

PHD THESIS

**SPATIAL DIMENSIONS OF BIODIVERSITY
VALUES:**

**ANALYSES OF PREFERENCE HETEROGENEITY
AND CONSERVATION PRIORITIES ACROSS
NATIONAL LANDSCAPES**

TOMAS BADURA

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Abstract

This thesis contributes to our understanding of the spatial dimension of biodiversity related values in the context of changing environment. It starts with making a case for the use economic valuation for improving decision making related to environmental change and with an overview of the main concepts and approaches for doing so (Chapter 1). The thesis highlights the need to better incorporate the spatial considerations in ecosystem assessments and the importance of robust natural science that underpins any such assessments. The thesis then provides three empirical analyses, two that employ discrete choice modelling to examine how spatial information influences preferences for environmental change, and one that focuses on modelling the biodiversity impacts of land use change. The contributions of the thesis are as follows. 1) development of a novel methodology for choice experiments that incorporates space in the survey design, experimental design and presentation of choice situations on individualised maps (Chapter 3); 2) application of this methodology to test how addition of individualised maps alongside commonly used Tabular format impacts on preferences and welfare values for environmental change (Chapter 3); 3) provision of evidence that state and country borders have an impact on preferences for the portrayed changes (Chapter 2 and 3); and 4) development of prediction models that allow evaluating land use change impacts on farmland bird species; this includes assessment of model performance and variables importance for future integration of the models with economic analyses (Chapter 4). The thesis closes with research implications and personal research plans.

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INTRODUCTION

0.1 The context

The past century has led to an astonishing increase in human development in terms of population, its wellbeing and the size of the global market economy (Guerry et al. 2015). This development has had and is having a significant impact on the biosphere and functioning of the Earth systems. A new term has been coined – Anthropocene – that highlights the fact that humans have caused a new geological epoch by altering the world in a manner that will be possible to discern in the fossil records (Crutzen 2006; Dirzo et al. 2014; Monastersky 2015). The world has been assessed to be outside its safe operating space that might endanger “resilience of major components of Earth-system functioning” by already crossing three out of nine so-called planetary boundaries through significantly altering nitrogen and carbon cycles and extensively degrading world’s biodiversity (Rockström et al. 2009; Steffen et al. 2015). Biodiversity in particular – the diversity of ecosystems, species and genes - is being lost at a startling rate even to an informed observer. Recent scientific literature is not shy of claiming that the world due to human activity entered “sixth mass extinction” in earth’s history (Barnosky et al. 2011) or caused “[a]nthropocene defaunation” (Dirzo et al. 2014). It has been estimated that the “[c]urrent rates of extinction are about 1000 times the likely background rate of extinction” (Pimm et al. 2014), with human-caused extinctions most recently referred to as “biological annihilation” (Ceballos et al. 2017). The gravity of this situation is profound. Alongside the immorality of such extinction, there is the simple utilitarian relationship that biodiversity and ecosystems support human societies and economies worldwide through provision of multiple goods and services such as food, fibre, materials and

quality of life goods such as recreation, as well as being fundamental and crucial elements in natural systems and processes such as climate regulation, water and nutrient cycling and the prevention of certain natural disasters (MA 2005). It is therefore increasingly understood that biodiversity loss and related degradation of ecosystem functioning is likely to endanger both current and future human well-being (e.g. MA 2005, TEEB 2010a). This has drawn increased policy attention to the economics of ecosystems and biodiversity.

Economics, one can say, is one of the root of biosphere degradation but also could be a key player in changing these trends. Recent decades have seen an increased cooperation of economic and natural scientists in pursuit of assessment and partial (given knowledge limitations) quantification of the relationships humans have with the environment. Initial economic arguments have been made for conserving biodiversity supported by multi-disciplinary analyses (e.g. Costanza et al. 1997; Balmford et al. 2002). In the hallmark Millennium Ecosystem Assessment study (MA 2005) the cooperation between economists and natural scientists led to a development of the Ecosystem Services Framework that conceptualised the anthropocentric relationship of humans with nature. Initially coined as “the benefits people obtain from ecosystems” (MA 2005), ecosystem services have been classified as provisioning, regulating, cultural and supporting services with ongoing classification efforts continuing¹. In 2011 The Economics of Ecosystem and Biodiversity (TEEB) was officially released in Nagoya, Japan, at the 10th Conference of Parties of the Convention on Biological Diversity. The TEEB platform aimed to provide a parallel report to Stern Report (Stern 2006) by making the economic case for biodiversity protection, as the Stern report made for Climate action. TEEB provided an assessment of “the value of nature”

¹ See The Common International Classification of Ecosystem Services (CICES), <https://cices.eu/>
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across the world and why it should be and how it can be incorporated in decisions. It published number of reports targeted at multiple stakeholder groups, with reports for scientific community (TEEB 2010b), international and national policy makers (TEEB 2011), local decision makers (TEEB 2012a), business (TEEB 2012b) and targeted internet communication with public. The increased policy attention to the “value of nature” has been reflected in international policy area since, including multiple initiatives at global (e.g. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)²; but also Wealth Accounting and the Valuation of Ecosystem Services (WAVES)³), regional (e.g. Mapping and Assessment of Ecosystems and their Services (MAES)⁴) and national policy context (e.g. UK National Ecosystem Assessments, UK NEA 2011 and 2014; Natural Capital Committee (NCC)⁵ in the UK). Key publications have been published on the role of economics in ecosystem service assessments (e.g. Heal 2000; Bateman et al. 2011) and multiple guidance books have been produced that provide guidance on valuing ecosystem related goods and services (e.g. Bateman et al. 2002; Heal et al. 2005; Freeman et al. 2014).

A stock point of view of the natural world, conceptualised into natural capital, has been brought to policy attention more recently and this broadened the ecosystem services debate. In this view natural capital, consisting of biotic and abiotic assets, provides flows of ecosystem services that either by itself or with other forms of capital generates goods and services of value to people. The stock and flow view is a useful way to conceptualise our relationship with the natural world that is in line with economic theory. Natural capital research

² <https://www.ipbes.net/>

³ <https://www.wavespartnership.org/>

⁴ <https://biodiversity.europa.eu/maes>

⁵ <https://www.gov.uk/government/groups/natural-capital-committee>

has grown substantially however still lacks wide practical implementation (e.g. Guerry et al 2015).

