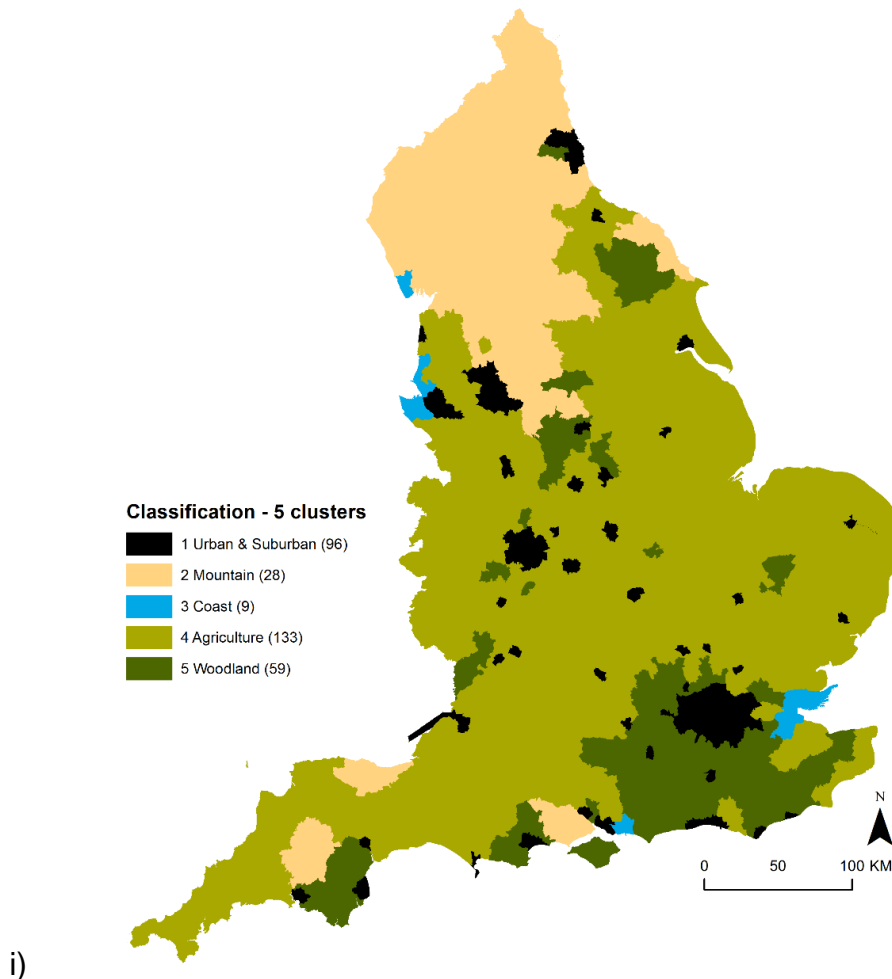


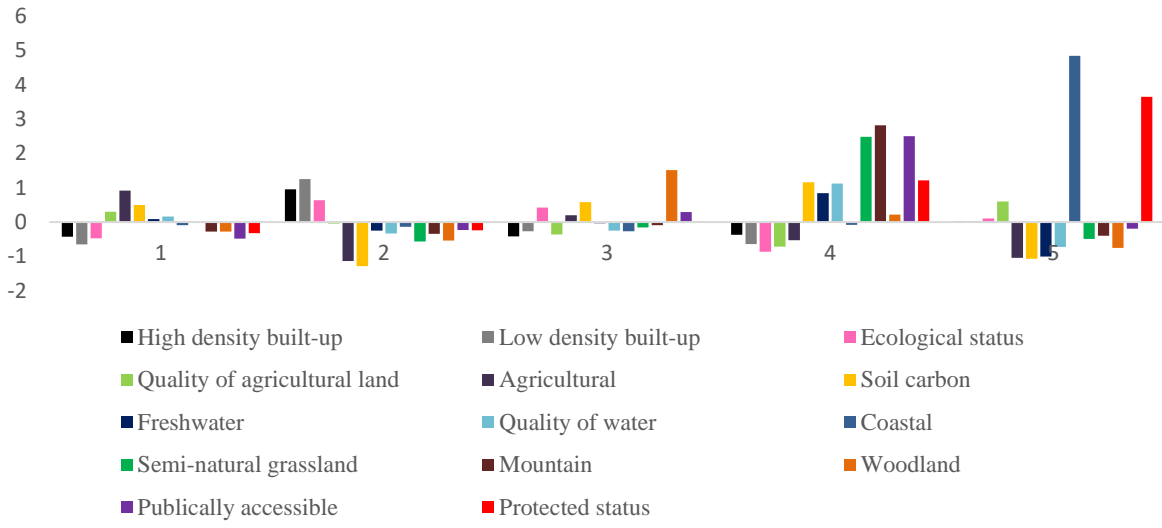
A.2. Sensitivity to number of clusters

Overall, both the natural capital clusters and association between deprivation and natural capital are relatively robust to changes in the number of clusters created.

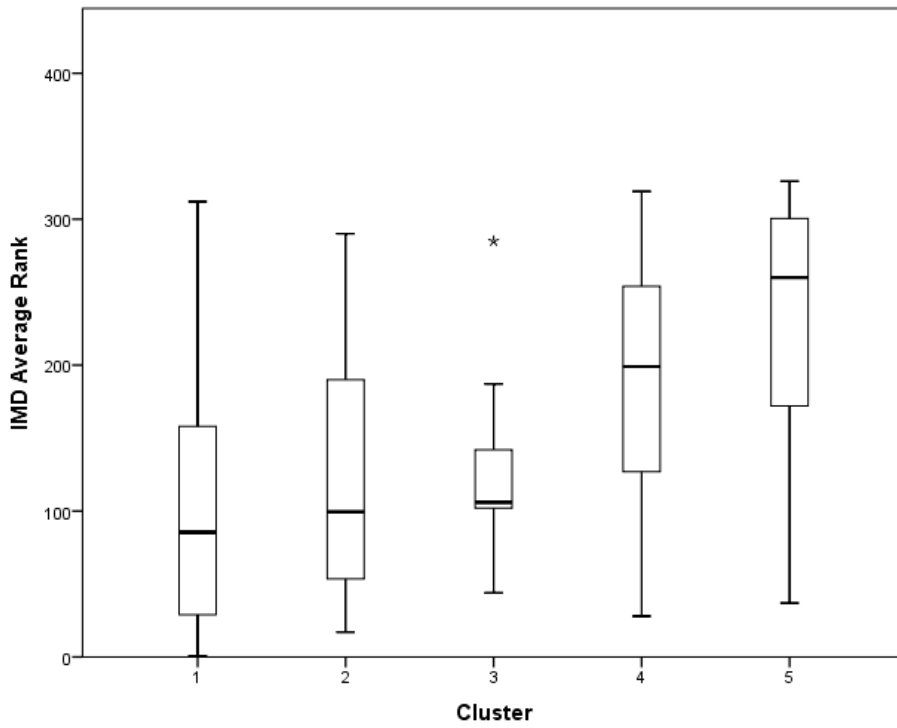
- a. 5-cluster solution i) spatial distribution ii) indicator z-scores for each cluster iii) IMD average ranks of districts in each cluster.

Note that in comparison to the 6-cluster solution there is one cluster for urban areas as opposed to separate urban and suburban clusters and slightly more districts assigned to the woodland cluster and slightly less to the agricultural cluster. Overall a similar spatial pattern emerges. Changes in average deprivation across clusters indicate woodland and agricultural areas are the least deprived and urban/suburban and mountainous areas are the most deprived.





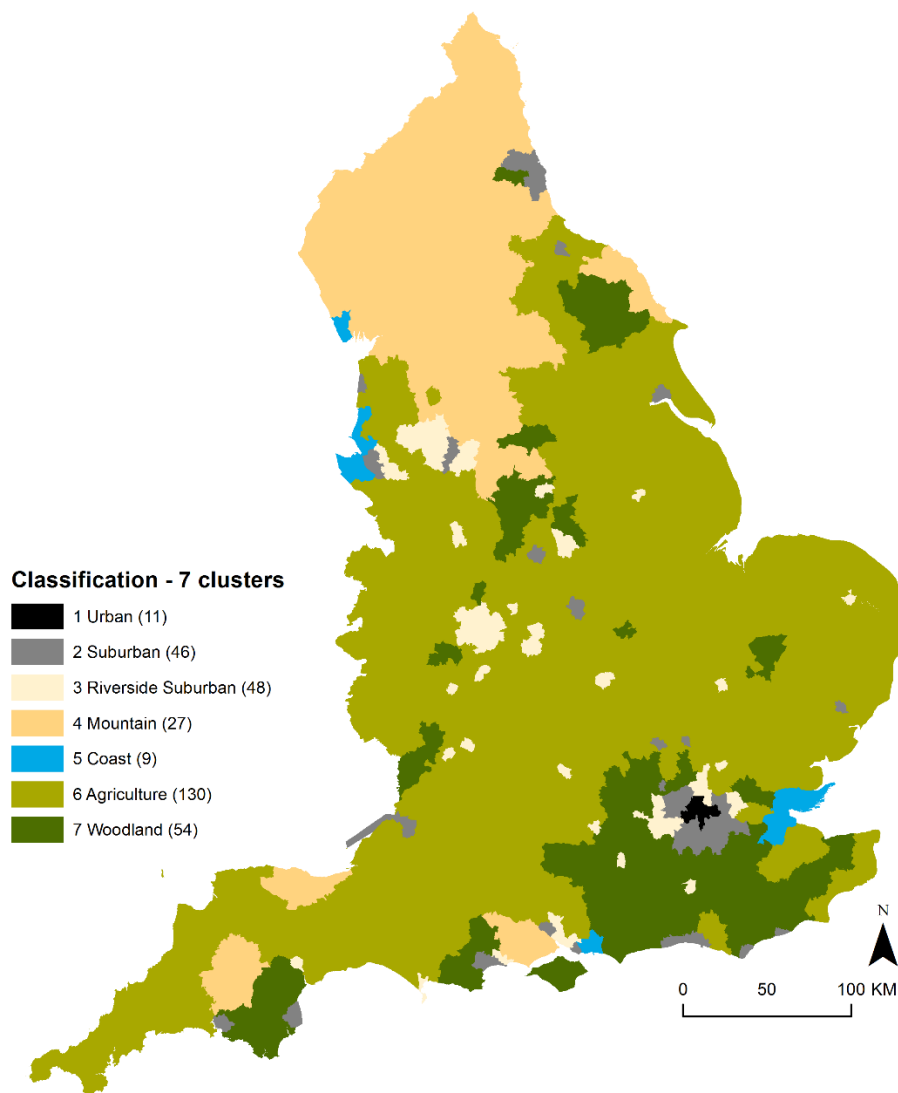
ii)



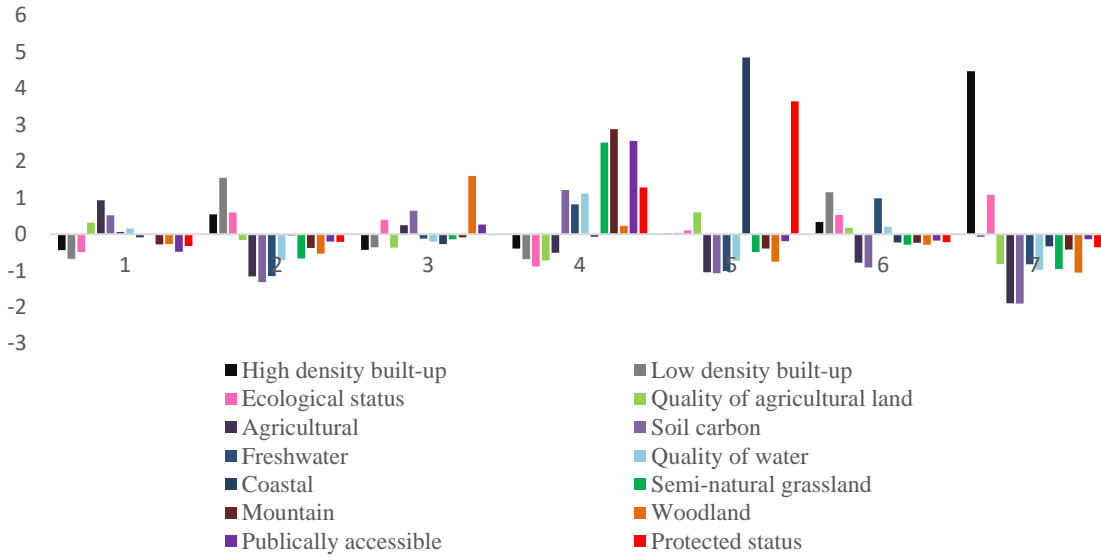
iii)

- b. 7-cluster solution i) spatial distribution ii) indicator z-scores for each cluster iii) IMD average ranks of districts in each cluster.

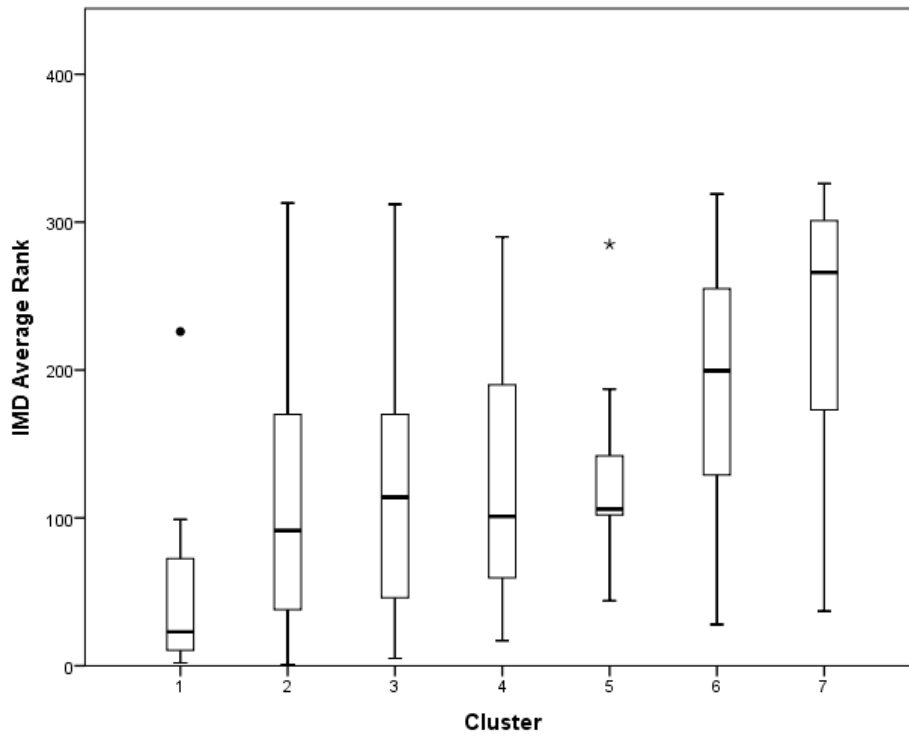
Note that there are three clusters for urban areas compared to the two clusters in the 6-cluster solution, specifically a new suburban cluster with greater natural capital (higher than average freshwater features and water quality). There are also slightly more districts assigned to the woodland cluster and slightly less to the agricultural cluster. Overall a similar spatial pattern emerges. Changes in average deprivation across clusters indicate woodland and agricultural areas are the least deprived and urban/suburban and mountainous areas are the most deprived. Deprivation values (both median and distribution) show minor difference between the two suburban clusters.



i)



ii)

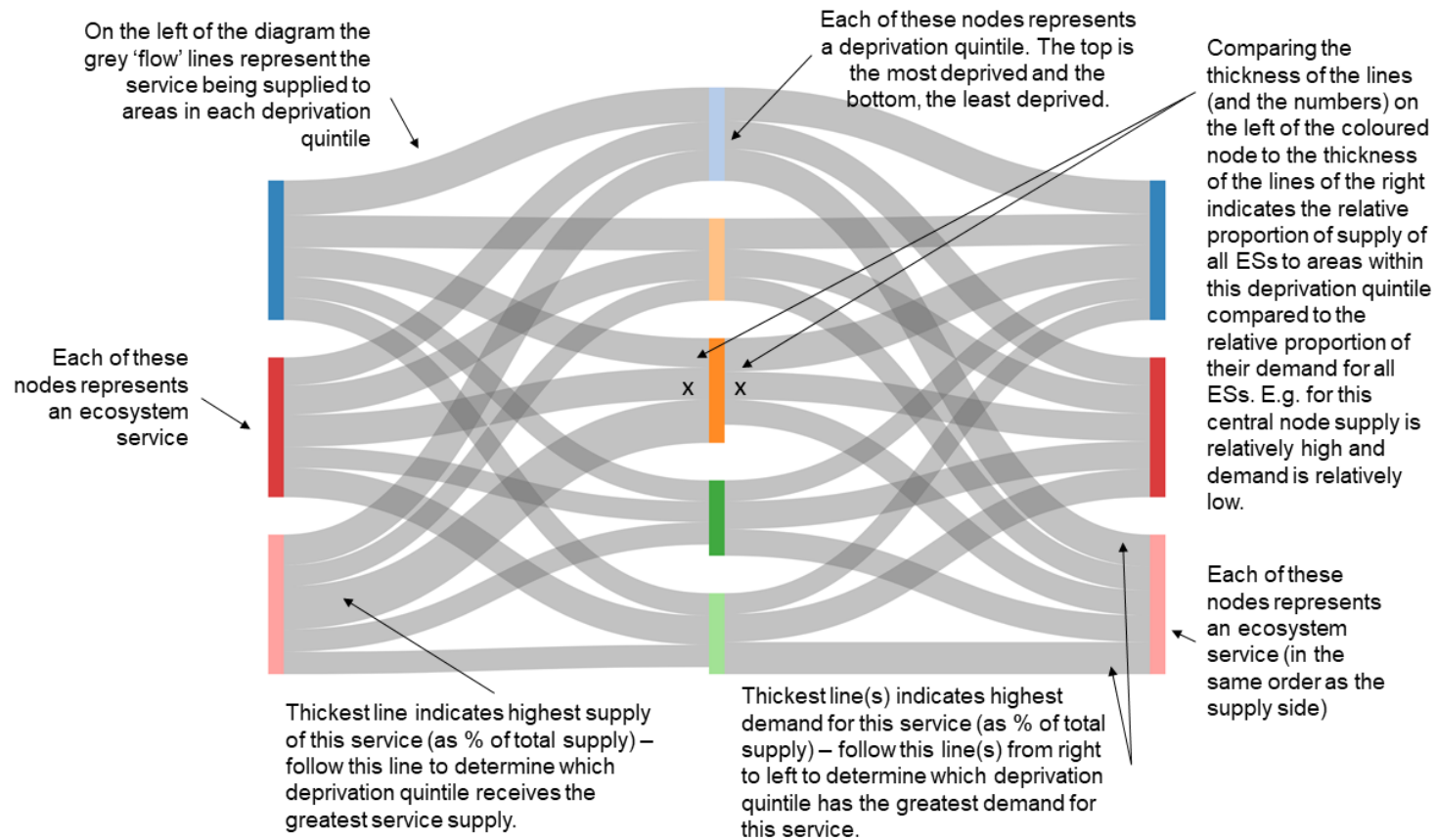


iii)

A.3. Sensitivity to selection of indicators

Natural capital indicator	Change in cluster membership (% total number of districts)
Ecological status	4.3
Quality of agricultural land	0.9
Agricultural	4.3
Coastal	0.9
Mountain	1.2
Low density built-up	1.9
High density built-up	14.8
Woodland	19.1
Freshwater	0.3
Quality of water	1.2
Publically accessible	1.2
Semi-natural grassland	0.9
Soil carbon	0.3
Protected status	4.3

A.4. Guidance on the interpretation of ES inequality Sankey diagrams presented in Chapter 8.