Conceptualising the interaction between the natural and human world into stocks and flows lends itself well to accounting thinking that is yet another development in the ecosystem related debates. Indeed, the accounting research and practice together with economists and natural scientists have been recently supporting attempts to reflect the value of nature in decisions on a macro level, though national accounting practice. The System of Environmental and Economic Systems Experimental Ecosystem Accounts (UN et al. 2014) aims to provide an overview of our ability to account for natural systems alongside the economic data captured in System of National Accounts (EC et al. 2009) that we use for deriving measures of economic performance such as GDP. While still largely in development with multiple issues outstanding from a conceptual and theoretical perspective (Obst et al. 2015; Hein et al. 2016) and in trial-and-error phase, multiple countries globally are experimenting with natural capital accounting.

The above initiatives and developments in the economics of ecosystem and biodiversity is mirrored in the global policy arena. Global, EU and National Biodiversity strategies and targets call for incorporation of the value of nature into decision making and accounting systems by 2020. This reflect the growing realisation that incorporating nature more closely into decision making is likely to support broader societal goals. The relevant commitments include:

- Globally agreed Convention on Biological Diversity's Strategic Plan for Biodiversity 2011-2020 includes *Strategic Goal A: Address the underlying causes of biodiversity loss*, Target 2 as follows:

By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.

- The European Union's Biodiversity Strategy to 2020, Target 2, Action 5 requires that:

Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national levels by 2020.

The multiple developments at both micro and macro levels of economic research as well as an increased policy attention to incorporation of the values of ecosystem related goods and services into decision making is the context for this thesis.

0.2 Focus and contributions of the thesis

This thesis aims to provide a methodological and empirical contribution to the field of economics of ecosystems and biodiversity as, in a non-exhaustive way, overviewed above. It builds on my long standing interest and previous work in the area and related policy processes. This includes work focused on valuation of ecosystem related goods and services from a methodological point of view (Badura 2012), policy assessments for the European Commission and international organisations and events (ten Brink et al. 2011, ten Brink et al. 2012), national level assessments (Kettunen et al. 2013, Bateman et al. 2014) and topical assessments (Russi et al. 2013), as well as handbook contributions (ten Brink et al. 2014) and practitioner guidance books (Badura and Kettunen 2013a, 2013b), and multiple workshops and

conferences. The thesis is also informed by my work related to Natural Capital Accounting developments (Badura et al. 2013, Badura et al. 2017).

The thesis aims to contribute to the efforts to better incorporate nature in decision making processes. The first Chapter (published as Badura et al. 2016) overviews how economic valuation can improve decision making and provides context for the research presented in this thesis. The Chapter argues that monetary valuation provides information that is easily understood by all stakeholders and facilitates comparative assessments across different policy areas. It discusses the main concepts, methods and recent advances in economic valuation that allow analysts to incorporate the many ways in which the natural environment contributes to human wellbeing within decision making processes. The chapter highlights how context is important in these assessments, including spatial considerations that are still largely underplayed in the valuation research and which are the main focus of this thesis. It is argued that valuation is a highly interdisciplinary endeavour and can be only as robust as the underpinning natural science input. Despite the numerous outstanding issues that environmental economics and natural science need to understand, as well as lacking practical take up of the research, the Chapter argues that valuation is well placed to improve decision making related to environmental change.

The thesis further consists of three methodological and empirical works that aim to contribute our understanding of the role of space in valuation of ecosystem related goods and services (Chapter 2 and Chapter 3), and ecological assessment of land use change impacts on biodiversity (Chapter 4). In particular, this thesis focuses on stated preference surveys which are one of the most prevalent techniques to value ecosystem related goods and services. Most of the environmental goods and services do not have established markets

and hence it is difficult to estimate their values. Stated preference valuation surveys overcome this by developing hypothetical markets and analysing choices made by survey or experimental participants regarding the provision and associated costs of non-market goods. Reflecting the need for both economic and natural science analyses for better incorporation of the natural environment in the decisions making, the thesis also derives ecological models that could be used alongside the stated preference data. The following text outlines the main contributions the three empirical chapters make.

The first contribution and main focus of the research presented in this thesis is methodological, in incorporating spatial considerations into stated preference valuation survey (Chapter 2 and Chapter 3). The spatial variation in the provision of environmental goods and services together with variation in where people live have a profound impact on the economic benefits that nature provides to people. Until these spatial effects are sufficiently understood and incorporated in valuation research, it will be of limited use to policy. Consequently, recent years have seen an increased attention to space in stated preference valuation research, however predominantly, the focus of this research has been on the way space is taken into account in the data analysis of choice decisions. Our contribution, in contrast, lies in the way how we collect stated preference data. More specifically, a novel way is developed how we design valuation surveys and how we portray spatially relevant choices in the stated preference context. In the thesis (Chapter 3) we develop a novel approach that incorporates both physical and political dimension of space into a choice experiment. We explicitly incorporate space into both experimental design and in how choice set is presented in terms of spatial information. We developed a functionality whereby spatially explicit, personally tailored choice situations are generated that portray the environmental change.

We believe that this methodology might be of use to valuation research to elicit spatially relevant values, but also to test how presentation of spatial information effects elicited values and preference. **Our second contribution (Chapter 3) consist of applying the developed methodology to assess and compare effects has presenting choice situations in a Tabular approach, used regularly in the CE literature, against personalized maps that show respondents their location and location of intervention scenarios.** We show that the map presentation influences preferences and related Willingness To Pay values, particularly related to spatial attributes and that presenting choice situations on maps show results more in line with theoretical expectations.

The third contribution of this thesis is in documenting political aspect of space in preferences for environmental change (Chapter 2 and Chapter 3). While the research focusing on physical distance in the context of stated preference research is abundant (see Chapter 3 for an overview), only limited research to our knowledge (e.g. Dallimer et al. 2013, Rogers and Burton 2017) has examined how political boundaries such as borders might influence preference for environmental interventions. In both empirical applications, one in Australia and one in the UK, we provide evidence of such influence. We identify preferences for location of environmental interventions being in the state or country respondents are located in. This occurs in countries where factual borders are of less importance than in other areas of the world such as the EU. Given the potential importance for policy implementation of such findings, we argue for further research into incorporation of political dimension of space into stated preference research.