References

- Abercrombie, L.C., Sallis, J.F., Conway, T.L., Frank, L.D., Saelens, B.E., Chapman, J.E. 2008. Income and Racial Disparities in Access to Public Parks and Private Recreation Facilities. *American Journal of Preventive Medicine*. **34**(1), 9-15.
- Acheson, D. 1998. *Independent Inquiry into Inequalities in Health Report*. London: The Stationary Office.
- Acreman, M.C., Harding, R.J., Lloyd, C., McNamara, N.P., Mountford, J.O., Mould, D.J., Purse, B.V., Heard, M.S., Stratford & Dury, S.J. 2011. Trade-off in ecosystem services of the Somerset Levels and Moors wetlands. *Hydrological Sciences Journal*. **56**:8, pp.1543-1565,
- Addy, C.L. et al. 2004. Associations of perceived social and physical environmental supports with physical activity and walking behaviour. *American Journal of Public Health*, **94**, pp.440-443.
- Adekola, O., Mitchell, G., & Grainger, A. (2015). Inequality and ecosystem services: The value and social distribution of Niger Delta wetland services. *Ecosystem Services*, **12**, 42-54
- Agyeman, J. 2002. Constructing environmental justice: Transatlantic tales. *Environmental Politics*. **11**, pp.31-53.
- Agyeman, J. & Evans, B. 2004. 'just sustainability': the emerging discourse of environmental justice in Britain? *Geographical Journal*, **170**, pp.155-164.
- Agyeman, J. Schlosberg, D., Craven, L. & Matthews, C. 2016. Trends and Directions in Environmental Justice: From Inequity to Everyday Life, Community, and Just Sustainabilities. *Annual Review of Environment and Resources*. **41**, pp.321-40.
- Ala-Hulkko, T., Kotavaara, O., Alahuhta, J., Helle, P. & Hjort, J. 2016. Introducing accessibility analysis in mapping cultural ecosystem services. *Ecological Indicators*. **66**, pp.416-427.
- Alcock I., White M.P., Wheeler B.W., Fleming L.E., Depledge M.H. Longitudinal effects on mental health of moving to greener and less green urban areas. *Environment Science & Technology*. 2014. **48**, pp.1247–1255.
- Andersson, E., Barthel, S., & Ahrne, K. 2007. Measuring social-ecological dynamics behind the generation of ecosystem services. *Ecological Applications*, **17**(5), 1267-1278.
- Andersson, E., Nykvist, B., Malinga, R., Jaramillo, F. & Lindborg, R. 2015. A social-ecological analysis of ecosystem services in two different farming systems. *Ambio*. **44** S1, pp.102-112.
- Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy, D.B. & Gaston, K.J. 2009. Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology*. **46**(4), pp.888-896.
- Andrew, M.E., Wulder, M.A., Nelson, T.A. & Coops, N.C. 2015 Spatial data, analysis approaches, and information needs for spatial ecosystem service assessments: a review, *GIScience & Remote Sensing*, **52**:3, 344-373.
- Anselin, L. 1995. Local Indicators of Spatial Association – LISA. *Geographical Analysis*. **27**(2), pp.93-115.
- Anselin, L. 2003. GeoDa 0.9 User's Guide. [Online]. [Accessed 19/06/2019]. Available from: <https://s3.amazonaws.com/geoda/software/docs/geoda093.pdf>
- Armson, D., Stringer, P. & Ennos, A.R. 2013. The effect of street trees and amenity grass on urban surface runoff in Manchester, UK. *Urban Forestry and Urban Greening*, **12**, 282-286. Astell-Burt, T., Feng, X. Q., Mavoja, S., Badland, H. M.,

- & Giles-Corti, B. 2014. Do low-income neighbourhoods have the least green space? A cross-sectional study of Australia's most populous cities. *BMC Public Health*, 14.
- Aznar-Sánchez, J.A., Belmonte-Ureña, L.J., López-Serrano, M.J. & Velasco-Muñoz, J.F. 2018. Forest Ecosystem Services: An Analysis of Worldwide Research. *Forests*. 9(8), p. 453.
- Babisch, W. 2000. Traffic noise and cardiovascular disease: epidemiological review and synthesis. *Noise Health*, 2(8), pp.9-32.
- Baden, B. M., Noonan, D.S. & Turaga, R.M.R. 2007. Scales of justice: Is there a geographic bias in environmental equity analysis? *Journal of Environmental Planning and Management*. 50(2), pp.163-185.
- Bagstad, K., Semmens, D.J., Waage, S. & Winthrop, R. 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosystem Services*. 5, pp.27-39.
- Bagstad, K.J., Cohen, E., Ancona, Z.H., McNulty, S.G., Sun, G. 2018. The sensitivity of ecosystem service models to choices of input data and spatial resolution. *Applied Geography*, 93, pp.25-36.
- Balvanera, P., Pfisterer, A.B., N. Buchmann, N., He, J-S., Nakashizuka, T., Raffaelli, D., Schmid, B. 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters*. 9, pp.1146-1156.
- Barbosa, O., Tratalos, J. A., Armsworth, P. R., Davies, R. G., Fuller, R. A., Johnson, P., & Gaston, K. J. 2007. Who benefits from access to green space? A case study from Sheffield, UK. *Landscape and Urban Planning*. 832(3), pp.187-195.
- Baró, F., Palomo, I., Zulian, G., Vizcaino, P., Haase, D. & Gómez-Baggethun, E. 2016. Mapping ecosystem service capacity, flow and demand for landscape and urban planning: A case study in the Barcelona metropolitan region. *Land Use Policy*. 57, pp.405-417.
- Barton, H. 2005. A health map for urban planners: Towards a conceptual model for healthy, sustainable settlements. *Built Environment*, 31(4).
- Barton, J. & Petty, J. 2010. What is the best dose of nature and green exercise for improving mental health? A multi-study analysis. *Environmental Science and Technology*. 44(10), pp. 3947-55.
- Bastian, O., Syrbe, R-U., Rosenberg, M., Rahe, D. & Grunewald, K. 2013. The five pillar EPPS framework for quantifying, mapping and managing ecosystem services. *Ecosystem Services*. 4, pp.15–24.
- Bateman I.J., Garrod G.D., Brainard J.S., Lovett A.A. 1996. **Measurement issues in the travel cost method: a geographical information systems approach.** *Journal of Agricultural Economics*. 47(2), 191-205.
- Beaumont, N., Austen, M., Atkins, J., Burdon, D., Degraer, S., Dentino, T.P., Deros, S., Holm, P., Horton, T., van Ierland, E., Marboe, A.H., Starkey, D.J., Townsend, M. & Zarzycki, T. 2007. Identification, definition and quantification of goods and services provided by marine biodiversity: implications for the ecosystem approach. *Marine Pollution Bulletin*. 54(3), pp.253–265.
- Bedfordshire, Cambridgeshire & Northamptonshire Wildlife Trusts. no date. *Nene Valley Nature Improvement Area: Achieving for the people, places, and economy of Northamptonshire and Cambridgeshire*. [Online]. [Accessed 16/06/2019]. Available from: <https://www.wildlifebcn.org/nene-valley-nia>.
- Beckett, K.P., Freer-Smoth, P. & Taylor, G. 2000 Effective tree species for local air quality management. *Journal of Arboriculture*, 26(1), 12-19.
- Been, V. & Gupta, F. 1997. Coming to the nuisance or going to the barrios? A longitudinal analysis of environmental justice claims. *Ecology Law Quarterly*, 24, pp.1-56.

- Beichler, S.A. 2015. Exploring the link between supply and demand of cultural ecosystem services – towards an integrated vulnerability assessment. *International Journal of Biodiversity Science, Ecosystem Services & Management*. **11**(3), 250-263.
- Bennett, E.M., Cramer, W., Begossi, A., Cundill, G., Diaz, S., Egoh, B.N et al. (2015) Linking biodiversity, ecosystem services, and human well-being: three challenges for designing research for sustainability. *Current Opinion in Environmental Sustainability*, **14**, 76-85
- Benra, F. & Nahuelhual, L. 2019. A trilogy of inequalities: Land ownership, forest cover and ecosystem services distribution. *Land Use Policy*. **82**, pp.247-257.
- Berbes-Blazquez, M., Oestreicher, J. S., Mertens, F., & Saint-Charles, J. 2014. Ecohealth and resilience thinking: a dialog from experiences in research and practice. *Ecology and Society*, **19**(2).
- Biddle, S.J.H. & Ekkekakis, P. 2005. Physically active lifestyles and well-being. In Huppert, B. et al. (eds.) *The science of well-being*, pp.140-168. Oxford, UK: Oxford University Press.
- Bin, O., & Kruse, J. B. 2006. Real estate market response to coastal flood hazards. *Natural Hazards Review*. **74**, pp.137–144.
- Bird, W. 2007. *Natural Thinking: Investigating The Links Between The Natural Environment, Biodiversity And Mental Health*. RSPB.
- Birkhofer, K., Diehl, E., Andersson, J., Ekroos, J., Früh-Müller, A., Machnikowski, F., Mader, V.L., Nilsson, L., Sasaki, K., Rundlöf, M., Wolters, V. & Smith, H.G. 2015. Ecosystem services – current challenges and opportunities for ecological research. *Frontiers of Ecology and Evolution*. **2**, 87.
- Birkmann, J. 2007. Risk and vulnerability indicators at different scales: Applicability, usefulness and policy implications. *Environmental Hazards*. **7**, pp.20-31.
- Blaschke, T. 2006. The role of the spatial dimension within the framework of sustainable landscapes and natural capital. *Landscape and Urban Planning*. **75**, 3-4, pp.198-226.
- Bonner, J., Grigg, S., Hime, S., Hewitt, G., Jackson, R. and Kelly, M. (2012) *Is natural capital a material issue? An evaluation of the relevance of biodiversity and ecosystem services to accountancy professionals and the private sector*, ACCA, Flora & Fauna International, KPMG LLP.
- Boone, C.G., Buckley, G.L., Grove, J.M., Sister, C. 2009. Parks and people: An environmental justice inquiry in Baltimore, Maryland. *Annals of the Association of American Geographer*, **99** (4), pp. 767-787
- Boone, C.G. 2010. Environmental justice, sustainability and vulnerability. *International Journal of Urban Sustainable Development*. **2**(1-2), pp.135-140.
- Boorman, D.B., Hollis, J.M. & Lilly, A. 1995 *Hydrology of soil types: a hydrologically based classification of the soils of the United Kingdom*. Institute of Hydrology Report No. 126. 137pp.
- Bottalico, F., Travaglini, D., Chirici, G., Garfi, V., Giannetti, F., De Marco, A., Fares, S., Marchetti, M., Nocentini, S., Paoletti, E., Salbitano, F. & Sanesi, G. 2017. A spatially-explicit method to assess the dry deposition of air pollution by urban forests in the city of Florence, Italy. *Urban Forestry & Greening*. **27**, pp.221-234.
- Bowen, W. 2002. An analytical review of environmental justice research: What do we really know? *Environmental Management*, **29**, pp.3-15.
- Bowler, D.E., Buyung-Ali, L.M., Knight, T.M. & Pullin, A.S. 2010. A systematic review of evidence for the added benefits to health of exposure to natural environments. *BMC Public Health*. **10**, p.456.
- Boyce, J.K., Kwicki, K., Ash, M. 2016. Measuring environmental inequality. *Ecological Economics*. **124**, pp.114-123.

- Boyd, J. & Banzaf, S. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, **63**, pp.616-626.
- Boyd, F., White, M.P, Bell, S.L. & Burt, J. 2018. Who doesn't visit natural environments for recreation and why: A population representative analysis of spatial, individual and temporal factors among adults in England. *Landscape and Urban Planning*. **157**, pp.102-113.
- Boyle, P. & Alvanides, S. 2003. Assessing Deprivation in English Inner City Areas: Making the Case for EC Funding for Leeds City. In: Stillwell, J. & Clarke, G. eds. *Applied GIS and Spatial Analysis*. Chichester: Wiley.
- Braat, F., van der Ploeg, S.W.F. & Bouma, F. 1979. *Functions of the Natural Environment: an economic-ecological analysis*. Netherlands: WWF.
- Braat, L.C. & de Groot, R. 2012. The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. *Ecosystem Services*. **11**, pp. 4-15.
- Brainard, J., Bateman, I. & Lovett, A. 2001. Modelling demand for recreation in English woodlands. *Forestry*, **74**, 5, 423-438.
- Brainard, J. S., Jones, A.P., Bateman, I.J. & Lovett, A.A. 2002. Modelling environmental equity: access to air quality in Birmingham, England. *Environment and Planning A*, **34**, pp.695-716.
- Brainard, J.S, Jones, A.P., Bateman, I.J. & Lovett, A.A. 2004. Exposure to Environmental Urban Noise Pollution in Birmingham, UK. *Urban Studies*. **41**(13), pp.2581-2600.
- Briggs, D., Abellan, J.J. & Fecht, D. 2008. Environmental inequity in England: Small area associations between socio-economic status and environmental pollution. *Social Science & Medicine*, **67**, pp.1612-1629.
- Broadmeadow, M.S.J. & Freer-Smith, P.H. (1996) *Urban Woodland and the Benefits for Local Air Quality*. London: Department of Environment, HMSO.
- Brown, G., Montag, J.M. & Lyon, K. 2012. Public Participation GIS: A Method for Identifying Ecosystem Services. *Society & Natural Resources*. **25**(7), pp.633-651.
- Brown, P. 1995. Race, class and environmental health - a review and systemization of the literature. *Environmental Research*, **69**, pp.15-30.
- Brown, L. & Murray, V. 2013. Examining the relationship between infectious diseases and flooding in Europe. A systematic literature review and summary of possible public health interventions. *Disaster Health*, **1**, pp.1-11.
- Brown, T.C., Bergstrom, J.C. & Loomis, J. 2007. Defining, valuing, and providing ecosystem goods and services. *Natural Resources Journal*. **47**(20), pp.329-376.
- Brunt, H., Barnes, J., Jones, S.J., Longhurst, J.W.S., Scally, G. & Hayes, E. 2017. Air pollution, deprivation and health: understanding relationships to add values to local air quality management policy and practice in Wales, UK. *Journal of Public Health*. **39**(3), pp. 485-497.
- Buchecker, M. & Degenhardt, B. 2015. The effects of urban inhabitants' nearby outdoor recreation on their well-being and their psychological resilience. *Journal of Outdoor Recreation and Tourism*. **10**, pp.55-62.
- Bullard, R. D. 1983. Solid wastes sites and the Black Houston community. *Sociological Inquiry*, **53**, pp.273-288.
- Bullard, R. 1990. *Dumping in Dixie: Race, Class, and Environmental Quality*. Westview, Boulder, CO.
- Burkhard, B., Kroll, F., Nedkov, S. and Muller, F. 2012. Mapping ecosystem service supply, demand and budgets. *Ecological Indicators*, **21**, 17-29

- Burkhard, B., Kandziora, M., Hou, Y. & Müller, F. 2014 Ecosystem service potentials, flows and demands – concepts for spatial localization, indication and quantification. *Landscape Online*, **34**, 1–32.
- Butler, R (2004) Geographical research on tourism, recreation and leisure: origins, eras and directions, *Tourism Geographies*, **6**(2), pp.143-162.
- Butterfield, D. & Relf, D. 1990. National survey of attitudes toward plants and gardening. In: Relf D, ed. The role of horticulture in human well-being and social development: a national symposium. In: Relf, D. (ed.) *The role of horticulture in human well-being and social development: a national symposium, 19–21 April 1990, Arlington, Virginia. Portland.*
- Buzzelli, M. 2007. Bourdieu does environmental justice? Probing the linkage between population health and air pollution epidemiology. *Health & Place*, **13**(1), pp.3-13.
- CABE Space (2010) *Urban green nation: Building the evidence base*. [online]. London: CABE. [03/03/19] Available from: <https://webarchive.nationalarchives.gov.uk/20110118110347/http://www.cabe.org.uk/files/urban-green-nation.pdf>
- Cambra, K., Martinez-Rueda, T., Alonso-Fustel, E., Cirarda, F. B., Audicana, C., Esnaola, S., & Ibanez, B. (2013). Association of proximity to polluting industries, deprivation and mortality in small areas of the Basque Country (Spain). *European Journal of Public Health*, **23**(1), 171-176.
- Campbell, J.B. & Wynne, R.H. (2011) *Introduction to Remote Sensing*. Fifth Edition. New York: The Guildford Press
- Caporn, S.J.M. & Emmett, B.A. 2009. Threats from air pollution and climate change to upland systems: Past, present and future. In Bonn, A., Allott, T., Hubacek, K. & Stewart, J. eds. *Drivers of Environmental Changes in Uplands*. London: Routledge.
- Casado-Arzuaga, I., Onaindia, M., Madariaga, I. & Verburg, P.H. (2014) Mapping recreation and aesthetic value of ecosystems in the Bilbao Metropolitan Greenbelt (northern Spain) to support landscape planning. *Landscape Ecology*, **29**, 1393-1405.
- Casalegno, S. et al. 2014. Regional scale prioritisation for key ecosystem services, renewable energy production and urban development. *PloS one*, **9**, pp.e107822-e107822.
- CBD 1992. *Convention on Biological Diversity*. [Online]. Entered in force on 29 December 1993. [Accessed 21/06/2019]. Available from: <https://www.cbd.int>.
- CBD. 2004. *The Ecosystem Approach*. Montreal: Secretariat of the Convention on Biological Diversity. [Online]. [Last Accessed 16/06/2019]. Available from: <https://www.cbd.int/ecosystem/>
- CEH 2011 *Land Cover Map 2007 Dataset Documentation, Version 1.0, 06 July 2011*, Wallingford.
- Centre for Cities. 2015. *Cities Outlook*. London: Centre for Cities. [Online]. [Accessed 29/05/2019]. Available from: www.centreforcities.org.
- Centre for Cities. 2019. *Cities Outlook*. London: Centre for Cities. [Online]. [Accessed 29/05/2019]. Available from: www.centreforcities.org.
- Cerqueira, Y., Navarro, L.M., Maes, J., Marta-Pedrosa, C., Pradinho Honrado, J. & Pereira, H.M. 2015. Ecosystem services: the opportunities of rewilding in Europe. In: Pereira, H.M. & Navarro, L.M. eds. *Rewilding European Landscapes*. London: Springer, pp.47-64.
- Chakraborty, A. & Armstrong, M. P. 1997. Exploring the Use of Buffer Analysis for the Identification of Impacted Areas in Environmental Equity Assessment. *Cartography and Geographic Information Systems*, **24**, pp.145-157.