In line with the interdisciplinary nature of ecosystem related research the thesis develops an ecological modelling framework that could be integrated

with the economic work (Chapter 4)⁶. **The fourth contribution of this thesis hence lies in development of spatially explicit predictive models of farmland bird populations that can be used for assessing land use change impacts on biodiversity.** The models test how well land use data (that we employ in Chapter 3) can be used for predicting farmland bird populations in Great Britain and which variables are the most pronounced drivers of bird population change. While the modelling work presented in this chapter is of relevance as a stand-alone research project, a broader aim in the context of the PhD thesis was pursued. This has been, since the project inception, to ensure the compatibility of the birds modelling outputs with the preference modelling study presented in Chapter 3. In turn, the valuation project of Chapter 3 was designed with the aim of integrating economic results with economic modelling of bird species. While representing work going beyond the scope of this thesis, this integration will enable an interdisciplinary analysis of land use change related to agri-environmental policy in the UK and is the future focus of my research.

The Concluding remarks summarise the findings of this thesis and the lessons learnt from the PhD research over the past four years. Potential research avenues based on the findings in this thesis and outline of future personal research plans are discussed in the closing text.

⁶ Please note that the researcher conducting this project is not a trained ecologist, nor has he ever received training in biological, ecological or ornithological research. He was kindly supervised for this project by Gavin Siriwardena from the British Trust of Ornithology who provided valuable advice throughout the project.

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CHAPTER 1: VALUING ~~ECOSYSTEM SERVICES~~ PREFERENCES FOR ECOSYSTEM RELATED GOODS AND SERVICES⁷

Tomas Badura*¹, Ian Bateman², Matthew Agarwala^{1,3,4},
Amy Binner²

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Corresponding author: TB - t.badura@uea.ac.uk

Affiliations:

¹ Centre for Social and Economic Research on the Global Environment (CSERGE), School of Environmental Sciences, University of East Anglia, Norwich Research Park, Norwich, NR4

² Land, Environment, Economics and Policy Institute (LEEP), University of Exeter, Exeter, Devon UK
EX4 4SB.

³ Grantham Research Institute on Climate Change and the Environment, London School of Economics and Political Science, Houghton Street, London, WC2A 2AE.

⁴ Department of Geography and Environment, London School of Economics and Political Science, Houghton Street, London, WC2A 2AE.

⁷ The Strikethrough is intended, however was not possible to include in the original, published, version of the Chapter.

1 Introduction: Rationale for valuation of ecosystem service related goods and services.

If all resources were infinite there would be no need for us to value their different contributions to human welfare; indeed there would be no need for economics. Regrettably we do not live in such a world. Indeed human ingenuity has devised multiple ways for us to exceed the capacity of the planet to provide all the resources which our rapidly expanding and increasingly affluent population demand. Given this situation, a fundamental responsibility of good governance is to encourage the allocation of scarce resources such that they best satisfy society's requirements and aspirations. Economic analyses can help evaluate the myriad options for resource allocation in terms of their inherent trade-offs (that is, in terms of what must be given up to pursue them) and identify that course of action which delivers the highest net benefit (i.e. which maximises the gap between the benefits of an option and its costs, including the opportunity costs of forgone alternatives).

When the costs and benefits of each course of action are readily observable, the task of identifying the option with the greatest net benefit is relatively straightforward. However, as is the case with the natural environment, when the effects of an option are imperfectly understood or relevant costs and benefits are difficult to assess, then identifying which course of action is the best poses a major challenge for decision making. Perhaps the greatest example of this challenge is provided by options concerning our use of the natural environment. Ecosystems and biodiversity provide important resources which impact upon a wide range of social and economic goals. From an economic perspective, the natural environment is a value generating resource which should be fully integrated into decision making systems

(Atkinson et al. 2012). However, this integration is often far from straightforward. The natural environment can be viewed as a repository of a variety of ecosystem processes, arrayed in a complex and interlinked web where multiple processes link together to determine the inputs to further processes. From an economic perspective, these processes become of relevance to human welfare when they deliver 'final ecosystem goods and services' (Landers and Nahlik 2013), otherwise known as 'final ecosystem services' (Fisher et al. 2009) or most commonly as 'ecosystem services'. While these have been the subject of much academic debate and reconceptualization (Fisher and Turner 2008; Fisher et al. 2009; Ott and Staub 2009; Haines-Young and Potschin 2010; Bateman et al. 2011; Johnston and Russell 2011; Staub et al. 2011), an apt early definition is that ecosystem services are those "components of nature, directly enjoyed, consumed or used to yield human well-being" (Boyd and Banzhaf 2007). The ecosystem service concept is therefore inherently anthropocentric, focussing upon nature's contributions to human wellbeing. While it is obviously true that underpinning natural processes are vital to the provision of ecosystem services, the two should not be confused. Indeed attempts to value the former processes are liable to result in double counting errors if such values are then added to those of ecosystem services. This concentration on wellbeing also means that common classifications of ecosystem services into "supporting", "regulating", "provisioning" and "cultural" categories (MA 2005), while being 'heuristically relevant' (Landers and Nahlik 2013), are of less pertinence to economic analysis where the focus is on those welfare-bearing (tangible) goods and (intangible) services which are directly related to final ecosystem services.

An important distinction therefore is that economic analysis does not attempt to directly value ecosystem services, but rather assesses the value of the contribution of ecosystem services to related welfare-bearing goods and

services. Occasionally these will be effectively identical. Natural landscapes are both final ecosystem services and valued economic services when viewed by those who consider them aesthetically pleasing. However, many ecosystem services only generate welfare bearing goods when they are used as inputs in a production process and combined with other human derived inputs (such as labour, machinery, expertise, etc.). For example, while natural processes provide wild fish, it is only when combined with fishing expertise, boats, nets, and so on, that fish are converted to food. Nature still provides essential and highly valuable inputs to such production processes, but to confuse ecosystem services with related goods over-values the former and undermines the validity of decision-support analyses.