- Chakraborty J, Collins T, Grineski S, Montgomery M, Hernandez M. 2014. Comparing disproportionate exposure to acute and chronic pollution risks: A case study in Houston, Texas. *Risk Analysis*. **34**, pp.2005–2020.
- Chakraborty, J. & Basu, P. 2019. Linking Industrial Hazards and Social Inequalities: Environmental Injustice in Gujarat, India. *International Journal of Environmental Research and Public Health*. **16**(42).
- Chang, C. et al. 2007. A preliminary study on the local cool-island intensity of Taipei city parks. *Landscape and Urban Planning*, **80**, pp.386-395.
- Charlier, T. 1993. 'Environmental Racism' Not Clear-Cut Issue in Memphis. *Memphis Commercial Appeal*, September 19, 1993.
- Charrad, M., Ghazzali, N., Boiteau, V., & Niknafs, A. 2014. Nbclust: An R Package for Determining the Relevant Number of Clusters in a Data Set. *Journal of Statistical Software*, **61**(6), 1-36.
- Chaudhary, S., McGregor, A., Houston, D. & Chettri, N. 2015. The evolution of ecosystem services: A time series and discourse-centred analysis. *Environmental Science & Policy*. **54**, pp.25-34.
- Chen, J., Ni, J., Xi, C., Li, S. & Wang, J. 2017 Determine intra-urban spatial accessibility disparities in multimodal public transport networks. *Journal of Transport Geography*, **65**, pp.123-133.
- Christie, M. & Rayment, M. 2012. An economic assessment of the ecosystem service benefits derived from the SSSI biodiversity conservation policy in England and Wales. *Ecosystem Services*. **1**(1), pp.70-84.
- Church, A. et al. 2014. UK National Ecosystem Assessment Follow-On. *Work Package Report 5: Cultural ecosystem services and indicators*. UK: UNEP-WCMC, LWEC.
- Clean Air Act 1993 (c.11). London: HMSO.
- Coen, S. E. & Ross, N. A. 2006. Exploring the material basis for health: Characteristics 25 of parks in Montreal neighbourhoods with contrasting health outcomes. *Health & Place*, **12**, pp.361-371.
- Cohen, S. 1973 Apartment Noise, Auditory Discrimination, and Reading Ability in Children. *Journal of Experimental Social Psychology*, *Public Health*, **120**(12), pp.1127-1132.
- Collingwood Environmental Planning. 2015. *Monitoring and Evaluation of Nature Improvement Areas: Final Report 2012-15*. Defra Research Project WC1061. [Online]. [Accessed 20/06/2019]. Available from: <http://randd.defra.gov.uk>.
- Comber, A., Brunsdon, C. & Green, E. 2008 Using a GIS-based network analysis to determine urban greenspace accessibility for different ethnic and religious groups. *Landscape and Urban Planning*, **86**, pp.103-114.
- COMEAP 2010. *The Mortality Effects of Long-Term Exposure to Particulate Air Pollution in the United Kingdom*. A report by the Committee on the Medical Effects of Air Pollutants. London: The Stationery Office.
- COMEAP 2018. *Associations of long-term average concentrations of nitrogen dioxide with mortality: A report by the Committee on the Medical Effects of Air Pollutants*. [Online]. Public Health England. [23/04/2019]. Available from: <https://assets.publishing.service.gov.uk>
- Coombes, E. et al. 2010. The relationship of physical activity and overweight to objectively measured green space accessibility and use. *Social Science & Medicine*, **70**(6), pp.816–822.
- Cooper. R et al. 2008. Mental Capital And Wellbeing: Making The Most Of Ourselves In The 21st Century. *State-of-Science Review: SR-DR2. The Effect of the Physical Environment on Mental Wellbeing*. GO Science/ Foresight.

- Cornell, S. 2011. The Rise and Rise of Ecosystem Services: Is “value” the best bridging concept between society and the natural world? *Procedia Environmental Sciences*, **6**, pp.88-95.
- Costanza, R. & Daly, H. 1992. Natural capital and sustainable development. *Conservation Biology*, **6**, pp.37-46.
- Costanza, R. and Folke, C. 1997. Valuing ecosystem services with efficiency, fairness, and sustainability as goals. In Daily, G. (ed.) *Nature's Services: Societal Dependence on Natural Ecosystems* (pp.49-68). Washington D.C. – Island Press.
- Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature*, **387** (6630), pp.241-248.
- Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S. & Grasso, M. 2017. Twenty years of ecosystem services: How far have we come and how far do we still need to go? *Ecosystem Services*, **28**, pp.1-16.
- Cotton, M. & Devine-Wright, P. 2013. Putting pylons into place: a UK case study of public perspectives on the impacts of high voltage overhead transmission lines. *Journal of Environmental Planning and Management*, **56**, pp.1225-1245.
- Crossman, N.D., Burkhard, B., Nedkov, S., Willemsen, L., Petz, K., Palomo, I., Drakou, E. 2013. A blueprint for mapping and modelling ecosystem services. *Ecosystem Services*, **4**, pp.4-14.
- CSDH 2008 Commission on Social Determinants of Health: *Closing the gap in a generation: Health equity through action on the social determinants of health*. Geneva, Switzerland: World Health Organization.
- Curtis, S., 2004. *Health and inequality, geographical perspectives*. London: Sage Publications
- Cushing, L., Morello-Frosch, R., Wander, M. & Pastor, M. 2015. The Haves, the Have-Nots, and the Health of Everyone: The Relationship Between Social Inequality and Environmental Quality. *Annual Review of Public Health*, **36**, pp.193-209.
- Cutter, S. L. 1995. Race, class and environmental justice. *Progress in Human Geography*, **19**, pp.111-122
- Cutter, S.L. 2009. Social Science Perspectives on Hazards and Vulnerability. In Beer, T. ed. *Geophysical Hazards: Minimizing Risk, Maximizing Awareness*. Dordrecht: Springer, pp.17-30.
- Cutter, S.L. and Finch, C. 2008. Temporal and spatial changes in social vulnerability to natural hazards. *PNAS*, **105**(7). P. 2301-2306.
- Dadson, S.J., Hall, J.W., Murgatroyd, A., Acreman, M., Bates, P., Beven, K., Heathwaite, L., Holden, J., Holman, I.P., Lane, S.N., O'Connell, E.P-R, Reynard, N., Sear, D., Thorne, C. & Wilby, R. (2017) A restatement of the natural science evidence concerning catchment-based 'natural' flood management in the UK. *Proceedings of the Royal Society A*, **473**: 20160706.
- Dahlgren, G. & Whitehead, M. 1991. *Policies and Strategies to Promote Equity in Health*, Stockholm, Institute for Future Studies.
- Dai, D. 2011. Racial/ethnic and socioeconomic disparities in urban green space accessibility: Where to intervene? *Landscape and Urban Planning*, **102**(4), pp.234-244.
- Daily, G. 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*, Washington D.C., Island Press.
- Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J. & Shallenberger, R. 2009. Ecosystem services in decision making: time to deliver. *Frontiers in Ecology & Environment*, **7**(1), pp.21-28.

- Dales, N.P., Brown, N.J. & Lusardi, J. (2014) Assessing the potential for mapping ecosystem services in England based on existing habitats. *Natural England Research Reports, Number 056*.
- Dallimer, M., Davies, Z.G., Irvine, K.N., Maltby, L., Warren, P.H., Gaston, K.J. & Armsworth, P.R. (2014) What Personal and Environmental Factors Determine Frequency of Urban Greenspace Use? *International Journal of Environmental Research and Public Health*, **11**, 7977-7992.
- Daly, H. (1973) *Toward a steady state economy*. Freeman: San Francisco.
- Daly, H.E. 1977. *Steady State Economics*. San Francisco: W.H.Freeman.
- Damery, S., Petts, J., Walker, G. & Smith, G. 2008. *Addressing environmental inequalities: waste management*. Bristol: Environment Agency.
- Davies, Z. G., Edmondson, J.L., Heinemeyer, A., Leake, J.R. & Gaston, K.J. 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. *Journal of Applied Ecology*, **48**, pp.1125-1134.
- Davis, A. Y., Belaire, J. A., Farfan, M. A., Milz, D., Sweeney, E. R., Loss, S. R., & Minor, E. S. (2012). Green infrastructure and bird diversity across an urban socioeconomic gradient. *Ecosphere*, **3**(11). Davies, H.J., Doick, K.J., Hudson, M.D. 2017. Challenges for tree officers to enhance the provision of regulating ecosystem services from urban forests. *Environmental Research*, **156**, pp.97-107.
- Daw, T., Brown, K., Rosendo, S. & Pomeroy, R. 2011 Applying the ecosystem services concept to poverty alleviation: the need to disaggregate human well-being. *Environmental Conservation*, **38**(4), pp.370-379.
- Day, B. H. & Smith, G. 2018. *Outdoor Recreation Valuation ORVal. User Guide: Version 2.0*. Exeter: Land, Environment, Economics and Policy LEEP Institute, University of Exeter.
- DCLG. 2013. Land Use Change Statistics in England: 2011 *Land Use Statistical Release*. London: Department for Communities and Local Government.
- DCLG. 2015. *The English Indices of Deprivation 2015: Technical Report*. London: DCLG. [Online]. [Accessed 20/06/2019]. Available from: <https://www.gov.uk/government/publications/english-indices-of-deprivation-2015-technical-report>
- DEFRA 2011. *The Natural Choice: securing the value of nature*. London: The Stationary Office
- DEFRA 2013. *Payment for Ecosystem Services (PES) Pilot on Flood Regulation in Hull*. Department for Environment, Food and Rural Affairs. [Accessed 10/11/2014]. Available from: <http://randd.defra.gov.uk/>
- DEFRA 2016 Defra's Payments for Ecosystem Services Pilot Projects 2012-15: Review of key findings. [Online]. [Accessed 21/06/2019]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/578005/pes-pilot-review-key-findings-2016.pdf
- DEFRA 2017 *Air Pollution in the UK report*. [online]. [03/03/2019] Available from: https://uk-air.defra.gov.uk/library/annualreport/viewonline?year=2017_issue_1#report_pdf
- DEFRA 2019. *Clean Air Strategy 2019*. [Online]. [Accessed 07/05/2019]. Available from: <https://www.gov.uk/government/publications/clean-air-strategy-2019>.
- De Groot, R.S., Wilson, M.A. & Boumans, R.M.J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*. **41**(3), pp. 393-408.
- De Groot, R., Van der Perk, J., Chiesura, A. & van Vliet, A. 2003. Importance and threat as determining factors for criticality of natural capital. *Ecological Economics*, **44**, pp.187-204.

- De Groot, R.S., Fisher, B., Christie, M., Aronson, J., Braat, L.R., Haines-Young, R., Gowdy, J., Maltby, E., Neuvill, A., Polasky, S., Portela, R. & Ring, I. 2010 **Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation**. In: Kumar, P. Ed. *TEEB Foundations, The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*. London: Earthscan.
- De Sousa Silva, C., Viegas, I., Panagopoulos, T. & Bell, S. 2018. Environmental Justice in Accessibility to Green Infrastructure in Two European Cities. *Land*, **74**, p.134.
- De Valck, J., Landuyt, D., Broekx, S., Liekens, I., De Nocker, L. & Vranken, L. 2017. Outdoor recreation in various landscapes: Which site characteristics really matter? *Land Use Policy*, **65**, pp.186-197.
- De Vries, S. et al. 2002. Natural environments - healthy environments? An exploratory analysis of the relationship between greenspace and health. *Environment and Planning A*, **35**, pp.1731.
- Dick, J., Maes, J., Smith, R.I., Paracchine, M-L. & Zulian, G. 2014. Cross-scale analysis of ecosystem services identified and assessed at local and European level. *Ecological Indicators*, **38**, pp.20-30.
- Dick, J., Andrews, C., Beaumont, D., Benham, S., Dodd, N., Pallett, D., Rose, R., Scott, T., Smith, R., Schafer, S., Turner, A. and Watson, H. 2016. Analysis of temporal change in delivery of ecosystem services over 20 years at long term monitoring sites of the UK Environmental Change Network. *Ecological Indicators*, **68**, 115-125.
- Dittrich, A., Seppelt, R., Václavík, T. and Cord, A.F. 2017 Integrating ecosystem service bundles and socio-environmental conditions – A national scale analysis from Germany. *Ecosystem Services*, **28C**, pp.273-282.
- Dobson, A. 2003. Social justice and environmental sustainability: Ne'er the twain shall meet? In J Agyeman, J., Bullard, R. & Evans, B. (Eds.), *Just Sustainabilities: Development in an Unequal World* (pp.83–95). London: Earthscan.
- Donovan, R. G. et al. 2005. Development and Application of an Urban Tree Air Quality Score for Photochemical Pollution Episodes Using the Birmingham, United Kingdom, Area as a Case Study. *Environment, Science and Technology*, **39**, pp.6730-6738.
- Dorling, D. 1996. Be cheerful, strive to be happy identifying disadvantaged areas: health, wealth and happiness. *Radical Statistics*, **62**, 8-21.
- Duku, C., Rathjens, H., Zwart, S.J. & Hein, L. 2015. Towards ecosystem accounting: a comprehensive approach to modelling multiple hydrological ecosystem services. *Hydrology and Earth Systems Science*, **19**, 4377-4396.
- Duncan, M. & Mummery, K. 2005. Psychosocial and environmental factors associated with physical activity among city dwellers in regional Queensland. *Preventive Medicine*, **40**, pp.363-372.
- Dzierżanowski, K., Popek, R., Gawrońska, H., Sæbø, A. & Gawroński, S.W. 2011. Deposition of Particulate Matter of Different Size Fractions on Leaf Surfaces and in Waxes of Urban Forest Species. *International Journal of Phytoremediation*, **13**(10), pp.1037-1046
- EAC 2012. Measuring well-being and development: Sustainable Development Indicators. Fifth Report of Session 2012-2013. Volume 1. London: HMSO. [Online]. [Accessed 25/06/2019]. Available from: <https://publications.parliament.uk/pa/cm201213/cmselect/cmenvaud/667/667.pdf>
- EC 2008. *Council Directive 2008/50/EC of 21 May 2008 on ambient air quality and cleaner air for Europe*

- EC 2007 Council Directive 2007/60/EC of the European Parliament and of the Council of 23 October 2007 on the assessment and management of Flood risks. [Online]. [19/04/2019]. Available from: http://ec.europa.eu/environment/water/flood_risk/key_docs.htm#Directive
- EC. 2011. *The EU Biodiversity Strategy to 2020*. Luxembourg: Publications Office of the European Union. [Online]. [Accessed 20/06/2019]. Available from: <http://ec.europa.eu/environment/nature/info/pubs/docs/brochures/2020%20Biod%20brochure%20final%20lowres.pdf>
- Echenique, M. H., Hargreaves, A.J., Mitchell, G. & Namdeo, A. 2011. Growing Cities Sustainably: Does Urban Form Really Matter? *Journal of the American Planning Association*, **78** (2), pp.121-137.
- EEA 2018. *Air quality in Europe - 2018 report*. No 12/2018. [Online]. Luxembourg: European Environment Agency. [23/04/2019]. Available from: <https://www.eea.europa.eu/publications/air-quality-in-europe-2018>
- Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, C.D. & van Jaarsveld, A.S. 2008. Mapping ecosystem services for planning and management. *Agriculture, Ecosystems & Environment*, **127** (1-2), pp.135-140.
- Egoh, B. et al. 2012 *Indicators for mapping ecosystem services: a review*. Report EUR 25456 EN. Publications Office of the European Union, Luxembourg.
- Ehrlich, P.R. & Mooney, A.H. 1981. *Extinction: The Causes and Consequences of the Disappearance of Species*. New York: Random House.
- Eigenbrod, F. et al. 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. *Journal of Applied Ecology*, **47**, pp.377-385.
- Eigenbrod, F., Bell, V.A., Davies, H.N., Heinemeyer, A., Armsworth, P.R. & Gaston, K.J. 2011. The impact of projected increases in urbanization on ecosystem services. *Proceedings of the Royal Society B*. **278**, pp. 3201-3208.
- Eigenbrod, F., Tang, Z., Eisner, S., Flörke, M. & Zhao, G. 2017. Spatial covariance of ecosystem services and poverty in China. *International Journal of Biodiversity Science, Ecosystem Services & Management*, **13**(1), pp.422-433.
- Ekins, P., Simon, S., Deutsch, L., Folke, C. & De Groot, R. 2003. A framework for the practical application of the concepts of critical natural capital and strong sustainability. *Ecological Economics*, **44**, 165-185.
- Ekkel, E.D. & de Vries, S. 2017. Nearby green space and human health: Evaluating accessibility metrics. *Landscape and Urban Planning*. **157**, pp.214-220.
- Ellaway, A. et al. 2005. Graffiti, greenery, and obesity in adults: secondary analysis of European cross-sectional survey. *British Medical Journal*, **331**, pp.611-2
- Elvers, H.D. et al. 2008. The Diversity of Environmental Justice: Towards a European Approach. *European Societies*, **10**, pp.835-856.
- Environment Agency 2006. *Developing social appraisal criteria for the Environment Agency. Science Report SC010044/SR3*. Bristol: Environment Agency. [Online]. [Accessed 01/06/2019]. Available from: <https://www.gov.uk/government/publications/developing-social-appraisal-criteria-for-the-environment-agency>.
- Environment Agency 2006. *Addressing Environmental Inequalities: Flood Risk*. Bristol: Environment Agency. [Online]. [Accessed 20/06/2019]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/291064/scho0905bjol-e-e.pdf.
- Environment Agency 2004. *Addressing Environmental Inequalities – Environment Agency position statement, October 2004*. Bristol: Environment Agency.
- Environment Agency 2017a. *Risk of Flooding from Multiple Sources: Dataset Documentation*. [Online]. [07/06/2018] Available from: data.gov.uk

- Environment Agency 2017b. *Working with Natural Processes to reduce flood risk* [online]. [04/04/2019] Available from <https://www.gov.uk/government/publications/working-with-natural-processes-to-reduce-flood-risk>
- Equality Act. 2010. (c.15). London: HMSO.
- Erkip, F. 1997. The distribution of urban public services: the case of parks and recreational services in Ankara. *Cities*, **14**(6), pp.353-361.
- Ernstson, H. 2013. The social production of ecosystem services: A framework for studying environmental justice and ecological complexity in urbanized landscapes. *Landscape and Urban Planning*, **109**(1), 7-17.
- Escobedo, F., & Nowak, D.J. 2009. Spatial heterogeneity and air pollution removal by an urban forest. *Landscape and Urban Planning*, **90**, pp.1020-110.
- Escobedo, F.J., Clerici, N., Staudhammer, C.L., Corzo, G.T. 2015. Socio-ecological dynamics and inequality in Bogotá, Colombia's public urban forests and their ecosystem services, *Urban Forestry and Urban Greening*, **14**(4), pp.1040-1053.
- ESPA 2018 Ecosystem Services for Poverty Alleviation: Programme Highlights 2009-2018. Edinburgh: Research into Results Limited. [Online]. [Accessed 21/06/2019]. Available from: http://www.espa.ac.uk/files/espa/ESPA_ProgrammeHighlights_Final_web.pdf
- ESRI 2011. *Arc Hydro Tools – Tutorial Version 2.0*. [Online]. Redlands: ESRI [03/03/19] Available from: downloads.esri.com/archydro.
- Evensen, K.H., Nordh, H. & Skaar, M. 2017. Everyday use of urban cemeteries: A Norwegian case study. *Landscape and Urban Planning*, **150**, pp.76-84.
- Exworthy, M. et al. 2003. Tackling Health Inequalities in the United Kingdom: The Progress and Pitfalls of Policy. *Health Services Research*, **38**, pp.1905-1922.
- Fairburn, J., Maier, W., & Braubach, M. 2016. Incorporating Environmental Justice into Second Generation Indices of Multiple Deprivation: Lessons from the UK and Progress Internationally. *International Journal of Environmental Research and Public Health*, **13**(8).
- Fairburn, J., Butler, B. & Smith, G. 2009. Environmental justice in South Yorkshire: locating social deprivation and poor environments using multiple indicators. *Local Environment*, **14**(2), p.139-154.
- Farley, J. & Costanza, R. 2010. Payments for ecosystem services: From local to global. *Ecological Economics*, **69**(11), pp.2060-2068.
- Felipe-Lucia, M.R., Martin-Lopez, B., Lavorei, S., Berraquero-Diaz, L., Escalera-Reyes, J., Comin, F.A. 2015. Ecosystem Service Flows: Why Stakeholders' Power Relationships Matter. *PloS ONE*, **10**(7), e0132232.
- Ferguson, M., Roberts, H.E., McEachan, R.R.C. & Dallimer, M. 2018. Contrasting distributions of urban green infrastructure across social and ethno-racial groups. *Landscape and Urban Planning*, **175**, pp.136-148.
- Fielding, J.L. & Burningham, K. 2005. Environmental inequality and flood hazard. *Local Environment*, **10**, pp.379-395.
- Firbank, L., Bradbury, R.B., McCracken, D.I. & Stoate, C. 2013. Delivering multiple ecosystem services from Enclosed Farmland in the UK. *Agriculture, Ecosystems & Environment*, **166**, pp.65-75.
- Fischer, L.K., Honold, J., Botzar, A., Brinkmeyer, D., Cvejić, R., Elands, B., Haase, D., Kabisch, N., Karle, S.J., Laforteza, R., Nastran, M., Nielsen, A.B., van der Jagt, A.P., Vierikko, K., Kowarik, I. (2018) Recreational ecosystem services in European cities: Sociocultural and geographical contexts matter for park use. *Ecosystem Services*, **31**, 455-467.
- Fisher, B. & Turner, R.K., 2008. Ecosystem services: classification for valuation. *Biological Conservation*, **141**, pp.1167–1169.

- Fisher, B., Turner, R. K. and Morling, P. 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics*, **68**(3), pp.643-653.
- Fisher, J.A., Patenaude, G., Meir, P., Nightingale, A.J., Rounsevell, M.D.A., Williams, M. & Woodhouse, I.H. 2013. Strengthening conceptual foundations: Analysing frameworks for ecosystem services and poverty alleviation research. *Global Environmental Change*, **23**, pp.1098-1111.
- Flood and Water Management Act 2010 (c.29). London: HMSO
- Forestry Commission 2019. *Urban Tree Challenge Fund: Guidance* [Online]. [Accessed 29/05/2019]. Available from: <https://www.gov.uk/guidance/urban-tree-challenge-fund>.
- Foster, J. 1997. Introduction: Environmental value and the scope of economics. In: Foster, J. (ed.) *Valuing Nature?* London: Routledge.
- Fotheringham, A.S. & Wong, D. 1991. The Modifiable Areal Unit Problem in Multivariate Statistical Analysis. *Environment and Planning A*, **23**, pp.1025-1044
- Fotheringham, A.S., Brunson, C. & Charlton, M. 2000. *Quantitative Geography: Perspectives on Spatial Data Analysis*. London: Sage Publications.
- Freeman, A.M. III. 1972. Distribution of environmental quality. In A. V. Kneese & B. T. Bower (Eds.), *Environmental quality Analysis: Theory and Method in the social sciences* (pp. 243-278). Baltimore, Md: The John Hopkins Press.
- Freer-Smith, P.H., Beckett, K.P. & Taylor, G. 2005. Deposition velocities to *Sorbus aria*, *Acer campestre*, *Populus deltoides* X *trichocarpa* "Beaupre", *Pinus nigra* and *Cupressocyparis leylandii* for coarse, fine and ultra- fine particles in the urban environment, *Environmental Pollution*, **133** (1), 157–67.
- Friends of the Earth 2004. *Incinerators and Deprivation*. London: FoE.GAO 1983. *Siting of Hazardous Waste Landfills And Their Correlation With Racial And Economic Status Of Surrounding Communities*. Gaithersburg: United States General Accounting Office.
- Fowler, D., Skiba, U., Nemitz, E., Choubedar, F., Branford, D., Donovan, R. and Rowland, P. 2004. Measuring aerosol and heavy metal deposition on urban woodland and grass using inventories of ²¹⁰Pb and metal concentrations in soil. *Water, Air and Soil Pollution*, **4**, 483–499.
- Fu, B., Wang, Y.K., Xu, P. & Yan, K. 2012. Mapping the flood mitigation services of ecosystems – A case study in the Upper Yangtze River Basin. *Ecological Engineering*, **52**, 238-246.
- Garcia Nieto, P., Garcia Llorente, M., Iniesta Arandia, I. & Martin Lopez, B. 2013. Mapping forest ecosystem services: from providing units to beneficiaries. *Ecosystem Services*, **4**, pp. 126-138
- Gascon, M., Triguero-Mas, M., Martinez, D., Dadvand, P., Forn, J., Plasencia, A. & Nieuwenhuijsen, M.J. 2015. Mental Health Benefits of Long-Term Exposure to Residential Green and Blue Spaces: A Systematic Review. *International Journal of Environmental Resources and Public Health*, **12**(4), pp. 4354-4379.
- Gehlke, C. E. and Biehl, K. 1934. Certain effects of grouping upon the size of the correlation coefficient in census tract material. *Journal of the American Statistical Association*, **S29**, pp.169-170.
- Geijzendorffer, I.R. & Roche, P.K. 2014. **Relevant scales of ecosystem service demand**. *Ecosystem Services*, **10**, pp. 49-51.
- Giles-Corti, M. et al. 2005. Increasing walking: how important is distance to, attractiveness, and size of public open space? *American Journal of Preventative Medicine*, **28**, pp.169-176.
- Gill, S. 2005. *Climate Change and Urban Greenspace*. Ph.D. thesis, University of Manchester.
- Gini, C. 1912. Variabilità e Mutuabilità. Contributo allo Studio delle Distribuzioni e delle Relazioni Statistiche. C. Cuppini, Bologna.

- Gobster, P.H. & Westphal, L.M. 2004. The human dimensions of urban greenways: Planning for recreation and related experiences. *Landscape and Urban Planning*, **68**(2-3), pp. 147-165.
- Goddard, M. A., Dougill, A. J., & Benton, T. G. 2010. Scaling up from gardens: biodiversity conservation in urban environments. *Trends in Ecology & Evolution*, **25**(2), 90-98.
- Goldman, R.L., Thompson, B.H. & Daily, G.C. 2007. Institutional incentives for managing the landscape: inducing cooperation for the production of ecosystem service. *Ecological Economics*. **64**, pp.333-343.
- Goldthorpe, J.H. *On Sociology*, vol. 2. (2nd ed.). Stanford: Stanford University Press.
- Gómez-Baggethun, E., de Groot, R., Lomas, P.L. & Montes, C. 2010. The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics*, **69** (6), 1209-1218.
- Gomez-Baggethun, E. and Barton, D. N. 2013. Classifying and valuing ecosystem services for urban planning. *Ecological Economics*, **86**, pp.235-245.
- Gomes Lopes, L. F., dos Santos Bento, J.M.R., Correia Cristovao, A.F.A. & Oliveira Baptista, F. 2015. Exploring the effect of land use on ecosystem services: The distributive issues. *Land Use Policy*, **45**, pp.141-149.
- Gordon, D & Pantazis, C 1997, Measuring poverty: breadline Britain in the 1990s. in D Gordon & C Pantazis (eds), *Breadline Britain in the 1990s*. Ashgate Publishing Ltd., pp. 5 - 48.
- Gordon-Larsen, P. et al. 2006. Inequality in the built environment underlies key health disparities in physical activity and obesity. *Pediatrics*, **117**, pp.417-424.
- Gould, J. 1986. *Quality of Life in American Neighborhoods: Levels of Affluence, Toxic Waste, and Cancer Mortality in Residential Zip Code Areas*. Westview, Boulder, CO.
- Graham, L.J., Spake, R., Gillings, S., Watts, K. & Eigenbrod, F. 2019. Incorporating fine-scale environmental heterogeneity into broad-extent models. *Methods in Ecology & Evolution*, **10**(6), pp.767-778.
- Grand-Clement, E., Anderson, K., Smith, D., Luscombe, D., Gatis, N., Ross, M. & Brazier, R.E. 2013. Evaluating ecosystem goods and services after restoration of marginal upland peatlands in South-West England. *Journal of Applied Ecology*, **50**, pp.324-334.
- Green, M. A., Vickers, D., & Dorling, D. 2014. A neighbourhood level mortality classification of England and Wales, 2006-2009. *Health & Place*, **30**, pp.196-204.
- Grêt-Regamey, A., Brunner, S.H., Altwegg, J. & Bebi, P. 2013. Facing uncertainty in ecosystem services based resource management. *Journal of Environmental Management*. **127S**, pp.S145-S154.
- Grineski, S., Collins, T.W., Chakraborty, J. & Montgomery, M. 2015. Hazardous air pollutants and flooding: a comparative interurban study of environmental injustice. *GeoJournal*, **80**, pp.145-158.
- Grundström, M. & Pleijel, H. 2014. Limited effect of urban tree vegetation on NO₂ and O₃ concentrations near a traffic route. *Environmental Pollution*, **189**, pp.73–76.
- Guerreiro, C.B.B., Foltescu, V. & de Leeuw, F. 2014. Air quality status and trends in Europe. *Atmospheric Environment*, **98**, pp.376-384.
- Guerry, A.D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G.C., Griffin, R., Ruckelhaus, M., Bateman, I.J., Duraiappah, A., Elmqvist, T., Feldman, M.W., Folke, C., Hoekstra, J., Kareiva, P.M., Keeler, B.L., Li, S., McKenzie, E., Ouyang, Z., Reyers, B., Ricketts, T.H., Rockström, J., Tallis, H. & Vira, B. 2015. Natural capital and ecosystem services informing decisions: From promise to practice, *PNAS*. **112** (24), pp.7348-7355.

- Haase, D., Larondelle, N., Andersson, E., Artmann, M., Borgström, S., Breuste, J., Gomez-Baggethun, E., Gren, A., Hamstead, Z., Hansen, R., Kabisch, N., Kremer, P., Langemeyer, J., Rall, E.L., McPhearson, T. 2014. A Quantitative Review of Urban Ecosystem Service Assessments: Concepts, Models, and Implementation. *Ambio*, **43**(4), pp.413-433.
- Haase, D., Kabisch, S., Haase, A., Andersson, E., Banzhaf, E., Baro, F., Brenck, M., Fischer, L.K., Frantzeskaki, N., Kabisch, N., Krellenberg, K., Kremer, P., Kronenberg, J., Larondelle, N., Mathey, J., Pauleit, S., Ring, I. Rink, D., Schwarz, N. & Wolff, M. 2017. Greening cities – To be socially inclusive? About the alleged paradox of society and ecology in cities. *Habitat International*, **64**, pp.41-48.
- Haines, M.M. et al. 2001. Chronic aircraft noise exposure, stress responses, mental health and cognitive performance in school children. *Psychological Medicine*, **31**, pp.265-277.
- Haines-Young, R. H. and Potschin, M. B. 2010 The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D. and Frid, C., eds., *Ecosystem Ecology: A New Synthesis*, Cambridge: CUP
- Haines-Young, R., Potschin, M. & Kienast, F. 2012 Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. *Ecological Indicators*, **21**, 39-53.
- Haines-Young, R. & Potschin, M. 2013. *Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August-December 2012*. EEA Framework Contract No EEA/IEA/09/003. [Online]. Nottingham. [Accessed 10/06/2014]. Available from: www.cices.eu.
- Haining R.P. 2003. *Spatial Data Analysis: Theory and Practice*. Cambridge: Cambridge University Press.
- Halcrow 2011. *Allan Water Natural Flood Management Techniques and Scoping Study*. Document RO1 Version 2. [Online]. Edinburgh: Halcrow. [03/03/19] Available from: http://www.cress.stir.ac.uk/allanwater/documents/R001.Allan_Water_NFM.pdf
- Halkidi, M., Batistakis, Y., & Vazirgiannis, M. 2001. On clustering validation techniques. *Journal of Intelligent Information Systems*, **17**(2-3), pp.107-145.
- Hall, J.M., Handley, J.F. & Ennos, A.R. 2012. The potential of tree planting to climate-proof high density residential areas in Manchester, UK. *Landscape and Urban Planning*, **104**, pp.410-417.
- Hamann, M, Biggs, R and Reyers, B. (2016) An Exploration of Human Well-Being Bundles as Identifiers of Ecosystem Service Use Patterns. *PLoS ONE* **11**(10). e0163476.
- Hamel, P. Bryant, B.P. 2017. Uncertainty assessment in ecosystem service analysis: Seven challenges and practical responses. *Ecosystem Services*. **24**, pp.1-15.
- Hand, K.L. & Doick, K.J. 2018 *i-Tree Eco Evaluation Part 1: Literature Review*. [Online]. March 2018. [04/04/2019]. Available from: <https://www.forestresearch.gov.uk/research/i-tree-eco/evaluating-role-i-tree-eco-protecting-and-expanding-urban-forest/>
- Handley, J., Pauleit, S., Slinn, P., Barber, A., Baker, M., Jones, C., Lindley, S. 2003. Accessible Natural Green Space Standards in Towns and Cities: A Review and Toolkit for their Implementation. *English Nature Research Reports Number 526*. English Nature, Peterborough, UK.
- Harper, S., Ruder, E., Roman, H.A., Geggel, A., Nweke, O., Payne-Sturges, D. & Levy, J.I. 2013. Using Inequality Measures to Incorporate Environmental Justice into Regulatory Analyses. *International Journal of Environmental Research and Public Health*, **109**, pp.4039-4059.

- Harrison, P. A., Berry, P. M., Simpson, G., Haslett, J. R., Blicharska, M., Bucur, M., Dunford, R., Egoh, B., Garcia-Llorente, M., Geamănă, Geertsema, W., Lommelen, E., Meiresonne, L. & Turkelboom, F. 2014. Linkages between biodiversity attributes and ecosystem services: A systematic review. *Ecosystem Services*, **9**, pp.191-203.
- Harrison, P.A., Dunford, R., Barton, D.N., Kelemen, E., Martin-Lopez, B., Norton, L., Termansen, M., Saarikoski, H., Hendriks, K., Gomez-Baggethun, E., Czucz, B., Garcia-Llorente, M., Howard, D., Jacobs, S., Karlsen, M., Kopperoinen, L., Madsen, A., Rusch, G...Zulian, G. 2018. Selecting methods for ecosystem service assessment: A decision tree approach. *Ecosystem Services*, **29C**, pp.481–498.
- Hartig, T., Mitchell, R., de Vries, S. & Frumkin, H. 2014. Nature and Health. Annual Review of Public Health, **35**, pp. 207-228.
- Hauck, J., Görg, C., Varjopuro, R., Ratamáki, O., Maes, J., Wittmer, H. & Jax, K. 2013. “Maps have an air of authority”: Potential benefits and challenges of ecosystem service maps at different levels of decision making. *Ecosystem Services*, **4**, pp.25-32.
- HC 1833. *Report from the select committee on pubic walks; with the minutes of evidence takem before them. Ordered by The House of Commons to be printed 27 June 1833.* [Online]. ProQuest Information and Learning Company. [23/04/2019] Available from: <https://parlipapers.proquest.com/parlipapers>.
- Hegetschweiler, K.T., de Vries, S., Amberger, A., Bell, S., Brennan, M., Siter, N., Olafsson, A.S., Voigt, A. and Hunziker, M. 2017 Linking demand and supply factors of urban green infrastructures: A review of European studies. *Urban Forestry & Urban Greening*, **21**, pp.48-59.
- Hein, L., van Koppen, K., de Groot, R. S. and van Ierland, E. C. 2006. Spatial scales, stakeholders and the valuation of ecosystem services, *Ecological Economics*, **57**(2), pp.209-228.
- Heisler, G. H. et al. 2007. Empirical modeling and mapping of below-canopy air 505 temperatures in Baltimore, MD and vicinity. In: *Proceedings of Seventh Urban Environment 506 Symposium*, sponsored by American Meteorological Society, 2007 San Diego, CA.
- Hernández-Morcillo, M., Plieninger, T. & Bieling, C. 2013. An empirical review of cultural ecosystem service indicators. *Ecological Indicators*, **29**, pp.434-444.
- Higgs, G. & Langford, M. 2009. GIScience, environmental justice, & estimating populations at risk: The case of landfills in Wales. *Applied Geography*, **29**, pp.63-76.
- Higgs, G. & White, S. 2000. Alternatives to census-based indicators of social disadvantage in rural communities. *Progress in Planning*, **53**, pp.1 – 81
- HMG. 2005. *Securing the future – delivering UK sustainable development strategy*. London: HMSO
- HMG 2015. *National Security Strategy and Strategic Defence and Security Review 2015: A Secure and Prosperous United Kingdom*. London: HM Stationary Office.
- HMG 2016. *National Flood Resilience Review. September 2016.* [online]. [19/04/2019] Available from: <https://www.gov.uk/government/publications/national-flood-resilience-review>
- HMG 2018. *A Green Future: Our 25 Year Plan to Improve the Environment*. London: DEFRA. [Online]. [Accessed 21/06/2019]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/693158/25-year-environment-plan.pdf
- HM Treasury. 2018. *The Green Book: Appraisal and Evaluation in Central Government*. London: The Stationery Office.

- Hodder, K.H., Newton, A.C., Cantarello, E. & Perrella, L. 2014. Does landscape-scale conservation management enhance the provision of ecosystem services? *International Journal of Biodiversity Science, Ecosystem Services & Management*, **10**(1), pp. 71-83.
- Hoffmann, E., Barros, H. & Ribeiro, A.I. 2017. Socioeconomic Inequalities in Green Space Quality and Accessibility – Evidence from a Southern European City. *International Journal Environmental Research in Public Health*, **148**, p.916.
- Holifield, R. 2012. Environmental Justice as Recognition and Participation in Risk Assessment: Negotiating and Translating Health Risk at a Superfund Site in Indian Country. *Annals of the Association of American Geographers*, **102**, pp.591-613.
- Holland, R.A., Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Thomas, C.D. & Gaston, K.J. 2011. The influence of temporal variation on relationships between ecosystem services. *Biodiversity Conservation*, **20**, pp.3285-3294.
- Holt, A.R., Mears, M., Maltby, L. & Warren, P. 2015 Understanding spatial patterns in the production of multiple urban ecosystem services. *Ecosystem Services*, **16**, pp.33-46.
- Holt, A. & Rouquette, J. 2018. Understanding natural capital in practice. Natural Capital Solutions. [Online]. [Accessed 21/06/2019]. Available from: <http://www.naturalcapitalsolutions.co.uk/wp-content/uploads/2018/10/Peel-Natural-Capital-In-Practice.pdf>
- Hölzinger, O. et al. 2014. City-wide Ecosystem Assessments - Lessons from Birmingham. *Ecosystem Services*, **9**, pp.98-105.
- Hooper, J. 2015. A destination too far? Modelling destination accessibility and distance decay in tourism, *GeoJournal*, **80**, 33-46.
- Hornigold, K., Lake, I & Dolman, P. 2016 Recreational Use of Countryside: No Evidence that High Nature Value Enhances a Key Ecosystem Service. *PLoS ONE*, **11**(11).
- Hou, Y., Burkhard, B. & Müller, F. 2013. Uncertainties in landscape analysis and ecosystem service assessment. *Journal of Environmental Management*, **127**, pp.S117–S131.
- Houlden, V., Weich, S., Porto de Albuquerque, J., Jarvis, S.A. & Rees, K. 2018. *The relationship between greenspace and the mental wellbeing of adults : a systematic review*. *PLoS One*, **13** (9), e0203000.
- Howe, C., Suich, H., Vira, B. & Mace, G.M. 2014. Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change*, **28**, 263-275.
- Horwitz, P. & Finlayson, C. M. 2011. Wetlands as Settings for Human Health: Incorporating Ecosystem Services and Health Impact Assessment into Water Resource Management. *BioScience*, **61**.
- Hubert, L., & Arabie, P. 1985. Comparing Partitions. *Journal of Classification*, **2**(2-3), pp.193-218.
- Huby, M., Cinderby, S., White, P. & de Bruin, A. 2009. Measuring inequality in rural England: the effects of changing spatial resolution. *Environment and Planning A*, **41**, pp.3023-3037.
- Humpel, N. et al. 2002. Environmental factors associated with adults' participation in physical activity. *American Journal of Preventive Medicine*, **22**(3), pp.188–199.
- Ibes, D. C. 2015. A multi-dimensional classification and equity analysis of an urban park system: A novel methodology and case study application. *Landscape and Urban Planning*, **143**, 9-10.
- iCASP 2019. *Yorkshire Integrated Catchment Solutions Programme*. [Online]. [Accessed 16/06/2019]. Available from: <https://icasp.org.uk/>.