As overviewed throughout this chapter, valuation of ecosystem service related goods can involve a number of complexities; not least of which is that many of these goods are not traded in markets and so lack market prices. Of course some of these goods are priced (e.g. food and timber), but many others are not (e.g. an equable climate, clean water, natural hazard regulation). Furthermore, the goods and services to which the natural environment contributes are frequently measured and reported in a broad range of units. For instance, greenhouse gas sequestration is reported in tons of carbon equivalent sequestered, water purification in cubic metres of water purified, and recreation in the number of visits to a site. Attempting to draw meaningful comparisons and evaluate trade-offs between these diverse units, especially when we consider the diversity of inter-related effects which arise from, say, land use change, is a task of Herculean proportions. Nevertheless, decision makers are routinely faced with precisely this challenge, which is only made more complicated by the fact that ecosystem services are not always complementary (sometimes an increase in food production comes at the expense of a decrease in water quality). Consequently, there is an obvious

advantage in making the various trade-offs inherent in decisions comparable through a common unit. Arguably any common unit could be used, however, there are particular advantages associated with the use of money; it is a pure unit of exchange with no inherent value. Furthermore, it is of course the unit which is most familiar to decision makers. Importantly, money units place the value of spending on welfare bearing ecosystem services on a level playing field with other investment options. Failure to include the economic value of ecosystem services within decision making has led to their long term abuse and worldwide decline (MEA 2005; TEEB 2011; UK-NEA 2011; CBD 2010). Reversal of this trend requires radical change, rejecting the status quo in decision making and ensuring genuine comparability of the value of maintaining ecosystem services with that of other investments. Economic valuation provides a key element in delivering this change.

Van Beukering et al. (2013) highlight a number of further advantages of the use of economic valuation of ecosystem services, including:

- The role of valuation as an advocacy tool which helps place the value of ecosystem services on the planning agenda by highlighting the importance of ecosystems for private sector profitability as well as for the provision of public goods and human health and security (e.g. air purification and natural hazards protection).
- Ecosystem valuation is also an important tool for assisting transparent and better informed decision making. Since many of the ecosystem benefits fall outside of market systems, they are often overlooked. This can lead to external costs and a level of benefits which might be sub-optimal for society as a whole.

- Valuation of ecosystems can also be used for damage assessment. It can be used in two major ways - either for setting up a compensation fee for potential environmental damage, or for resolving legal disputes between conflicting parties.
- Finally, valuation can also be used for determining taxes, fees or charges for use of ecosystem services which can. Increasing the cost of usage of ecosystem services will, in effect, discourage their usage and hence support conservation.

Valuation research has recently received increased attention in academia and policy. Methodological developments in and related empirical studies on valuation have proliferated over the past three decades, with a notable increase in academic output over the past ten years (Fisher et al. 2009). As a result, economists can draw upon a substantial body of research and a range of tools (see section 3) for assessing the contribution of the natural environment to human wellbeing. These tools are demonstrated through a number of studies which have assessed the values of ecosystem services on differing scales and for various purposes (e.g. Costanza et al. 1997, 2014; Bateman et al. 2013; UK-NEA 2011, 2014; Goldstein et al. 2012; Lawler et al. 2014). Valuation research is becoming ready to use for policy application.

The proliferation of valuation studies is particularly timely as climate change and the natural environment are increasingly prominent policy issues at the local, national and international scales. A series of landmark reviews and assessments have mainstreamed valuation research into broader policy debates (MA 2005; TEEB 2011a, 2011b, 2012; UK NEA 2011; UK NEAFO 2014). Informed, and perhaps even inspired by these, a number of international, EU and national strategic commitments directly acknowledge the need to integrate the values associated with biodiversity and ecosystems into decision

making. Most notably, Target 2 of the Convention on Biological Diversity's (CBD) Strategic Plan for Biodiversity 2011-2020 (CBD 2010) calls for the integration of biodiversity values into planning processes, accounting and reporting processes, while Action 5 of EU Biodiversity strategy (European Commission 2011) refers to assessing the economic value of ecosystem services and their integration into accounting and reporting systems. At a national level, the UK's White paper on the Natural Environment (HM Government 2011) builds directly on the findings of UK's first National Ecosystem Assessment (UK NEA 2011), emphasising the importance of integrating the value of ecosystem services into decision making and, led to the establishment of the UK Natural Capital Committee (NCC), which is responsible for reporting on the state of natural capital and available opportunities to improve its management (see NCC 2013, 2014). Arguably the NCC represents a worldwide first in that it has a direct remit to advise on changes to government economic decisions. Other countries are establishing similar bodies and the value of the natural environment also figures in global efforts to revise national accounting practices (see UN SEEA⁸ - SEEA 2013, 2014; WAVES⁹- e.g. WAVES 2012, 2014; and EU MAES¹⁰ – e.g. European Commission 2013) and plays a role in wider environmental debates related to moving beyond GDP measures (UNU-IHDP and UNEP, 2012) achieving 'green growth' and addressing climate change (Agarwala et al. 2014) (WWF 2014).

However, despite growing evidence of the economic benefits to maintaining a healthy environment, several obstacles continue to limit the use of ecosystem valuation in improving decision making. Most particularly, the

⁸ <http://unstats.un.org/unsd/envaccounting/seea.asp>

⁹ <https://www.wavespartnership.org/>

¹⁰ <http://biodiversity.europa.eu/maes>

institutional acceptance of such methods, combined with the skills to implement valuation remains a significant barrier. Similarly it has to be acknowledged that, while many ecosystem service related goods can be robustly valued, there remain technical limits to valuation. These are compounded by further gaps in the natural science evidence base surrounding the complex functioning of the natural environment, including natural thresholds and tipping points (Bateman et al. 2011).

Of course, given the inherent focus of economics on optimizing resource use, there will be some cases in which economic analyses do not favour environmental conservation. There may be some instances in which maximizing society's overall benefits entails a certain degree of environmental destruction. This is an important issue to understand, especially in the environmental advocacy sector. At the same time, valuation represents an important opportunity to change the way we think about, appreciate the importance of, and manage our natural environment.