- IPBES 2019 *IPBES Global Assessment Preview*. [Online]. [Accessed 21/06/2019]. Available from: <https://www.ipbes.net/news/ipbes-global-assessment-preview>.
- IPBES. No date. *What is IPBES?* [Online]. [Accessed 22/06/2019] Available from: <https://www.ipbes.net/about>
- Ising, H. & Kruppa, B. 2004. Health effects caused by noise: Evidence in the literature from the past 25 years. *Noise & Health*, **6**, pp.5-13.
- Islam, S.N & Winkel, J. 2017. Climate Change and Social Inequality. DESA Working Paper No. 152. New York: UN Department of Economic and Social Affairs.
- Jackson, D. L. 2000 *Guidance on the interpretation of the Biodiversity Broad Habitat Classification (terrestrial and freshwater types): Definitions and the relations with other classifications*. JNCC Report 307
- Jackson, L.E., Daniel, J., McCorkle, B., Sears, A. & Bush, K.F. 2013. Linking ecosystem services and human health: the Eco-Health Relationship Browser. *International Journal of Public Health*. **58** (5), pp.747-755.
- James, P., Banay, R.F., Hart, J.E. & Laden, F. 2015. A Review of the Health Benefits of Greenness. *Current Epidemiology Reports*. **22**, pp.131-142.
- Jansson, A.M. 1984. *Integration of Economy and Ecology: An Outlook for the Eighties*. Stockholm: University of Stockholm.
- Jarman, B. 1983 Identification of underprivileged areas. *British Medical Journal*, pp.1705-1709.
- Jax, K., Furman, E., Saarikoski, H., Barton, D. N., Dick, J., Delbaere, B., Duke, G., Görg, C., Gomez-Baggethun, E., Harrison, P., Maes, J., Perez-Soba, M., Turkelboom, F., Saarela, S.-R., Van Dijk, J. & Watt, A. D. (2017). Handling a messy world: Lessons learned when trying to make the ecosystem services concept operational. *Ecosystem Services*, **29** C.
- Jenerette, G. D., Harlan, S. L., Stefanov, W. L., & Martin, C. A. (2011). Ecosystem services and urban heat riskscape moderation: water, green spaces, and social inequality in Phoenix, USA. *Ecological Applications*, **21**(7), pp.2637-2651.
- Jennings, V. & Bamkole, O. 2019. The Relationship between Social Cohesion and Urban Green Space: An Avenue for Health Promotion. *International Journal of Environmental Research and Public Health*. **16**(3), pp.452.
- Jennings, V., Larson, L. & Yun, J. 2016. Advancing Sustainability through Urban Green Space: Cultural Ecosystem Services, Equity, and Social Determinants of Health. *International Journal of Environmental Research & Public Health*, **13**, pp.196.
- Jephcote, C. & Chen, H. 2012. Environmental injustices of children's exposure to air pollution from road-transport within the model British multicultural city of Leicester: 2000-09. *Science of the Total Environment*, **414**, pp.140-151.
- Jim, C.Y. & Chen, W.Y. 2008. Assessing the ecosystem service of air pollutant removal by urban trees in Guangzhou (China). *Journal of Environmental Management*. **88**, pp.665-676.
- Jobsvoigt, N., Watson, V. & Kenter, J. 2014. Looking below the surface: The cultural ecosystem service values of UK marine protected areas MPAs. *Ecosystem Services*, **10**, pp.97-110.
- Jones, A., Hillsdon, M. & Coombes, E. 2009. Greenspace access, use, and physical activity: Understanding the effects of area deprivation. *Preventive Medicine*, **49**(6), 500-505.
- Jones, L., Norton, L., Austin, Z., Browne, A. L., Donovan, D., Emmett, B. A., Grabowski, Z.J., Howard, D.C., Jones, J.P.G., Kenter, J.O., Manley, W., Robinson, D.A., Short, C., Siriwardena, G.M., Stevens, C.J., Storkey, J., Waters, R.D., Willis, G. F. 2016. Stocks and flows of natural and human-derived capital in ecosystem services. *Land Use Policy*, **52**, pp.51-162.

- Kaczynski, A.T. Beseny, G.M., Stani, S.A.W., Kooshari, M.J., Oestman, K.B., Bergstrom, R., Potwarka, L.R & Reis, R.S. 2014 Are park proximity and park features related to park use and park-based physical activity among adults? Variations by multiple socio-demographic characteristics. *International Journal of Behavioural Nutrition and Physical Activity*, **11**, p.146.
- Kakwani, N., Wagstaff, A. & van Doorslaer, E. 1997. Socioeconomic inequalities in health: Measurement, computation, and statistical inference. *Journal of Econometrics*, **77**, pp.87-103.
- Kandziora, M., Burkhard, B., Müller, F. 2013. Mapping provisioning ecosystem services at the local scale using data of varying spatial and temporal resolution. *Ecosystem Services*, **4**, pp.47-59.
- Kaplan, R. & Kaplan, S. 1989. *The Experience of Nature*, New York, Cambridge University Press.
- Kaźmierczak, A. & Cavan, G. 2011 Surface water flooding risk to urban communities: Analysis of vulnerability, hazard and exposure. *Landscape and Urban Planning*, **103** (2), pp.185-197.
- Kienast, F., Bolliger, J., Potschin, M., de Groot, R., Verburg, P.H., Heller, I., Wascher, D., Haines-Young, R. 2009. Assessing landscape functions with broad-scale environmental data: insights gained from a prototype development for Europe. *Environmental Management*, **44**, pp.1099-1120.
- Kienast, F., Degenhardt, B., Weilenmann, B., Wäger, Y. & Buchecker, M. (2012) GIS-assisted mapping of landscape suitability for nearby recreation. *Landscape and Urban Planning*, **105**, pp.385-399.
- Kim, J. & Nicholls, S. (2016) Influence of the measurement of distance on assessment of recreation access. *Leisure Sciences*, **38**(2), pp.118-139.
- King, R.T. 1966. Wildlife and man. *New York Conservationist*, **20**(6), pp.8-11.
- Koppen, G., Sang, A.O. & Tveut, M.S. (2014) Managing the potential for outdoor recreation: Adequate mapping and measuring of accessibility to urban recreational landscapes. *Urban Forestry & Urban Greening*, **13**, pp.71-83.
- Kremer, P., Hamstead, Z., Haase, D., McPhearson, T., Frantzeskaki, N., Andersson, E., Kabisch, N., Larondelle, N., Rall, E.L., Voigt, A., Baró, F., Bertram, C., Gómez-Baggethun, E., Hansen, R., Kaczorowska, A., Kain, J.-H., Kronenberg, J., Langemeyer, J., Pauleit, S., Rehdanz, K., Schewenius, M., van Ham, C., Wurster, D. & Elmqvist, T. 2016. Key insights for the future of urban ecosystem services research. *Ecology and Society*. **212**, p.29.
- Krieg, E. J. & Faber, D. R. 2004. Not so Black and White: environmental justice and cumulative impact assessments. *Environmental Impact Assessment Review*, **24**, pp.667-694.
- Krieger, A. M., & Green, P. E. 1999. A cautionary note on using internal cross validation to select the number of clusters. *Psychometrika*, **64**(3), pp.341-353.
- Kuhlicke, C., Scolobig, A., Tapsell, S., Steinführer, A. & De Marchi, B. 2011. Contextualizing social vulnerability: findings from case studies across Europe. *Natural Hazards*, **58**(2), pp.789-810.
- Lakerveld, R. P., Lele, S., Crane, T. A., Fortuin, K. P. J., & Springate-Baginski, O. 2015. The social distribution of provisioning forest ecosystem services: Evidence and insights from Odisha, India. *Ecosystem Services*. **14**, pp.56-66.
- Landers, D. and Nahlik, A. 2013. *Final Ecosystem Goods and Services Classification System (FEGS-CS)*, Washington, D.C.: U.S. Environmental Protection Agency.
- La Notte, A., D'amato, D., Mäkinen, H., Paracchini, M.L., Liqueste, C., Egoh, B., Geneletti, D. & Crossman, N. 2017. Ecosystem services classification: A systems ecology perspective of the cascade framework. *Ecological Indicators*, **74**, pp. 392-402.

- Lankia, T., Kopperoinen, L., Pouta, E. & Neuvonen, M. 2015. Valuing recreational ecosystem service flow in Finland. *Journal of Outdoor and Tourism*, **10**, pp.14-28.
- Larondelle, N., Haase, D. & Kabisch, N. 2014. Mapping the diversity of regulating ecosystem services in European cities. *Global Environmental Change*, **26**, pp.119-129.
- Larondelle, N. & Lauf, S. 2016. Balancing demand and supply of multiple urban ecosystem services on different spatial scales. *Ecosystem Services*, **22**, pp.18-31.
- La Rosa, D., Takatori, C., Shimizu, H. & Privitera, R. 2018. A planning framework to evaluate demands and preferences by different social groups for accessibility to urban greenspaces. *Sustainable Cities and Society*, **36**, pp.346-362.
- Laterra, P., Orúe, M.E. & Booman, G.C. 2012. Spatial complexity and ecosystem services in rural landscapes. *Agriculture, Ecosystems & Environment*, **154**, pp.56-67.
- Laterra, P., Naheulhual, L. & Vallejos, M. Berrouet Cadavid, L.M., Arroyo Pérez, E., Enrico, L., Jiménez-Sierra, C., Mejia, K., Meli, P., Rincón Ruiz, A., Salas, D., Spirić, J., Camilo Villegas, J., Villegas-Palacio, C. 2019. Linking inequalities and ecosystem services in Latin America. *Ecosystem Services*, **36**, 100875
- Lautenbach, S., Kugel, C., Lausch, A., Seppelt, R. 2011. Analysis of historic changes in regional ecosystem service provisioning using land use data. *Ecological Indicators*, **11**, pp. 676-687
- LCC 2016. *Leeds – The Compassionate City: Tackling Inequalities. Equality Progress 2015-2016*. Leeds City Council. [Online]. [Accessed 21/06/2019]. Available from:
<https://www.leeds.gov.uk/docs/Annual%20Report%202016%20Final%20version.pdf>
- Lebart, L., Morineau, A., & Piron, M. 2000. *Statistique Exploratoire Multidimensionnelle*. Paris: Dunod.
- Leeds Health & Wellbeing Board 2016. *Health & Wellbeing Strategy 2016 – 2021*. [Online] [Accessed 29/05/2019]. Available from:
<https://www.leeds.gov.uk/docs/Health%20and%20Wellbeing%202016-2021.pdf>
- Lehmann, I., Martin, A. & Fisher, J. 2018. Why should Ecosystem Services Be Governed to Support Poverty Alleviation? Philosophical Perspectives on Positions in the Empirical Literature. *Ecological Economics*, **149**, pp.265-273.
- Lele, S., O. Springate-Baginski, R. Lakerveld, D. Deb, and P. Dash. 2013. Ecosystem services: origins, contributions, pitfalls, and alternatives. *Conservation and Society*, **11**, pp.343-358.
- Lelieveld, J., Evans, J.S., Fnais, M., Giannadaki, D., Pozzer, A. 2015. The contribution of outdoor air pollution sources to premature mortality on a global scale. *Nature*, **525**, pp.367–371.
- Li, H. & Liu, Y. 2016. Neighborhood socioeconomic disadvantage and urban public green spaces availability: A localized modelling approach to inform land use policy. *Land Use Policy*, **57**, pp.470-478.
- Liquete, C. et al. 2013. Current Status and Future Prospects for the Assessment of Marine and Coastal Ecosystem Services: A Systematic Review, *Plos One*, **8**.
- Little, P. 1977 Deposition of 2.75, 5.0, and 8.5 μm particles to plant and soil surfaces, *Environmental Pollution*, **12**, pp.293-305.
- Liu, W., Chen, W. & Dong, C. 2017. Spatial decay of recreational services of urban parks: characteristics and influencing factors. *Urban Forestry and Urban Greening*, **25**, pp.130-138.
- Liu, J., Cao, Z., Zou, S., Liu, H., Hai, X., Wang, S., Duan, J., Xi, B., Yan, G., Zhang, S. & Jia, Z. 2018. An investigation of the leaf retention capacity, efficiency and

- mechanism for atmospheric particulate matter of five greening tree species in Beijing, China. *Science of the Total Environment*, **616-617**, pp.417-426.
- Lotfi, S. & Koohsari, M. J. 2009. Measuring objective accessibility to neighborhood facilities in the city (A case study: Zone 6 in Tehran, Iran). *Cities*, **26**(3), pp.133–140.
- Lovell, S.T. & Taylor, J.R. 2013. Supplying urban ecosystem services through multifunctional green infrastructure in the United States. *Landscape Ecology*, **28**, pp.1447-1463.
- Lucas, K. et al. 2004. *Environment and Social Justice: Rapid Research and Evidence Review*. Policy Studies Institute.
- Luisetti, T., Turner, R.K., Jickells, T., Elliot, M., Schaafsma, M., Beaumont, N., Malcolm, S., Burdon, D., Adams, C. & Watts, W. 2014. Coastal zone ecosystem services: From science to values and decision making; a case study. *Science of the Total Environment*, **493**, pp. 682-693
- MA 2005. *Millenium Ecosystem Assessment: A Framework for Assessment*. Washington DC.
- Mace, G. M., Hails, R. S., Cryle, P., Harlow, J. and Clarke, S. J. .2015. Towards a risk register for natural capital. *Journal of Applied Ecology*, **52**(3), pp.641-653.
- Macedo, J. & Haddad, M.A. 2016 Equitable distribution of open space: Using spatial analysis to evaluate urban parks in Curitiba, Brazil. *Environment and Planning B: Planning and Design*, **43**(6), pp.1096-1117.
- Meadows, D. 1998. *Indicators and Information Systems for Sustainable Development*. Hartland: The Sustainability Institute.
- Maes, J., Braat, L.C., Jax, K., Hutchins, M., Furman, E., Termansen, M., Luque, S., Paracchini, M.L., Chauvin, C., Williams, R., Volk, M., Lautenbach, S., Kopperoinen, L., Schelhaas, M., Weinert, J., Goossen, C.M., Dumont, E., Strauch, M., Görg, C., Dor-mann, C., Katwinkel, M., Zulian, G., Varjopuro, R., Ratamäki, O., Hauck, J., Forsius, M., Hengeveld, G.M., Perez-Soba, M., Bouraoui, F., Scholz, M., Schulz-Zunkel, C., Lepistö, A., Polishchuck, Y., Bidoglio, G. 2011 *A Spatial Assessment of Ecosystem Services in Europe: Methods, Case Studies and Policy Analysis – Phase 1 PEER Report No 3*. Ispra: Partnership for European Environmental Research, p.143.
- Maguire, K. & Sheriff, G. 2011. Comparing Distributions of Environmental Outcomes for Regulatory Environmental Justice Analysis. *International Journal of Environmental Research and Public Health*. **85**, pp.1707-1726.
- Maantay, J. 2002. Mapping environmental injustices: Pitfalls and potential of geographic information systems in assessing environmental health and equity. *Environmental Health Perspectives*, **110**, pp.161-171
- Maas, J., Verheij, R.A., Groenewegen, P.P., de Vries, S. & Spreeuwenberg, P. 2006. Green space, urbanity, and health: how strong is the relation? *Journal of Epidemiology & Community Health*, **60** (7), pp.587-592.
- Manes, F., Marando, F., Capotorti, G., Blasi, C., Salvatori, E., Fusaro, L., Ciancarella, L., Mircea, M., Marchetti, M., Chirici, G. & Munafò, M. 2016 Regulating Ecosystem Services of forests in ten Italian Metropolitan Cities: Air quality improvement by PM₁₀ and O₃ removal. *Ecological Indicators*, **67**, pp.425-440.
- Maragno, D., Gaglio, M., Robbi, M., Appiotti, F., Fano, E.A., Gissi, E. 2018 Fine-scale analysis of urban flooding reduction from green infrastructure: An ecosystem services approach for the management of water flows. *Ecological Modelling*, **386**, pp.1-10.
- Marmot, M. 2010. *Fair Society, Healthy Lives: The Marmot Review. Strategic Review of Health Inequalities in England Post-2010*. London: University College London.
- Marsh, G. F. 1874. *The Earth as Modified by Human Action*. New York: Arno.

- Martin, A., Gross-Camp, N., Kebede, B., McGuire, S., & Munyarukaza, J. 2014. Whose environmental justice? Exploring local and global perspectives in a payments for ecosystem services scheme in Rwanda. *Geoforum*, **54**, pp.167-177.
- Martinez-Harms, M. J. & Balvanera, P. 2012. Methods for mapping ecosystem service supply: a review. *International Journal of Biodiversity Science Ecosystem Services & Management*, **8**, pp.17-25.
- Maskell, L.C., Crowe, A., Dunbar, M.J., Emmett, B., Henrys, P., Keith, A.M., Norton, L.R., Scholefield, P., Clark, D.B., Simpson, I.C. & Smart. 2013. Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity. *Journal of Applied Ecology*, **50**, pp.561-571.
- McDermott, M., Mahanty, S., & Schreckenberg, K. 2013. Examining equity: A multidimensional framework for assessing equity in payments for ecosystem services. *Environmental Science & Policy*, **33**, pp.416-427.
- McDonald, A., Clarke, M., Boden, P. & Kay, D. 2010. Social Justice and Water. In: Hester, R. E. & Harrison, R. M. (eds.) *Sustainable Water*. Royal Society of Chemistry.
- McInnes, R. J. 2013. Recognizing ecosystem services from wetlands of international importance: an example from Sussex, UK. *Wetlands*, **33**(6), pp.1001-1017.
- McKercher, B. 1996. Differences between tourism and recreation in parks. *Annals of Tourism Research*, **23**(3), pp.563-575.
- McLeod, H. et al. 2000. The relationship between socio-economic indicators and air pollution in England and Wales: implications for environmental justice. *Regional Environmental Change*, **1**, pp.78-85.
- Maguire, K. & Sheriff, G. 2011. Comparing distributions of environmental outcomes for regulatory environmental justice analysis. *International Journal of Environmental Research and Public Health*, **8**(5), pp.1707-1726.
- Mears, M., Brindley, P., Maheswaran, R., Jorgensen, A. 2019. Understanding the socioeconomic equity of publicly accessible greenspace distribution: The example of Sheffield, UK. *Geoforum*, In Press.
- Menne, B. & Murray, V. 2013. *Floods in the WHO European region: Health effects and their prevention*. World Health Organization Europe. Denmark.
- Met Office 2016 Further rainfall and flooding across north of the UK [online]. [05/04/19] Available from:
https://www.metoffice.gov.uk/climate/uk/interesting/december2015_further
- MHCLG 2018. *National Planning Policy Framework: Equality Impact Assessment* [online]. London: Ministry of Housing Communities & Local Government [Last accessed 29/05/2019]. Available from:
<https://www.gov.uk/government/consultations/draft-revised-national-planning-policy-framework>.
- Miao, Q., Chen, D., Buzzelli, M. and Aronson, K .J. 2015. Environmental equity research: review with focus on outdoor air pollution research methods and analytic tools Arch. *Environment & Occupational Health*, **70**, pp.47–55
- Milcu, A.I., Hanspach, J., Abson, D. & Fischer, J. 2013 Cultural ecosystem services: a literature review and prospects for future research. *Ecology and Society*, **18**(3), pp.44.
- Mitchell, G. & Dorling, D. 2003. An environmental justice analysis of British air quality. *Environment and Planning A*, **35**, pp.909-929.
- Mitchell, G. & Norman, P. 2012. Longitudinal environmental justice analysis: Co-evolution of environmental quality and deprivation in England, 1960-2007. *Geoforum*, **43**(1), pp.44-57.
- Mitchell, G., Norman, P. & Mullin, K. 2015. Who benefits from environmental policy? An environmental justice analysis of air quality change in Britain, 2001-2011. *Environmental Research Letters*, **10**, 105009.