The following sections examine three areas of ecosystem valuation research. We first discuss the main concepts underlying this area of research, outlining the problems in our understanding of the natural world and how this influences our ability to assess environmental values. Second, we overview the basic methods of ecosystem valuation, examine options for generalising and transferring results across locations and decisions, and review a pioneering integrated decision-support tool. Finally, we discuss the future prospects for environmental valuation, considering the main challenges faced by researchers and practitioners in this area, and how valuation can act as an institution of change for society-environment relationships.

2 Basic concepts of environmental valuation

All ecosystem management decisions (and any environmental degradation) have intertemporal implications (e.g. Mäler 2008). Given this, we can conceptualise the natural environment as a collation of stocks of ecosystem assets, generating flows of ecosystem services over time (e.g. Atkinson et al. 2012; Barbier 2007; Bateman et al. 2011). In this context, economic appraisal of ecosystem service related goods involves the valuation of service flows over time. The value of an ecosystem asset is therefore net present value of future flows of ecosystem services. The important question then is how the asset value changes in response to human interventions; how future prospects change in response to what is happening to ecosystems and biodiversity now.

The economic definition of value is based on the choices and trade-offs people make in relation to the good or service in question. The most commonly applied indicator of value is a good's market price. But when no such price exists, as is the case for many environmental goods and services, values can be derived either from related markets or from stated behaviour in hypothetical situations (see methods, section 3). Value is then estimated in terms of four measures (Hicks 1943)¹¹: (1) an individual's maximum willingness to pay (WTP) to obtain an increase in the provision of a welfare bearing good or service; (2) their maximum WTP to prevent a reduction in such provision; (3) the minimum amount that an individual is willing to accept in compensation for the loss of a welfare bearing good or service (their willingness to accept; WTA) and; (4) the amount they are WTA to forego a welfare gain from

¹¹ For a case study assessment of all four Hicksian measures within an environmental context see Bateman et al., (2000).

increased provision of such a good.¹² WTP and WTA are reflected in the preferences and choices people express in either existing or hypothetical markets. Summing individual preferences over the relevant population provides an estimate of the corresponding aggregate value of a given change in the provision of an ecosystem related economic good or service. Note that correctly identifying the relevant population is of crucial importance to avoiding over- or under-estimation of related values (Bateman et al. 2006).

The economic concept of value can be divided into use and non-use values (Pearce et al. 1989; Pearce and Turner 1990). These are not mutually exclusive as people can hold both types of values for the same ecosystem related good. Further, use values can be divided into three categories: direct, indirect and option values. Direct use values arise from a direct interaction with an environment and include extractive (e.g. fisheries, timber) and non-extractive values (e.g. recreation or aesthetic value of a natural view). Indirect use values stem from ecosystem service related goods which are not used directly (e.g. water and air purification or natural hazards protection). Option values arise from the potential future use of ecosystems (e.g. from medical research).

Non-use values are not related in any way to current or future use of ecosystem goods or services by the individual expressing the value in question; they arise simply from knowing the continued and maintained existence of an ecosystem (or elements thereof) is secure. Such values are often related to charismatic species and rare habitats (e.g. the Sumatran tiger, Bateman et al. 2010; or Brazilian Amazon, Horton et al. 2003; Morse-Jones et al. 2012). Non-use values can be divided into three categories: existence,

¹² Academic literature in valuation studies generally favours WTP over WTA, especially due to fact that WTP is constrained by income, while WTA is not.

bequest and altruistic. Existence value relates to the satisfaction people obtain from the existence of ecosystems and biodiversity, quite separate from its use. Bequest value relates to the welfare people gain from knowing that ecosystems and biodiversity will be passed on to future generations. Finally, altruistic or other-regarding value (e.g. Ferraro et al. 2003) relate to the satisfaction individuals gain from ensuring that ecosystems and biodiversity is available for other people in their generation. Note that this nomenclature deliberately eschews the use of the term 'intrinsic value'. As noted by Bateman et al. (2011), the word 'intrinsic' is defined by the Merriam-Webster dictionary as "belonging to the essential nature or constitution of a thing". Therefore the intrinsic value of say an endangered species belongs to that species alone and cannot be defined by another entity such as a human. Of course humans can and do hold values for species and these include the use value held by those who visit conservation sites to view wildlife. Similarly a wider group of individuals may hold non-use values for the continued existence of wild species or that the biodiversity is preserved for others, including future generations. However, these are anthropocentric rather than intrinsic values. To claim that we have any knowledge of intrinsic values might lend a confusing and, perhaps, erroneous air of moral justification to assessments. It is the moral and ethical discussions that we might need to address the 'intrinsic' in these debates and which are in many cases enshrined in our laws (e.g. legal protection of endangered species and ecosystems).

In practical terms, economic valuation is (or should be!) limited to those elements of value for which reliable and robust estimates can be obtained. Although in principle it is possible to value all ecosystem services, for some values of natural environment we are currently unable to provide reliable estimates, either due to our limited understanding of the natural world or because of the lack of methodological development. For instance, Bateman et

al. (2011) argue against the use of stated preference based monetary estimates for the non-use value of biodiversity. Such value assessments often show inconsistent results due to lack of familiarity with the concept and importance of biodiversity, as well as lack of any obvious, incentive compatible payment vehicle. Furthermore, a number of studies have found that environmental values are sensitive to objectively irrelevant variations in the framing of valuation questions (e.g. Horowitz and McConnell 2002; Bateman et al. 2008). However, it has also been noted that such anomalies arise in experiments using market priced goods (Bateman et al. 1997a, 1997b; Loomes et al. 2003). Humans are essentially psychological beings and it is unsurprising that some degree of inconsistency can arise across contexts (e.g. Kahneman and Tversky 1979; Kahneman 2011; Bateman et al. 2005; Bateman et al. 2007). This means that an element of any robustness check must be, perhaps regrettably but inevitably, judgement. The degree of anomaly is more important than its mere statistically verified presence. As ever the principle should be that any information is useful if it improves the quality of decision making, and not otherwise. Nonetheless, assessment of those non-market, ecosystem related goods which can and cannot be robustly valued provides important results for policy advice. Further, identification of what we can and cannot quantify and/or value can be useful information for policy and management decisions and can highlight further areas for valuation research (see, for example, the case of non-monetary assessment of biodiversity in Bateman et al., 2014).