- Mitchell, G. & Walker, G. 2007. Methodological Issues in the Assessment of Environmental Equity and Environmental Justice. In Deakin et al. eds. *Sustainable Urban Development: The Environmental Evaluation Methods*. Oxon: Routledge, 1-19.
- Mohai, P., Pellow, D. & Roberts, J.T. 2009. Environmental Justice. *Annual Review of Environment and Resources*, **34**, pp.405-430.
- Molteno, S., Morris, J. & O'Brien, L. 2012. *Public access to woodlands and forests: a rapid evidence review* [online]. Surrey: Forest Research. [03/03/19] Available from <https://webarchive.nationalarchives.gov.uk/20131001200852/http://www.defra.gov.uk/forestrypanel/files/Public-Access-RER1.pdf>
- Moreno-Jiménez, A., Cañada-Torrecilla, R., Vidal-Domínguez, M.J., Palacios-García, A. & Martínez-Suárez, P. 2016. Assessing environmental justice through potential exposure to air pollution: A socio-spatial analysis in Madrid and Barcelona, Spain. *Geoforum*, **69**, pp.117-131.
- Morris, D.G. & Flavin, R.W. 1990. A digital terrain model for hydrology. *Proceedings 4th International Symposium on Spatial Data Handling, Jul 23-27, Zurich*, **1**, 250-262.
- Morris, J., O'Brien, E., Ambrose-Oji, B., Lawrence, A., Carter, C., & Peace, A. 2011. Access for all? Barriers to accessing woodlands and forests in Britain. *Local Environment*, **16**(4), pp.375-396.
- Morse, S., Vogiatzakis, I., & Griffiths, G. 2011. Space and Sustainability. Potential for Landscape as a Spatial Unit for Assessing Sustainability. *Sustainable Development*, **19**(1), pp.30-48.
- Mullin, K., Mitchell, G., Nawaz, N.R. & Waters, R. 2018. Natural capital and the poor in England: Towards an environmental justice analysis of ecosystem services in a high income country. *Landscape and Urban Planning*, **176**. pp.10-21.
- Nahuelhual, L., Benra, F., Laterra, P., Marin, S., Arriagada, R., Julian, C. 2018. Patterns of ecosystem services supply across farm properties: Implications for ecosystem services-based policy incentives. *Science of the Total Environment*, **634**, pp.941-950.
- Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., Ricketts, T.H., 2008. Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences*, **105** (28), pp.9495–9500.
- National Parks and Access to Countryside Act 1949 (c. 97, 12, 13 & 14 Geo 6). London: HMSO.
- Natural Capital Coalition. No date. *What is Natural Capital?* [Online]. [Accessed 20/06/2019]. Available from: <https://naturalcapitalcoalition.org/natural-capital-2/>
- Natural England 2014. *Natural Character Areas* [online]. [Accessed 16/06/2019] Available from: <http://publications.naturalengland.org.uk/category/587130>
- Natural England 2015. *Monitor of engagement with the natural environment*. Natural England Joint Report JP009. [Accessed online 10/06/2015]. Available from: <https://www.gov.uk/government/collections/monitorof-engagement-with-the-natural-environmentsurvey-purpose-and-results>
- Natural England 2018a. *Monitor of Engagement with the Natural Environment: The national survey on people and the natural environment. Headline report 2018: Analysis of latest results (March 2017 to February 2018) and nine years of the survey from 2009 to 2018*. [online] [03/03/19]. Available from <https://www.gov.uk/government/collections/monitor-of-engagement-with-the-natural-environment-survey-purpose-and-results>
- Natural England 2018b. *Monitor of Engagement with the Natural Environment: The national survey on people and the natural environment. Headline Report*

- Accompanying Data*. [online] [03/03/19]. Available from <https://www.gov.uk/government/collections/monitor-of-engagement-with-the-natural-environment-survey-purpose-and-results>
- NCC. 2013. *The State of Natural Capital: Towards a framework for measurement and valuation*. Retrieved from - https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/516707/ncc-state-natural-capital-first-report.pdf
- NCC 2015. *The State of Natural Capital: Protecting and Improving Natural Capital for Prosperity and Wellbeing. Third report to the Economic Affairs Committee*.
- NCC 2019. *What is natural capital?* [Online]. [Accessed 29/04/2019]. Available from: <https://naturalcapitalcoalition.org/natural-capital-2/>
- Nedkov, S., Burkhard, B. 2012. Flood regulating ecosystem services – mapping supply and demand in the Etropole Municipality. Bulgaria. *Ecological Indicators*, **21**, pp.67–79
- NERC (CEH) 2011. Land Cover Map 2007 Dataset documentation, Version 1.0. [Online]. Wallingford: CEH. [03/03/19]. Available from: <https://www.ceh.ac.uk/sites/default/files/LCM2007%20dataset%20documentation.pdf>
- Nesbitt, L., Meitner, M.J., Girling, C.L., Sheppard, S.R.J. & Lu, Y. 2019. Who has access to urban vegetation? A spatial analysis of distributional green equity in 10 US cities. *Landscape and Urban Planning*, **181**, pp.51-79.
- Newton, A.C., Hodder, K., Cantarello, Perrella, L., Birch, J.C., Robins, J., Douglas, S., Moody, C. & Cordingley, J. 2012. Cost-benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. *Journal of Applied Ecology*, **49**, pp. 571-580.
- Ng, C.F. 2000. Effects of building construction noise on residents: A quasi-experiment. *Journal of Environmental Psychology*, **20**, pp.375-85.
- NHS 2014. Protecting health and reducing harm from extreme heat and heatwaves. *Heatwave Plan for England*. Department of Health.
- Nicholls, S. 2001 Measuring the accessibility and equity of public parks: A case study using GIS. *Managing Leisure*, **6**(4), pp.201–219.
- Nicholls, S., & Shafer, C. S. 2001 Measuring accessibility and equity in a local park system: The utility of geospatial technologies to park and recreation professionals. *Journal of Park and Recreation Administration*, **19**(4), pp.102–124.
- Nicholson, E., Mace, G.M., Armsworth, P.R., Atkinson, G., Buckle, S., Clements, T., Ewers, R.M., Fa, J.E., Gardner, T.A., Gibbons, J., Grenyer, R., Metcalfe, R., Mourato, S., Muûls, M., Osborn, D., Reuman, D.C., Watson, C. & Milner-Gulland, E.J. 2009. Priority research areas for ecosystem services in a changing world. *Journal of Applied Ecology*. **46**(6).
- Nielson, T. S. & Hansen, K. B. 2007. Do green areas affect health? Results from a Danish survey on the use of green areas and health indicators. *Health & Place*, **13**, pp.839-850.
- Ning, Z.H., Chambers, R. & Abdollahi, K. 2015. Modeling air pollutant removal, carbon storage, and CO2 sequestration potential of urban forests in Scotlandville, Louisiana, USA. *iForest: Biogeosciences and Forestry*, **9**, 860-867.
- Nordh, H. and Evensen, K.H. 2018. Qualities and functions ascribed to urban cemeteries across the capital cities of Scandinavia. *Urban Forestry & Urban Greening*, **33**, pp.80-91.
- Norman, P. 2010 Identifying change over time in small area socio-economic deprivation. *Applied Spatial Analysis and Policy*, **3**(2-3), pp.107-138.

- Norton, L., Greene, S., Scholefield, P. and Dunbar, M. 2016 The importance of scale in the development of ecosystem service indicators?, *Ecological Indicators*, **61**, 130-140.
- Northamptonshire Local Nature Partnership, n.d. *Health and Wellbeing in the Environment: Natural Environmental Solutions for a Healthier Northamptonshire* [Online]. [Last accessed 29/05/2019]. Available from: <https://ecosystemsknowledge.net/resources/national-initiatives/LNP>.
- Nowak, D.J., Crane, D.E. 2000. The Urban Forest Effects (UFORE) Model: quantifying urban forest structure and functions. In: Hansen M., Burk T. (Eds.), *Integrated tools for natural resources inventories in the 21st century: proceedings of the IUFRO conference. General Technical. Reports NC-212*, St. Paul, 714–720.
- Nowak, D. J., Crane, D.E., Stevens, J.C. 2006. Air pollution removal by urban firtrees and shrub in the United States. *Urban Forestry & Urban Greening*, **4**, pp.115-123.
- Nowak, D.J., Hoehn, R.E., Crane, D.E., Stevens, J.C., Walton, J.T. 2007a. *Assessing Urban Forest Effects and Values: New York City's Urban Forest. Northern Resource Bulletin NRS-9*. USDA Forest Service, Newtown Square, PA, p. 24.
- Nowak, D.J., Hoehn, R.E., Crane, D.E., Stevens, J.C., Walton, J.T. 2007b. *Assessing Urban Forest Effects and Values: Philadelphia's Urban Forest. Northern Resource Bulletin NRS-7*. USDA Forest Service, Newtown Square, PA, p. 24.
- Nowak, D.J., Hirabayashi, S., Bodine, A. & Hoehn, R. 2013. Modeled PM_{2.5} removal by tree in ten U.S. cities and associated health effects. *Environmental Pollution*, **178**, pp.395-402.
- Nowak, D. J., Hirabayashi, S., Ellis, E. & Greenfield, E.J. 2014. Tree and forest effects on air quality and human health in the United States. *Environmental Pollution*, **193**, pp.119-129.
- O'Brien, L., & Morris, J. 2014. Well-being for all? The social distribution of benefits gained from woodlands and forests in Britain. *Local Environment*, **19**(4), pp.356-383.
- Odum, E.P. & Odum, H.T. 1972. Natural areas as necessary components of man's total environment. *Transactions of the North American Wildlife and Natural Resources Conference*, Washington DC. **37**, pp.178-189.
- Oh, K. & Jeong, S. 2007. Assessing the spatial distribution of urban parks using GIS. *Landscape and Urban Planning*, **82**(1), pp.25-32.
- ONS. 2009. Understanding patterns of deprivation *Regional Trends*, 41, pp. 93-114.
- ONS. No date. *Census Geography*. [Online]. [Accessed 23/06/2019]. Available from: <https://www.ons.gov.uk/methodology/geography/ukgeographies/censusgeography>.
- ONS. 2015a. *Natural capital accounting 2020 roadmap: Interim review and forward look*. Retrieved from - <http://webarchive.nationalarchives.gov.uk/20160106130009/http://www.ons.gov.uk>
- ONS. 2015b. *Sustainable Development Indicators*. Retrieved from - <http://www.ons.gov.uk/peoplepopulationandcommunity/wellbeing/articles/sustainabledevelopmentindicators/2015-07-13>
- ONS 2017. *Principles of Natural Capital Accounting*. Retrieved from - <https://www.ons.gov.uk/economy/nationalaccounts/uksectoraccounts/methodologies/naturalcapital>
- Openshaw S. 1984. *The Modifiable Areal Unit Problem*. Norwich: Geobooks.
- Ostrom, E. 2009. A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science*. **325** (5939), pp.419-422.

- Ouyang, Z., Zheng, H., Xiao, Y., Polasky, S., Liu, J., Xu, W., Wang, Q., Zhang, L., Xiao, Y., Rao, E., Jiang, L., Lu, F., Wang, X., Yang, G., Gong, S., Wu, B., Zeng, Yang, W. & Daily, G.C. 2016. Improvements in ecosystem services from investments in natural capital. *Science*, **352**(6292), pp.1455-1459.
- Páez, A., Scott, D.M. & Morency, C. 2012. Measuring accessibility: positive and normative implementations of various accessibility indicators. *Journal of Transport Geography*, **25**, pp.141-153.
- Palomo, I., Felipe-Lucia, M. R., Bennett, E. M., Martin-Lopez, B. and Pascual, U. 2016. Disentangling the Pathways and Effects of Ecosystem Service Co-Production, *Ecosystem Services: From Biodiversity to Society, Pt 2*, **54**, pp.245-283.
- Paracchini, M.L., Zulian, G., Kopperoinen, L., Maes, J., Schägner, J.P., Ter-mansen, M., Zandersen, M., Perez-Soba, M., Scholefield, P.A. & Bidoglio, G. 2014. Mapping cultural ecosystem services: a framework to assess the potential for outdoor recreation across the EU. *Ecological Indicators*, **45**, pp.371–385.
- Partelow, S., & Winkler, K.J. 2016. Interlinking ecosystem services and Ostrom's framework through orientation in sustainability research. *Ecology and Society*, **21**(3), p.27.
- Pascual, U., Muradian, R., Rodriguez, L. C., & Duraiappah, A. (2010). Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. *Ecological Economics*, **69**(6), pp.1237-1244.
- Pastor, M. et al. 2001. Which came first? Toxic facilities, minority move-in, and environmental justice. *Journal of Urban Affairs*, **23**, pp.1-21.
- Patterson, R., Risby, A. & Chan, M-Y. 2012. Consumption of takeaway and fast food in a deprived inner London Borough: are they associated with childhood obesity? *BMJ Open*, **2**, e000402.
- Pearce, D. 2002. An Intellectual History of Environmental Economics. *Annual Review of Energy and the Environment*, **27**, pp.57-81.
- Pearce, J., & Kingham, S. 2008. Environmental inequalities in New Zealand: A national study of air pollution and environmental justice. *Geoforum*, **39**(2), pp.980-993.
- Pearce, J.R., Richardson, E.A., Mitchell, R.J. and Shortt, N.K. 2010. Environmental justice and health: the implications of the socio-spatial distribution of multiple environmental deprivation for health inequalities in the United Kingdom. *Transactions of the Institute of British Geographers*, **354**, pp.522-539.
- Pennine Prospects 2019. *South Pennine Projects & Programmes*. [Online]. [Last Accessed 16/06/2019]. Available from: <https://www.pennineprospects.co.uk/projects-programmes>
- Perrings, C., Folke, C. & Mäler, K.G. 1992. The ecology and economics of biodiversity loss: the research agenda. *Ambio*, **21**, pp.201-211.
- Perry, T. & Nawaz, R. 2008. An investigation into the extents and impacts of hard surfacing of domestic gardens in an area of Leeds, United Kingdom. *Landscape and Urban Planning*, **86**(1), pp.1-13.
- Petersen, J. E. and Gocheva, K. 2015. *EU reference document on Natural Capital Accounting, Revised draft for consultation, 6 January 2015*, EU MAES.
- Pham, T. T. H., Apparicio, P., Seguin, A. M., Landry, S., & Gagnon, M. (2012). Spatial distribution of vegetation in Montreal: An uneven distribution or environmental inequity? *Landscape and Urban Planning*, **107**(3), 214-224.
- PHE 2014. *From evidence into action: opportunities to protect and improve the nation's health*. London: Public Health England.
- PHE 2018. Health profile for England: 2018. [Online]. [Accessed 21/06/2019]. Available from: <https://www.gov.uk/government/publications/health-profile-for-england-2018>.
- Pigou, A. 1920. *The Economics of Welfare*. London: MacMillan.

- Pinoncelly, V. 2016. *Poverty, place and inequality: Why place-based approaches are key to tackling poverty and inequality*: Royal Town Planning Institute.
- Pitt, M. 2007 *Learning Lessons from the 2007 Floods: An Independent Review by Sir Michael Pitt (The Pitt Review)*. [Online]. U.K. Government, London.[19/04/2019] Available from: <https://webarchive.nationalarchives.gov.uk/20100812084907/http://archive.cabinetoffice.gov.uk/pittreview>
- Plieninger, T., Dijks, S., Oteros-Rozas, E., Bieling, C. 2013. Assessing, mapping, and quantifying cultural ecosystem services at community level. *Land Use Policy*. **33**, pp.118–129.
- Plummer, M.L. 2009. Assessing benefit transfer for the valuation of ecosystem services. *Frontiers in Ecology and the Environment*, **7**, pp. 38-45
- Pope, C. A. et al. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *Journal of the American Medical Association*. **287**, pp.1132-1141.
- Pope, R., Wu, J. & Boone, C. 2016. Spatial patterns of air pollutants and social groups: a distributive environmental justice study in the phoenix metropolitan region of USA. *Environmental Management*. **58**(5), pp.753-766.
- Portman, M.E. 2013. Ecosystem services in practice: Challenges to real world implementation of ecosystem services across multiple landscapes – A critical review. *Applied Geography*. **45**, 185-192.
- Posthumus, H., Rouquette, J.R., Morris, J., Gowing, D.J.G. & Hess, T.M. 2010. A framework for the assessment of ecosystem goods and services; a case study on lowland floodplains in England. *Ecological Economics*. **69**, pp.1510-1523.
- Potschin, M. B. & Haines-Young, R. H. 2011. Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography*. **35**, pp.575-594.
- Potschin-Young, M.B., Haines-Young, R.H. Görg, C., Heink, U., Jax, K. & Schleyer, C. 2018. Understanding the role of conceptual frameworks: Reading the ecosystem service cascade. *Ecosystem Services*. **29C**, pp.428-440.
- Powe, N. A. & Willis, K. G. 2004. Mortality and morbidity benefits of air pollution (SO₂ and 571 PM₁₀) absorption attributable to woodland in Britain. *Journal of Environmental Management*. **70**, pp.119-128.
- Power, A.G. 2010. Ecosystems and agriculture: tradeoffs and synergies. *Philosophical Transactions of the Royal Society of London B Biological Science*, **365**(1554), 2959-2971.
- Pugh, T.A.M., MacKenzie, A.R., Whyatt, J.D. & Hewitt, C.N. 2012. Effectiveness of Green Infrastructure for Improvement of Air Quality in Urban Street Canyons. *Environmental Science and Technology*. **46**, 7692-7699.
- Purcell, A. T. et al. 1994. Preference or preferences for landscape? *Journal of Environmental Psychology*. **14**, pp.195-209.
- Pye, S., Stedman, J., Adams, M. & King, K. 2001. *Further analysis of NO₂ and PM₁₀ air pollution and social deprivation, A report produced for DEFRA, The National Assembly for Wales and The Northern Ireland Department of the Environment*. Report AEAT/ENV/R/0865 AEA Technology Environment.
- Qui, L., Lindberg, S. & Nielsen, A.B. 2013. Is biodiversity attractive? On-site perception of recreational and biodiversity values in urban green space. *Landscape and Urban Planning*. **119**, 136-146.
- Quijas, S., Jackson, L.E., Maass, M., Schmid, B., Raffaelli, D. & Balvanera, P. 2012. Plant diversity and generation of ecosystem services at the landscape scale: expert knowledge assessment. *Journal of Applied Ecology*. **49**(4), 929-940.
- Radford, K.G. & James, P. 2013. Changes in the value of ecosystem services along a rural-urban gradient: A case study of Greater Manchester, UK. *Landscape and Urban Planning*. **1091**, pp.117-127.