Here it is also important to highlight the difference between exercises attempting to reflect the total accounting value of the services provided by an ecosystem and economic analyses of the unit (marginal) value of relatively small changes in those services and their related goods. While there have been attempts to calculate the total value of world/region/country's ecosystems (e.g. the influential work of Costanza et al. 1997; 2014), this approach is being

considered problematic in economic theory terms (e.g. Toman 1998; Bockstael et al. 2000; Heal et al. 2005) and not useful for policy purposes. One line of critique argues that any such (total) value of ecosystems is an underestimate of infinity, as humanity relies on the natural world for its own existence. Most importantly, it is argued that very few policies concern total loss of ecosystems and such exercises are of no (or little) use for practical decisions. We take a somewhat intermediate position. Accounting exercises, whether they are for market or non-market goods, necessarily rely upon certain strong assumptions. Most particularly they typically ignore the increase in marginal values which generally arise as stocks are depleted and related services fall. This is why an assessment of the total accounting value of the world's ecosystem services can be some finite sum while economic intuition points to the real value being infinite. Nevertheless, accounting exercises do play an important role in raising the issue of environmental degradation and biodiversity loss by bringing it to wider policy and public attention. Green national accounts become of greater relevance when they are considered over time as they can flag up trends in stocks and highlight potential areas for policy action (e.g. ONS 2014). However, accounting studies cannot identify the optimal efficient response to those trends. This is where economic marginal values come into their own as they can single out the most efficient courses of action in response to some ecosystem service concern.

Economic analysis of ecosystem services therefore examine marginal values, reflecting the fact that most policies consider changes in ecosystem services provision on a limited rather than absolute scale. Multiplying the marginal value by the relevant amount of units implied by the policy change can then provide the change in ecosystem service values for the policy in question. A problem arises from the fact that, as mentioned above, the marginal value of certain ecosystem service related goods are not constant

(Brander et al. 2006). Consider how the marginal recreation value of urban green space declines with its increasing stock – i.e. the more recreation area people have in a city, the less they value additional areas. While the initial provision of an area providing such services might create very high values per hectare, further additions will generate progressively lower increases in recreational value (i.e. increasingly lower marginal values). Of course this relationship operates in reverse as well; as the availability of green space falls so its marginal value increases. Similarly, when the stock of ecosystems is being used unsustainably and the potential to substitute these services via other (man-made) goods and services is limited, it is likely that the marginal values of the flows of ecosystem services will increase (e.g. tourism values related to endangered species). Moreover, the marginal values might undergo rapid change, such as those caused by technological developments. For example, when whale oil was a key source of indoor lighting, overfishing drove whale populations to near extinction and, predictably, increasing scarcity fuelled higher prices (increasing marginal value). However, with the introduction of electric light bulb, the marginal value of whale oil underwent rapid decrease and the pressure on some whale populations decreased significantly (though some never fully recovered). The issue of changing marginal values is common to many ecosystem services and requires attention in value assessments (see Brander et al. 2006, for an example).

While economic analysis of ecosystem services can be undertaken at a particular site for local decision making, economic analysis can also be part of larger ecosystem service assessments across differing scales. Bateman et al. (2011) provides a general framework and nomenclature for integrating ecosystems and economic analyses in such assessments. Two types of assessment can be made. Sustainability analyses typically focus on past changes in natural capital stocks and seek to determine whether past

development was on a sustainable path in terms of increasing or decreasing capital assets. Conversely, programme evaluation analyses are forward-looking and aim to evaluate development options. This often comprises use of policy scenarios, forecasts of environmental change and trends in domestic and world markets. UK-NEA (2011) and UK-NEA (2014) together provide both types of analyses.

2.1 Importance of context

Values can be highly context dependent, therefore valuation frameworks need to recognise how space, time, biological factors and institutions (social, cultural, political and economic) influence the values assessed. The spatial element of valuation studies is of key importance for many ecosystem services. For example, the benefits of water related services (e.g. water purification and provision) are often experienced downstream from where they were generated, while natural hazard regulation services (e.g. flood and storm protection) can be positioned a considerable distance from the human populations and infrastructure they protect. Moreover, some ecosystem values, particularly use values, decrease as the distance between the asset and the valuing individual increases (a phenomena known as the distance decay effect see e.g. Bateman et al. 2006). Bateman et al. (2011b) show how the location of outdoor recreation sites matters, reporting values between £1,000 and £65,000 per annum for recreation, depending on the proximity to significant conurbations. The spatial dimension of ecosystem service values, including the distance decay effect, is increasingly being incorporated into valuation studies with a use of Geographical Information Systems (GIS) tools and significantly contributes to valuation research development (e.g. see Goldstein et al. 2012; Lawler et al. 2014; Bateman et al. 2013 and 2014). In an ample illustration of the importance of spatiality in environmental policies

and economic valuation of ecosystem related goods and services, Bateman et al. (2013) show for the case of Great Britain how spatially targeted environmental-economic policies could help to achieve greater benefits from land use than uniformly applied policies.

Ecosystem related values are also influenced by the time profile over which they are measured and accounted. Many of the values of ecosystem services occur over long time scales (e.g. carbon sequestration in peat soils and its impact on climate). To account for the temporal dimension of benefits, economists use (social) discount rates, which reflects the theoretical and empirical observation that, for a wide variety of reasons, people prefer receiving benefits sooner than later (Pearce and Ulph 2005). There is, however, no clear consensus on the choice of discount rates and it is source of considerable debate. While one, descriptive, viewpoint argues that the choice of discount rate should reflect the current preferences and market choices about future (generation), the prescriptive response is that this is more a matter of ethical perspectives. Given that the choice of discount rate has a significant impact on the level of total streams of evaluated future benefits and hence on related policy implications, it is therefore a point for careful consideration. As such, one approach is to use of a range of discount rates, considering the time period and uncertainties involved, as well as the policy/project being evaluated. Another is to consider different discount rates for manufactured goods and for ecosystem services, with the latter being significantly lower (Baumgärtner et al. 2015). Gowdy (2010) provides an extensive discussion of discounting in the context of ecosystem valuation research while Heal and Millner (2014) consider approaches for accommodating different perspectives within the context of climate change decision making.