- Randall, K. et al. 1990. Effects of plantscapes in an office environment on worker satisfaction. In: Relf, D. (ed.) *The role of horticulture in human well-being and social development: a national symposium, 19–21 April 1990*, Arlington, Virginia: Portland.
- Raudsepp-Hearne, C, Peterson, G.D. and Bennett, E.M. 2010 *Ecosystem service bundles for analyzing tradeoffs in diverse landscapes*. *PNAS*. **107**(11), pp.5242-5247.
- Raudsepp-Hearne, C. & Peterson, G.D. 2016. Scale and ecosystem services: how do observation, management, and analysis shift with scale – lessons from Québec. *Ecology and Society*. **21**(3), p.16.
- Raum, S. 2018. A framework for integrating systematic stakeholder analysis in ecosystem services research: Stakeholder mapping for forest ecosystem service in the UK. *Ecosystem Services*. 29A, 170-184.
- Raymond, C. M., Gottwald, S., Kuoppa, J., & Kytta, M. 2016. Integrating multiple elements of environmental justice into urban blue space planning using public participation geographic information systems. *Landscape and Urban Planning*. **153**, 198-208.
- Regecova, V. & Kellerova, E. 1995. Effects of urban noise pollution on blood pressure and heart rate in preschool children. *Journal of Hypertension*, **13**, pp.405-412
- Reyers, B., Biggs, R., Cumming, G.S., Hejnowicz, A.P. & Polasky, S. 2013 Getting the measure of ecosystem services: a social–ecological approach. *Frontiers in Ecology and the Environment*. **11**(5), pp. 268-273
- Richards, D.R. & Friess, D.A. 2015. A rapid indicator of cultural ecosystem service usage at a fine spatial scale: Content analysis of social media photographs. *Ecological Indicators*. **53**, pp.187-195.
- Rieb, J.T, Chaplin-Kramer, R., Daily, G.C., Armsworth, P.R., Bohning-Gaese, K., Bonn, A., Cumming, G.S., Eigenbrod, F., Grimm, V., Jackson, B.M., Marques, A., Pattanayak, S.K., Pereira, H.M., Peterson, G.D., Ricketts, T.H., Robinson, B.E., Schroter, M., Schulte, L.A., Riechers, M., Barkmann, J. & Tscharntke, T. 2016. Perceptions of cultural ecosystem services from urban green. *Ecosystem Services*. **17**, pp.33-39.
- Rivas, B., Kumar, P. & Hagen-Zanker, A. 2017. Exposure to air pollutants during commuting in London: Are there inequalities among different socio-economic groups? *Environmental International*. **101**, 143-157.
- Robinson, W. S. 1950. "Ecological Correlations and the Behavior of Individuals". *American Sociological Review*. **15**(3): 351–357.
- Rogers, K., Jarratt, T. and Hansford, D. 2011. *Torbay's Urban Forest. Assessing urban forest effects and values. A report on the findings from the UK i-Tree Eco pilot project*. Exeter: Treeconomics.
- Rook, G.A. 2013. Regulation of the immune system by biodiversity from the natural environment: An ecosystem service essential to health. *PNAS*. **110**(46), pp. 18360-18367.
- Rossi, S.D., Byrne, J.A., Pickering, C.M. 2015. The role of distance in peri-urban national park use: Who visits them and how far do they travel? *Applied Geography*. **63**. 77-88.
- Rouquette, J. 2016. *Mapping Natural Capital and Ecosystem Services in the Nene Valley* [Online]. [Last accessed 29/05/2019]. Available from <https://ecosystemsknowledge.net/sites/default/files/wp-content/uploads/EKNuploads/Natural-Capital-Ecosystem-Services-in-the-Nene-Valley-Summary-Report.pdf>.
- Rova, S. and Pranovi, F. 2017. Analysis and management of multiple ecosystem services within a social-ecological context. *Ecological Indicators*. **72**, 436-443.

- Rufat, S., Tate, E., Burton, C.G. & Maroof, A.S. 2015. Social vulnerability to floods: Review of case studies and implications for measurement. *International Journal of Disaster Risk Reduction*, **14**(4), p. 470-486.
- Sæbø, A., Popek, R., Nawrot, B., Hanslin, H.M., Gawronska, H. & Gawronska, S.W. 2012. Plant species differences in particulate matter accumulation on leaf surfaces. *Science of the Total Environment*. **427-428**, 347-354.
- Salonen, M. & Toivonen, T. 2013. Modelling travel time in urban networks: comparable measures for private car and public transport. *Journal of Transport Geography*. **31**, 143-153.
- Sandifer, P.A., Sutton-Grier, A.E. & Ward, B.P. 2015. Exploring connections among nature, biodiversity, ecosystem services, and human health and wellbeing: Opportunities to enhance health and biodiversity conservation. *Ecosystem Services*. **12**, pp. 1-15.
- Schipperijn, J., Ekholm, O., Stigsdotter, U.K., Toftager, M., Bentsen, P., Kamper-Jorgensen, F. & Randrup, T.B. 2010. Factors influencing the use of green space: Results from a Danish national representative survey. *Landscape and Urban Planning*. **95**(3), pp.130-137.
- Schleyer, C., Görg, C., Huack, J. & Winkler, K.J. 2015. Opportunities and challenges for mainstreaming the ecosystem services concept in the multi-level policy-making within the EU. *Ecosystem Services*. **16**, pp.174-181.
- Schroeder, H. W. & Green, T. L. 1985. Public preferences for tree density in municipal parks. *Journal of Arboriculture*. **11**, pp.272-277.
- Schröter, M., Albert, C., Marques, A., Tobon, W., Lavorel, S., Maes, J., Brown, C., Klotz, S. & Bonn, A. 2016. National Ecosystem Assessments in Europe: A Review. *Bioscience*. **66**(10), 813-828.
- Schröter, M., Stumpf, K.H., Loos, J., van Oudenhoven, A.P.E., Bohnke-Henrichs, A., Abson, D.J. 2017. Refocusing ecosystem services towards sustainability. *Ecosystem Services*. **25**, pp.35-43.
- Schumacher, E.F. 1973. *Small is beautiful: Economics as if People Mattered*. London: Blond and Briggs, London.
- Schulp, C.J.E., Burkhard, B., Maes, J., Van Vliet, J. & Verburg, P.H. 2014. Uncertainties in Ecosystem Maps: A Comparison on the European Scale. *PLoS ONE*. **9**(10), e109643.
- Schüle, S.A., Gabriel, K.M.A. & Bolte, G. 2017. Relationship between neighbourhood socioeconomic position and neighbourhood public green space availability: An environmental inequality analysis in a large German city applying generalized linear models. *Environmental Health*. **220**(4), pp.711-718.
- Schüle, S.A., Hiltz, L.K., Dreger, S. & Bolte, G. 2019. Social Inequalities in Environmental Resources of Green and Blue Spaces: A Review of Evidence in the WHO European Region. *International Journal of Environmental Research and Public Health*. **16**, p.1216.
- Seeland, K. & Nicolè, S. 2006. Public green space and disabled users. *Urban Forestry & Urban Greening*. **5**(1), pp.29-34.
- Selmi, W., Weber, C., Rivière, E., Blond, N., Mehdi, L. & Nowak, D. 2016. Air pollution removal by trees in public green spaces in Strasbourg city, France. *Urban Forestry & Urban Greening*, **17**, pp.192-201.
- Senatsverwaltung für Stadtentwicklung und Umwelt 2013. Availability of Public, Near Residential Green Space [Online]. [19/03/2019] Available from: www.stadtentwicklung.berlin.de/umwelt/umweltatlas/eda605_01.htm.
- Seppelt, R., Dormann, C.F., Eppink, F.V., Lautenbach, S. & Schmidt, S. 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. *Journal of Applied Ecology*. **48**, pp.630-636.

- Seppelt, R., Turner, M.G. & Bennett, E.M. 2017. When, where, and how nature matters for ecosystem services: challenges for the next generation of ecosystem service models. *BioScience*. 67(9), pp.820–833
- Serna-Chavez, H. M. et al. 2014. A quantitative framework for assessing spatial flows of ecosystem services. *Ecological Indicators*, 39, pp.24-33.
- Shägner, J. P. et al. 2013. Mapping ecosystem services' values: Current practice and future prospects. *Ecosystem Services*. 4, pp.33-46.
- Sharp, R., Tallis, H.T., Ricketts, T., Guerry, A.D., Wood, S.A., Chaplin-Kramer, R., Nelson, E., Ennaanay, D., Wolny, S., Olwero, N., Vigerstol, K., Pennington, D., Mendoza, G., Aukema, J., Foster, J., Forrest, J., Cameron, D., Arkema, K., Lonsdorf, E., Kennedy, C., Verutes, G., Kim, C.K., Guannel, G., Papenfus, M., Toft, J., Marsik, M., Bernhardt, J., Griffin, R., Glowinski, K., Chaumont, N., Perelman, A., Lacayo, M. Mandle, L., Hamel, P., Vogl, A.L., Rogers, L., Bierbower, W., Denu, D., and Douglass, J. 2018. *InVEST 3.7. User's Guide*. The Natural Capital Project, Stanford University, University of Minnesota, The Nature Conservancy, and World Wildlife Fund. [Online]. [Accessed 23/06/2019]. Available from: <http://releases.naturalcapitalproject.org/invest-userguide/latest/>
- Sharps, K., Masante, D., Thomas, A., Jackson, B., Redhead, J., May, L., Prosser, H., Cosby, B., Emmett, B. & Jones, L. 2017. Comparing strengths and weaknesses of three ecosystem services modelling tools in a diverse UK river catchment. *Science of the Total Environment*. 584-585, pp. 118-130.
- Shashua-Bar, L. and Hoffman, M. 2000. Vegetation as a Climatic Component in the Design of an Urban Street: An Empirical Model for Predicting the Cooling Effect of Urban Green Areas with Trees. *Energy and Buildings*. 31, pp.221-235.
- Shucksmith, M. 2012. Class, Power and Inequality in Rural Areas: Beyond Social Exclusion? *Sociologia Ruralis*, 52(4), pp.377-397.
- Sikor, T. 2013 Introduction: Linking Ecosystem Services with Environmental Justice. In Sikor, T. ed. *The Justices and Injustices of Ecosystem Services*. Abingdon: Routledge, pp.1-18.
- Sister, C., Wolch, J. & Wilson, J. 2010. Got green? Addressing environmental justice in park provision. *GeoJournal*. 75, pp.229-248.
- Sjöman, J.D. & Gill, S.E. 2014. Residential runoff – The role of spatial density and surface cover, with a case study in the Højeå river catchment, southern Sweden. *Urban Forestry & Urban Greening*, 13, pp.304-314.
- Smith, W.H. 1981. *Air Pollution and Forests: Interactions Between Air Contaminants and Forest Ecosystems*. Springer-Verlag, New York, NY.
- Smith, D.P. & Philips, D.A. 2001. Socio-cultural representations of greentrified Pennine rurality. *Journal of Rural Studies*. 17, pp. 457-469.
- Smith, T., Noble, M., Noble, S., Wright, G., McLennan, D. & Plunkett, E. 2015. *The English Indices of Deprivation Technical Report*. London: DCLG.
- Song, C., Lee, W.-K., Choi, H.-A., Kim, J., Jeon, S.W. & Kim, J.S. (2016) Spatial assessment of ecosystem functions and services for air purification of forests in South Korea. *Environmental Science & Policy*. 63, pp.27-34.
- Soto, J. R., Escobedo, F. J., Adams, D. C., & Blanco, G. 2016. A distributional analysis of the socio-ecological and economic determinants of forest carbon stocks. *Environmental Science & Policy*, 60, pp.28-37.
- Spake, R., Bellamy, C., Graham, L.J., Watts, K., Wilson, T., Norton, L.R., Wood, C.M., Schmucki, R., Bullock, J.M. & Eigenbrod, F. 2019. An analytical framework for spatially targeted management of natural capital. *Nature Sustainability*. 2, pp.90-97.
- Spangenberg, J.H., von Haaren, C., Settele, J. 2014. The ecosystem service cascade: further developing the metaphor. Integrating societal processes to

- accommodate social processes and planning, and the case of bioenergy. *Ecological Economics*. **104**, pp. 22-32
- Speak, A.F., Mizgajski, A., Borysiak, J. 2015. Allotment gardens and parks: Provision of ecosystems with an emphasis on biodiversity. *Urban Forestry & Urban Greening*. **14**(4), pp.772-781.
- Speirs, L. J. 2003. Sustainable planning: the value of green space. In: Beriatoss, E. et al. (eds.) *Sustainable Planning and Development*.
- Spinney, J. E. L. & Millward, H. 2013. Investigating travel thresholds for sports and recreation activities. *Environment and Planning B: Planning and Design*, **40**, pp.474-488.
- Sport England 2016. *Active People Survey 10: October 2015-September 2016. Once a week participation in funded sports amongst people aged 16 years and over*. [Online] [03/03/19] Available from <https://www.sportengland.org>
- Stansfield, S. et al. 2000. *Work related factors and ill health: The Whitehall II Study*. Contract Research Report 266/2000. London: The Stationery Office.
- Stevens, G. A., Dias, R. H., & Ezzati, M. 2008. The effects of 3 environmental risks on mortality disparities across Mexican communities. *Proceedings of the National Academy of Sciences*, **105**(44), pp.16860-16865.
- Stevenson, S. et al. 1998. Examining the inequality and inequity of car ownership and the effects of pollution and health outcomes such as respiratory disease. *Epidemiology*. **9**, pp.S29
- Stürck, J., Poortinga, A., Verburg, P.H. 2014. Mapping ecosystem services: the supply and demand of flood regulation services in Europe. *Ecological Indicators*. **38**, pp.198–211.
- Su, J.G., Morello-Frosch, R., Jesdale, B.M., Kyle, A.D., Shamasunder, B. & Jerrett, M. 2009. An Index for Assessing Demographic Inequalities in Cumulative Environmental hazards with Application to Los Angeles, California. *Environment, Science & Technology*. **43**(20), pp.7626-7634.
- Suckall, N., Fraser, E.D.G., Cooper, T. & Quinn, C. 2009. Visitor perceptions of rural landscapes: A case study in the Peak District National Park, England. *Journal of Environmental Management*. **90**(2), pp.1195-1203.
- Suich, H., Howe, C. & Mace, G. 2015. Ecosystem services and poverty alleviation: a review of the empirical links. *Ecosystem Services*. **12**, pp.137-147.
- Swensen, G., Nordh, H. & Brendalmo, J. 2016. A green space between life and death? A case study of activities in Gamiebyen Cemetery in Oslo, Norway. *Norsk Geografisk Tidsskrift-Norwegian. Journal of Geography*, **70**(1), 41-53.
- Syrbe, R.U. & Walz, U. 2012. Spatial indicators for the assessment of ecosystem services. Providing, benefiting and connecting areas and landscape metrics. *Ecological Indicators*, **21**, pp.80-88.
- Takano, T. et al. Urban residential environments and senior citizens' longevity in megacity areas: the importance of walkable green spaces. *Journal of Epidemiology and Community Health*, **56**, pp.913-918.
- Talen, E. 1998. Visualizing fairness: Equity maps for planners. *Journal of the American Planning Association*, **64**, pp.22–38.
- Tallis, H., Kareiva, P., Marvier, M. & Chang, A. 2008. An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences*. **105**(28), pp. 9457-9464
- Tallis, M., Taylor, G., Sinnett, D. & Freer-Smith, P. 2011. Estimating the removal of atmospheric particulate pollution by the urban tree canopy of London, under current and future environments. *Landscape and Urban Planning*, **103**(2), pp.129-138.

- Tapsell, S.M., Penning-Rowsell, E.C., Tunstall, S.M. & Wilson, T.L. 2002. Vulnerability to flooding: health and social dimensions. *Philosophical Transactions of the Royal Society London A*, **360**, p.1511-1525.
- Taylor, A. F. 2009. Children With Attention Deficits Concentrate Better After Walk In The Park *Journal of Attention Disorders*, **12**, pp.402-409.
- TEEB 2010. The Economics of Ecosystems and Biodiversity for Business – Executive Summary. [Online]. [Last Accessed 16/06/2019]. Available from <http://www.teebweb.org/our-publications/teeb-study-reports/business-and-enterprise/>
- The Air Quality Standards Regulations 2010. S.I. 2008/301. London: The Stationary Office.
- The President. 1994. Executive Order 12898 of February 11, 1994: *Federal Actions To Address Environmental Justice in Minority Populations and Low-Income Populations*.
- Thomas, H. & Nisbet, T.R. 2016. Slowing the flow in Pickering: Quantifying the effect of catchment woodland planting on flooding using the soil conservation service curve number method. *International Journal of Safety and Security Engineering*, **6**(3), pp.466-474.
- Tiwary, A Sinnett, D., Peachey, C., Chalabi, Z., Vardoulakis, S., Fletcher, T., Leonardi, G., Grundy, C., Azapagic, A. and Hutchings, T. R. 2009. An integrated tool to assess the role of 596 new plantings in PM10 capture and the human health benefits: A case study in London. *Environmental Pollution*, **157**, pp.2645-2653.
- Tiwary, A., Williams, I.D., Heidrich, O., Namdeo, A., Bandaru, V. & Calfapietra, C. 2016. Development of multi-functional streetscape green infrastructure using a performance index approach. *Environmental Pollution*, **208**, pp.209-220.
- Tobler W., 1970. A computer movie simulating urban growth in the Detroit region. *Economic Geography*. **46**(S), pp.234–240.
- Tonne, C., Mila, C., Fecht, D., Alvarez, M., Gulliver, J., Smith, J., Beevers, S., Anderson, H.R., Kelly, F. 2018. Socioeconomic and ethnic inequalities in exposure to air and noise pollution in London. *Environment International*. **115**, pp.170-179.
- Townsend, P. 1987. Deprivation. *Journal of Social Policy*. **16**, pp.125-146.
- Tratalos, J., Fuller, R.A., Warren, P.H., Davies, R.G. & Gaston, K.J. (2007) Urban form, biodiversity potential and ecosystem services. *Landscape and Urban Planning*, **83**, pp.308-317
- Treeconomics London 2015. *Valuing London's Urban Forest: Results of the London i-Tree Eco Project*. London: Treeconomics London.
- Troy, A and Wilson, M.A. 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. *Ecological Economics*, **60**, pp.435– 49.
- Turner, K.G., Vestergaard, Odgaard, M., Bocher, P.K., Dalgaard, T. & Svenning, J-C. 2014. Bundling ecosystem services in Denmark: Trade-offs and synergies in a cultural landscape. *Landscape and Urban Planning*. **125**, pp.89-104.
- Twohig-Bennett, C. & Jones, A. 2018. The health benefits of the great outdoors: A systematic review and meta-analysis of greenspace exposure and health outcomes. *Environmental Research*. **166**, pp.628-637.
- UKNEA. 2011. *UK National Ecosystem Assessment Technical Report*. Retrieved from - <http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx>
- UKNEA. 2014. *UK National Ecosystem Assessment Follow-on Phase: Synthesis of Key Findings*. Retrieved from - <http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx>