2.2 The crucial role of natural science input

Natural science input lies at the heart of any ecosystem service related valuation. Economics is well suited to assess the link between quantified ecosystem services and people's wellbeing. However, it is crucial to understand the 'production function' of ecosystem services from a natural sciences perspective, e.g. how different biological factors influence ecosystem functions. This is just as important (if not more so) as understanding how management decisions influence ecosystems and the services they provide.

A particular challenge is the still limited understanding of the role biodiversity plays in ecosystem functioning and in the provision of ecosystem services (see Naeem et al. 2012 for a review; Mace et al. 2012; Mace 2014). For many ecosystems we lack sufficient understanding of natural thresholds and tipping points to reflect this within subsequent economic analyses. The problem here is partly the economics, but rather the crucial knowledge of biophysical relationships. The principles are reasonably well understood. For example, we understand the theory of how fisheries can breach sustainability thresholds when populations collapse after a period of unsustainable harvest. Similarly, an increased use of fertilisers can result in run-off nutrient pollution causing sudden major eutrophication incidents in certain lakes (e.g. Carpenter and Brock 1999; Mäler et al. 2003). Prolonged degradation of environmental systems can lead to state changing and potentially irreversible tipping points with accompanying major welfare losses (Rockström et al. 2009).

Ideally analyses of potential tipping points should appraise the value of maintaining resilience within natural systems. Ensuring that the natural environment maintains the capacity to self-equilibrate aftershocks is clearly vital, especially if the frequency and intensity of such shocks is liable to increase in line with a general degradation of natural systems caused by

pressures such as climate change. However, the natural science information required to adequately assess (and thereby value) the maintenance of resilience levels is to date, rarely available. An exception to this is provided by Walker et al. (2010) who consider the problem of deforestation leading to salinization of soils in Australia. In the absence of such information, concerns regarding the maintenance of resilience may be better handled through the adoption of precautionary approaches such as Safe Minimum Standards (SMS). Here economic valuation and decision making operate as usual until a threshold in ecosystem functioning is identified. SMS are then employed to ensure the resilience of resources with economics being confined to the identification of cost-effective solutions for delivery of those standards (Bateman et al. 2011).

3 Methods for valuing ecosystem related goods and services

There are a number of methods available for estimating the value of ecosystem service related economic goods and services. In some cases market prices can be relevant (e.g. for timber or food), although adjustments may have to be made for any distortions in those prices (e.g. to allow for the impact of price subsidies or constraints). However, many important goods and services, including a large number associated with ecosystem services, arise outside market contexts¹³. This problem has been long recognised and from just a few path-breaking studies in the middle of the last century a burgeoning literature has developed detailing methods for the economic valuation of non-market goods. This section outlines the major methods employed for valuing ecosystem related goods and services and provides illustrative examples of each. Specifically dedicated reviews and guidelines for ecosystem valuation methods have been developed and for more details the reader is referred to some of these (e.g. Bateman et al. 2002; Champ et al. 2003; Freeman et al. 2014; Bouma and van Beukering, 2015).

Broadly speaking, valuation methods can be categorised as:

- Market valuation methods where market price information is used to indicate the value of related non-market goods. Approaches include the direct use of market prices (adjusted as necessary) for ecosystem service related goods, production function approaches and, where appropriate, the use of cost

¹³ A major reason for this is the lack of private property rights associated with many ecosystem services (e.g. Pearce and Turner 1990; Pearman et al. 2011)

information (for example in assessing the marginal costs of abating greenhouse gases).

- Revealed preference methods where analysis of the purchase of market priced goods is used to indicate the implicit value of strongly related non-market goods. Approaches include hedonic pricing (e.g. assessing the uplift in property prices attached to houses in quieter location) and variants of the travel cost method (where an individual's willingness to incur costs reveals their value for the recreational sites they visit).
- Stated preference methods, where hypothetical markets are developed and survey or experimental participants make choices regarding the provision and associated costs of non-market goods.

3.1 Market valuation methods

The market price of certain goods provides useful information regarding the value of related ecosystem services. However, two practical problems need to be addressed before an unbiased estimate of underlying non-market values can be obtained. First, as mentioned above, prices need to be adjusted for any market distortion (such as government subsidies or taxes) as well as for non-competitive practices. Second, ecosystem services may constitute only a portion of the value of inputs underpinning the production of a given marketed good. Other inputs such as labour, expertise, machinery, and other manufactured or social capital are also required to produce such goods. To the extent that these other capital inputs might be reallocated to the production of alternative goods, so the price of any given good should not be wholly designated as being the value of embodied ecosystem service inputs. These various contributions to value can be disentangled through the estimation of

production functions, revealing the value added by each input. For example, Fezzi et al. (2014) examine the contribution of climatic conditions (specifically temperature and precipitation) to agricultural output in the UK. By controlling for the contribution of other inputs (such as fertiliser) and policy interventions (such as subsidies), they isolate the effects of climate and use this to examine the likely impact of future climate change. These effects are expressed both in terms of changes in land use and, crucially, the economic value implications of climate change. Applying this method requires collection of data on and understanding of how changes in the quality and quantity of ecosystem services affect the costs of production of the final good, the supply and demand for that good and for the other factors of production (Koetse et al. 2014). This method can in principle be applied to the valuation of inputs from a variety of ecosystem services, ranging from the maintenance of beneficial species such as pollinators to protection from tropical storms (Barbier 2007).

Some studies have used costs as approximation of the value of ecosystem service inputs. One approach is to look at the damage costs avoided by not allowing an ecosystem service to degrade (e.g. storm and flood protection; Badola and Hussain 2005). Similarly, it is possible to analyse the expenditure and behaviour people incur to avoid such damage (Rosado et al. 2000). Some studies also look at the cost of replacement or restoration of an ecosystem service. However, the latter two are considered problematic as these costs might have little relationship to the values they aim to approximate (e.g. Barbier 2007; Heal 2000).