- Ulrich, R.S. View through a window may influence recovery from surgery. *Science*, **224**, pp.420-422.
- UN 2017. *Sustainable Development Goals*. [Online]. [Accessed 21/06/2019]. Available from: <https://www.un.org/sustainabledevelopment/sustainable-development-goals/>
- UN 1992b. *Rio Declaration on Environment and Development: Report of the United Nations Conference on the Human Environment*. Stockholm, 5-16 June 1972.
- UN 2015. *Transforming our world: the 2030 Agenda for Sustainable Development*. (25 September 2015). [Online]. A/70/L.I. [Accessed 21/06/2019]. Available from: <https://sustainabledevelopment.un.org/>
- UNECE 1998 *Convention on access to information, public participation in decision making and access to justice in environmental matters*. Adopted on 25 June 1998. Entered into force on 30 October 2001. [Online]. [Accessed 22/06/2019]. Available from: <http://ec.europa.eu/environment/aarhus/index.htm>
- UNECE 2017 *Declaration of the sixth ministerial conference on environment and health*. EURO/Ostrava2017/6. 15th June 2017. Ostrava, Czech Republic. [Online]. [Accessed 20/09/2019]. Available from: http://www.euro.who.int/__data/assets/pdf_file/0007/341944/OstravaDeclaration_SIGNED.pdf?ua=1
- United Church of Christ. 1987. *Toxic wastes and race in the United States: A national report on the racial and socio-economic characteristics with hazardous waste sites*. New York: United Church of Christ, Commission for Racial Justice.
- University of Essex. Institute for Social and Economic Research, NatCen Social Research, Kantar Public. 2018. *Understanding Society: Waves 1-8, 2009-2017 and Harmonised BHPS: Waves 1-18, 1991-2009*. [data collection]. 11th Edition. UK Data Service. SN: 6614.
- USDA-NRCS 1986. *Urban hydrology for small watersheds*. TR55. [Online] United States Department of Agriculture Natural Resources Conservation Service-Conservation Engineering Division.[11/03/2019] Available from: www.nrcs.usda.gov.
- US EPA. no date. *Learn About Environmental Justice*. [Online]. [Accessed 21/06/2019]. Available from: <https://www.epa.gov/environmentaljustice/learn-about-environmental-justice>.
- Van den Berg, M., Wendel-Vos, W., van Poppel, M., Kemper, H., van Mechelen, W. & Maas, J. 2015. Health benefits of green spaces in the living environment: A systematic review of epidemiological studies. *Urban Forestry & Urban Greening*, **14**(4), pp.806-816.
- Van der Biest, K. et al. 2015. Evaluation of the accuracy of land-use based ecosystem service assessments for different thematic resolutions. *Journal of Environmental Management*, **156**, pp.41-51.
- Van Der Horst, D. & Toke, D. 2010. Exploring the landscape of wind farm developments; local area characteristics and planning process outcomes in rural England. *Land Use Policy*, **27**, pp.214-221.
- Van de Walle, M., M.T. Sykes, P.A. Harrison, G.W. Luck, P. Berry, R. Bugter, T.P. Dawson, C.K. Feld, et al. 2009. *Review paper on concepts of dynamic ecosystems and their services*. The Rubicode Project Rationalising Biodiversity Conservation in Dynamic Ecosystems. http://www.rubicode.net/rubicode/RUBICODE_Review_on_Ecosystem_Services.pdf.
- Van Herzele, A., De Clercq, E. M., & Wiedemann, T. 2005. Strategic planning for new woodlands in the urban periphery: through the lens of social inclusiveness. *Urban Forestry & Urban Greening*, **3**(3-4), 177-188

- Van Riper, C.J. & Kyle, G.T. 2014. Capturing multiple values of ecosystem services shaped by environmental worldviews: a spatial analysis. *Journal of environmental management*, **145**, pp.374-84.
- Verhagen, W., Kukkala, A.S., Moilanen, A., van Teeffelen, A.J.A. & Verburg, P.H. 2016. Use of demand for and spatial flow of ecosystem services to identify priority areas. *Conservation Biology*, pp.860-871.
- Verhagen, W., Kukkala, A.S., Moilanen, A., van Teeffelen, A.J.A. & Verburg, P.H. 2016. Use of demand for and spatial flow of ecosystem services to identify priority areas. *Conservation Biology*, **31**(4), pp.860-871.
- Viel, J. F., Hagi, M., Upegui, E., & Laurian, L. 2011. Environmental justice in a French industrial region: Are polluting industrial facilities equally distributed? *Health & Place*, **17**(1), pp.257-262.
- Vigerstol, K.L. & Aukema, J.E. 2011. A comparison of tools for modelling freshwater ecosystem services. *Journal of Environmental Management*. **92**, pp.2403-2409.
- Vihervaara, P., Rönkä, M. & Walls, M. 2010. Trends in Ecosystem Service Research: Early Steps and Current Drivers. *Ambio*, **394**, pp.314 – 324.
- Villa, F., Bagstad, K.J., Voigt, B., Johnson, G.W., Portela, R., Honzak, M. & Batker, D. 2014. A Methodology for Adaptable and Robust Ecosystem Services Assessment. *Plos One*, **9**(3), e91001.
- Villamagna, A.M., Angermeier, P.L., Bennett, E.M. 2013. Capacity, pressure, demand, and flow: a conceptual framework for analyzing ecosystem service provision and delivery. *Ecological Complexity*, **15**, pp.114–121.
- Vogt, J., Gillner, S., Hofmann, M., Tharang, A., Dettmann, S., Gerstenberg, T., Schmidt, C., Gebauer, H., Van de Riet, K., Berger, U. & Roloff, A. 2017. Citree: A database supporting tree selection for urban areas in temperate climate. *Landscape and Urban Planning*. **157**, pp.14-25.
- Vos, P.E.J., Maiheu, B., Vankerkom, J. & Janssen, S. 2013. Improving local air quality in cities: To tree or not to tree? *Environmental Pollution*. **183**, pp.113-122.
- Wagstaff, A., van Doorslaer, E. & Paci, P. 1991. On the measurement of inequalities in health. *Social Science and Medicine*. **33**(5), pp.545-557.
- Walker, G., Mitchell, G., Fairburn, J., & Smith, G. 2003. *Environmental Quality and Social Deprivation. Phase II: national analysis of flood hazard, IPC industries and air quality*. Bristol: Environment Agency.
- Walker, G., Fay, H. and Mitchell, G. 2005a. *Environmental justice impact assessment: An evaluation of requirements and tools for distributional analysis*. Report to Friends of the Earth, 76pp.
- Walker, G., Mitchell, G., Fairburn, J. & Smith, G. 2005b. Industrial pollution and social deprivation: evidence and complexity in evaluating and responding to environmental inequality. *Local Environment*. **10**, pp.361-377.
- Walker, G., Burningham, K., Fielding, J., Smith, G., Thrush, D. and Fay, H. 2006. *Addressing Environmental Inequalities: Flood Risk*. Bristol: Environment Agency.
- Walker G. 2009. Beyond distribution and proximity: exploring the multiple spatialities of environmental justice. *Antipode*. **41**. pp.614–36
- Walker, G. 2010. Environmental justice, impact assessment and the politics of knowledge: The implications of assessing the social distribution of environmental outcomes. *Environmental Impact Assessment Review*. **30**(5), pp.312-318.
- Walker, G. & Burningham, K. 2011. Flood risk, vulnerability and environmental justice: Evidence and evaluation of inequality in a UK context. *Critical Social Policy*. **312**, pp.216-240.

- Wallace, K. 2007. Classification of ecosystem services: Problems and solutions. *Biological Conservation*. **139**, pp.235-246.
- Wallace, K. 2008. Ecosystem services: Multiple classifications or confusion?, *Biological Conservation*. **141**(2), pp.353-354.
- Warhurst, J.R., Parks, K.E., McCulloch, L. & Hudson, M.D. 2014. Front gardens to car parks: Changes in garden permeability and effects on flood regulation. *Science of the Total Environment*. **485-486**, pp.329-339.
- Watkins, S.L. & Gerrish, E. 2018. The relationship between urban forests and race: A meta-analysis. *Journal of Environmental Management*. **209**, pp.152-168.
- WAVES. No date. Wealth Accounting and the Valuation of Ecosystem Services (WAVES): *About Us*. [Online]. [Accessed 21/06/2019]. Available from: <https://www.wavespartnership.org/en/about-us>
- Wei, H., Fan, W., Wang, X., Lu, N., Dong, X., Zhao, Y., Ya, X. & Zhao, Y. 2017. Integrating supply and social demand in ecosystem services assessment: A review. *Ecosystem Services*. **25**, pp.15-27.
- Weimann H, Rylander L, Albin M, Skarback E, Grahn P, Ostergren P-O., Bjork, J. 2015. Effects of changing exposure to neighbourhood greenness on general and mental health: A longitudinal study. *Health & Place*. **33**, pp.48–56.
- Wells, N. & Evans, G. 2003. Nearby Nature; A buffer of life stress among Rural Children. *Environment and Behavior*. **35**, pp.311-330.
- Wen, M. 2013. Spatial Disparities in the Distribution of Parks and Green Spaces in the USA. *Annals of Behavioural Medicine*. **45**(S1), pp.18-27.
- Weng, Q. 2001. Modelling Urban Growth Effects on Surface Runoff with the Integration of Remote Sensing and GIS. *Environmental Management*. **28** (6), pp.737-748.
- Westman, W., 1977. How much are nature's services worth. *Science*. **197**, pp.960–964.
- West Northamptonshire Joint Planning Unit 2014. *West Northamptonshire Joint Core Strategy Local Plan Part 1*. [online]. [Accessed 16/06/2019]. Available from: <http://www.westnorthamptonshirejpu.org/connect.ti/website/view?objectId=5130832#5130832>.
- West Yorkshire Combined Authority 2017. *Leeds City Region: Green and Blue Infrastructure Strategy 2017-2036* [Online]. [Accessed 29/05/2019]. Available from: www.aireandcalderpartnership.org.
- Weyland, F. & Larterra, P. 2014. Recreation potential assessment at large spatial scales: A method based in the ecosystem services approach and landscape metrics. *Ecological Indicators*. **39**, pp.34-43.
- Wheater, H. S. 2006. Flood hazard and management: a UK perspective. *Philosophical Transactions of the Royal Society A*. **364**, pp.2135-2145.
- Wheeler, B. W. 2004. Health-related environmental indices and environmental equity in England and Wales. *Environment and Planning A*. **36**, pp.803-822.
- White, M.P. et al. 2013. Would you be happier living in a greener urban area? A fixed-effects analysis of panel data. *Psychological Science*. **24**(6), pp.920-928.
- Whitehead, M. 2014. *Due North: The report of the Inquiry on Health Equity for the North*. Liverpool: University of Liverpool and Centre for Local Economic Strategies.
- Whitford, V., Ennos, A.R. & Handley, J.F. 2001. "City form and natural process" – indicators for the ecological performance of urban areas and their application to Merseyside, UK. *Landscape and Urban Planning*. **57**, pp.91-103.
- Whittington, D., Smith, V., Okorafor, A., Okore, A., Liu, J.L. & McPhail, A. 1992. Giving respondents time to think in contingent valuation studies: A developing country application. *Journal of Environmental Economics and Management*. **22**(3), pp.205-225.

- WHO. 1948. *Preamble to the Constitution of the World Health Organization as adopted by the International Health Conference*, New York, 19-22 June, 1946; signed on 22 July 1946 by the representatives of 61 States (Official Records of the World Health Organization, no. 2, p. 100) and entered into force on 7 April 1948.
- WHO 2006. *Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide: Summary of risk assessment*. [Online]. [05/04/2019] Available from: https://www.who.int/phe/health_topics/outdoorair/outdoorair_aqg/en/
- WHO. 2015. *Connecting Global Priorities: Biodiversity and Human Health. A State of Knowledge Review*. [Online]. [Accessed 30/07/2015]. Available from: http://apps.who.int/iris/bitstream/10665/174012/1/9789241508537_eng.pdf?ua=1
- WHO 2019. World Health Organization: Ten threats to global health in 2019. [Online]. [Accessed 10/06/2019]. Available from: <https://www.who.int/emergencies/ten-threats-to-global-health-in-2019>.
- Wilbanks, T. 2006. How Scale Matters: Some Concepts and Findings. In: Reid, W., Berkes, F., Wilbanks, T. and Capistrano, D., eds., *Bridging Scales and Knowledge Systems: Linking Global Science and Local Knowledge in Assessments*. , Washington: Island Press, pp.25-40.
- Wildlife and Countryside Act 1981. (c.69). London: HMSO.
- Wilkerson, L.M., Mitchell, M.G.E., Shanahan, D., Wilson, K.A., Ives, C.D., Lovelock, C.E. & Rhodes, J.R. 2018. The role of socio-economic factors in planning and managing urban ecosystem services. *Ecosystem Services*. **31**, pp.102-110.
- Wilkinson, R. & Pickett, K. 2009. *The Spirit Level: Why equality is better for everyone*. London: Allen Lane.
- Willemsen, L., Verburg, P.H., Hein, L., van Mensvoort, M.E.F. 2008. Spatial characterization of landscape functions. *Landscape & Urban Planning*, **88**, pp.34–43.
- Winn, J.P., Bellamy, C.C. & Fisher, T. 2018. *EcoServ-GIS: a toolkit for mapping ecosystem services*. *Scottish Natural Heritage Research Report No. 954*.
- Wilson, M. A., & Howarth, R. B. (2002). Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation. *Ecological Economics*. **41**(3), pp.431-443.
- Wolch, J. R., Byrne, J., & Newell, J. P. 2014. Urban green space, public health, and environmental justice: The challenge of making cities 'just green enough'. *Landscape and Urban Planning*. 125, pp.234-244.
- Wolff, S., Schulp, C.J.E. & Verburg, P.H. 2015. Mapping ecosystem services demand: A review of current research and future perspectives. *Ecological Indicators*. 55, pp.159-171.
- Wong, M. S., Peng, F., Zou, B., Shi, W. Z., & Wilson, G. J. 2016. Spatially Analyzing the Inequity of the Hong Kong Urban Heat Island by Socio-Demographic Characteristics. *International Journal of Environmental Research and Public Health*. **13**(3).
- Wood, D.F. 1961. The Distances-Traveled Technique for Measuring Value of Recreation Areas: An Application. *Land Economics*. 37 (4), pp363-369.
- Woodland Trust 2017 *Space for people: Targeting action for woodland access* [online]. Grantham: Woodland Trust [03/03/19] Available from <https://www.woodlandtrust.org.uk>.
- Wüstermann, H., Kalisch, D., Kolbe, J. 2016. *Towards a national indicator for urban green space provision and environmental inequalities in Germany: Method and findings*. SFB Discussion Paper 2016-022. Berlin: Humboldt-Universität zu Berlin.

- Yli-Pelkonen, V., Setälä, H. & Viippola, V. 2017. Urban forests near roads do not reduce gaseous air pollutant concentrations but have an impact on particles levels. *Landscape and Urban Planning*, 158, 39-47.
- Yok Tan, P., Wang, J., Sia, A. 2013. Perspectives on five decades of the urban greening of Singapore. *Cities*, **32**, pp.24-32.
- Xiao, Y., Wang, Z., Li, Z., & Tang, Z. 2017. An assessment of urban park access in Shanghai – Implications for the social equity in urban China. *Landscape and Urban Planning*, **157**, pp.383-393.
- Xu, Q., Zheng, X. & Zheng, M. 2019. Do urban planning policies meet sustainable urbanization goals? A scenario-based study in Beijing, China. *Science of the Total Environment*. **670**, pp. 498-507.
- Zank, B., Bagstad, K.J., Voigt, B. & Villa, F 2016. Modeling the effects of urban expansion on natural capital stocks and ecosystem service flows: A case study in the Puget Sound, Washington, USA. *Landscape and Urban Planning*, **149**, pp.31-42.
- Zeng, Z., Tang, G., Hong, Y., Zeng, C & Yang, Y. 2017. Development of an NRCS curve number global dataset using the latest geospatial remote sensing data for worldwide hydrologic applications. *Remote Sensing Letters*, **8**(6), pp.528-536.
- Zhang, L., Moran, M.D., Makar, P.A., Brook, J.R. & Gong, S. 2002 Modelling gaseous dry deposition in AURAMS: a unified regional air-quality modelling system. *Atmospheric Environment*, **36**, pp.537-560.
- Zhao, C. & Sander, H.A. 2018. Assessing the sensitivity of urban ecosystem service maps to input spatial data resolution and method choice. *Landscape and Urban Planning*, **175**, pp.11-22.
- Zsamboky, M. et al. 2011. *Impacts of climate change on disadvantaged UK coastal communities*. York: Joseph Rowtree Foundation.
- Zulian, G., Polce, C. & Maes, J. 2014 ESTIMAP: a GIS-based model to map ecosystem services in the European Union. *Annali di Botanica*, **4**, pp.1–7.

Acknowledgement of Data Sources

- Boorman, D.B., Hollis, J.M. & Lilly, A., 1995. Hydrology of soil types: a hydrologically based classification of the soils of the United Kingdom. Institute of Hydrology Report No. 126. 137pp
- CEH 2007. Soil carbon natural capital map. catalogue.ceh.ac.uk
- CEH 2014 Ecological Status. catalogue.ceh.ac.uk
- DCLG. 2015. English indices of deprivation 2015.
<https://www.gov.uk/government/statistics/english-indices-of-deprivation-2015>
- DEFRA Open Data. defra.opendata.arcgis.com.
- DfT 2017.AADF <https://www.dft.gov.uk/traffic-counts>
- Environment Agency. Risk of Flooding from Multiple Sources.
<https://data.gov.uk/dataset/c75a9cd2-9dc2-4070-acd5-6bc86d8a3de2/risk-of-flooding-from-multiple-sources-risk-contribution#licence-info>
- Environment Agency. 2013. Water Framework Directive waterbodies 2013 status.
www.geostore.com/environment-agency
- Landmap: Bluesky
<https://catalogue.ceda.ac.uk/uuid/1999cc89208158c8616845f01ac93100>
- Morris DG and Flavin RW, 1990. A digital terrain model for hydrology. Proc 4th International Symposium on Spatial Data Handling. Vol 1 Jul 23-27 Zurich,pp 250-262.Morris DG and Flavin RW, 1994. Sub-set of UK 50 m by 50 m hydrological digital terrain model grids. NERC, Institute of Hydrology, Wallingford.
- Natural England. Open Data GeoPortal www.gis.naturalengland.org.uk
- NERC (CEH) 2011. Land Cover Map 2007 Dataset documentation, Version 1.0. [Online]. Wallingford: CEH. [03/03/19]. Available from:
<https://www.ceh.ac.uk/sites/default/files/LCM2007%20dataset%20documentation.pdf>
- Office for National Statistics, 2011 Census: Aggregate data (England and Wales) [computer file]. UK Data Service Census Support. Downloaded from: <http://infuse.mimas.ac.uk> . This data is licensed under the terms of the Open Government Licence [<http://www.nationalarchives.gov.uk/doc/open-government-licence/version/2/>].
- Office for National Statistics (2011). 2011 Census: boundary data (England and Wales) UK Data Service. SN:5819 UKBORDERS: Digitised Boundary Data, <http://discover.ukdataservice.ac.uk/catalogue/?sn=5819&type=Data%20catalogue>, Retrieved from <http://census.ukdataservice.ac.uk/get-data/boundary-data.aspx>. Contains public sector information licensed under the Open Government Licence v3.
- OS 2007. OS MasterMap Integrated Transport Network Layer: User guide and technical specification v1.0. [online]. Ordnance Survey. [03/03/19] Available from:https://digimap.edina.ac.uk/webhelp/os/data_files/os_manuals/osmm_itn_userguide_v1.0.pdf
- OS 2014. Mastermap Topographic Vectors and Points of Interest.
<https://digimap.edina.ac.uk/>
- OS 2015. Meridian. OS Open Rivers. © Crown copyright and database rights 2015 Ordnance Survey (100025252). www.ordnancesurvey.co.uk
- Public Rights of Way. Accessed via <http://www.rowmaps.com/datasets/>. Contains Ordnance Survey data © Crown copyright and database right [2015]
- RSPB. 2015. UK Reserves 2015. www.rspb.org.uk
- The Woodland Trust. 2014. Accessible Woodland Database.
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