One obvious limitation of the above methods is that they can only apply to ecosystem services which are directly related to the production of market price goods. Alternative approaches are required in cases where that relation is more indirect, and it is to such methods which we now turn.

3.2 Revealed preference methods

Many ecosystem services are associated with non-market, unpriced goods. Here revealed preference approaches such as the travel cost or hedonic pricing method can prove useful. The travel cost method is a commonly used approach for valuing recreational benefits. It relies on the premise that values of recreation benefits are implicitly shown in people's behaviour in travel markets. Through analysis of travel expenses in terms of actual travel costs (e.g. fuel, car maintenance, costs of public transport, aeroplane tickets, etc.), time costs and admittance fees it is possible to assess the implicit price of access to the recreation site and incur the value of recreational benefits. Employing the travel cost method Egan et al. (2009) combined information from an extensive household survey on recreational usage of 129 lakes in Iowa, US, with the detailed information on lakes' water quality to estimate the demand function for recreational trips to Iowa's lakes conditioned on their water quality. The study further examined the welfare implications of three different policy scenarios for water quality improvements and provided policy recommendations, highlighting the importance of prioritization of clean-up activities of smaller number of lakes distributed across the state for generating the greatest recreational benefits.¹⁴

Another route towards revealed preference valuation is provided by the hedonic pricing method. This measures the implicit price of an ecosystem service related good as revealed in the observed price of an associated, market priced good. The most common application of the hedonic method is via the property market as house prices, once they are stripped of the influence of structural factors (number of bedrooms, garden size, etc.), neighbourhood

¹⁴ Note that the Egan et al. (2009) did not include alternative specific constants which has become a norm in this type of models. See e.g. Timmins and Murdock (2007).

variables (local unemployment rates, etc.) and accessibility characteristics (access to places of work, high quality schools, etc.) reveal clear associations with local environmental quality. Common applications include the valuation of aesthetic views, air quality, flood risk and many other local amenities. For example, Day et al. (2007) use the hedonic approach to reveal the values homeowners place upon quieter neighbourhoods showing how marginal values increase with noise levels (and also revealing that urban dwellers seem more attenuated to a given level of car noise than to the same level of aircraft noise).

3.3 Stated preference valuation methods

All the methods listed so far rely on observed behaviour either directly or indirectly occurring in extant markets. However, an alternative approach is to generate hypothetical markets through which survey or experimental respondents can be asked to express their WTP or WTA for changes in the provision of ecosystem service related goods. This is particularly useful for assessing preferences regarding situations which have not yet occurred. For example, water company customers can be asked questions regarding potential future improvements to rivers and lakes in their area (Metcalf et al. 2012). Stated preference methods are also the only approach available for estimating pure non-use values such as those associated with the survival of endangered species (Morse-Jones et al. 2012), although, as outlined previously, we have reservations about the robustness of such methods if respondents do not have well-formed preferences for such changes (and suggest the use of SMS in such circumstances; Bateman et al. 2013).

Two variants of the stated preference approach are in common usage, the contingent valuation method (CVM) and choice modelling (CM) technique. As can be inferred from its name, the CVM approach elicits individuals' values

contingent on there being a (hypothetical) market within which those values can be expressed. All four WTP/WTA measures can be elicited using CVM provided that the market can be conveyed in a manner which respondents find credible. As the first of the stated preference methods to be widely applied, CVM has been subject to considerable critical appraisal and it is certainly true that the questionnaire framing effects long recognised by psychologists in other contexts and the preference anomalies identified by experimental economists in the laboratory frequently translated over to stated preference applications. However, this has generated a wider awareness across the economics profession of the complexity of human preferences within both environmental and other contexts. In response to this, CVM applications have undergone stringent examination with increasing emphasis being placed upon the crucial role of design and implementation in applications. Carson et al. (1994) conducted a CVM study to estimate the monetary measure of the compensation for negative impact of chemicals on wildlife species in California, USA. Employing a referendum format of the CVM, where respondents vote for or against a particular policy, the study estimated the WTP for decreasing the recovery period of the four affected species from 50 to 5 years to be around \$575 million (with a standard error of \$27 million). Notably, the development of the survey used in this study was conducted over the course of 32 months and provides a fine example of a comprehensive CVM study.

CM approaches are, in many respects, similar to the CVM. Again they can, in theory, be applied to almost any ecosystem related good and rely upon hypothetical markets to elicit respondents' choices. However, while CVM typically asks respondents about a single change in provision of a good, CM approaches elicit choices regarding multiple such changes. This is achieved by noting that many goods are composed of multiple attributes. For example, a

given land use might involve different areas (or 'levels') of woodland, farmland and conservation land. By varying these levels we define multiple goods. CM respondents are then asked to choose between some set of these goods. As multiple definitions of these goods can be created so respondents can be asked to answer many such choice questions, generating large amount of preference data. By adding a variable cost to each of these goods the analyst can observe how respondents trade off money against changes in the levels of each attribute.

3.4 Value transfer and its variants

In many cases the costs of undertaking high quality integrated natural science and economic valuation assessments is vastly outweighed by the net benefits generated from the improved decision making facilitated by such studies. However, in some instances the resources necessary for such studies are unavailable. Provided that some estimate of the physical impacts of an intervention is available then values may be approximated through value transfer¹⁵ methods.

In essence, value transfer takes information from previously assessed 'study' sites and utilises this to estimate values for some alternative 'policy' sites.¹⁶ It is not a valuation method per se, as it is based on results from previous valuation studies. Nonetheless, it is a potentially useful and cost-effective technique for value assessments and reflects a pragmatic approach, recognising that it is not possible (or necessary) to value all ecosystems and their services when we have enough base studies from which values can be (robustly!) extrapolated.

A key requirement for the correct use of value transfer methods is that, on the assumption that precise matches between policy and study sites are unlikely, any differences are understood, quantified and incorporated within the transfer process. Common adjustments between sites are to reflect differences in standards of living, varying levels of population, different spatial configurations and substitute availability, or different levels of provision change. The general aim of adjusting for the differences between the sites when using value transfer approaches is to minimise the 'transfer errors'

¹⁵ Sometimes also called benefit transfer. This is not fully correct as this technique can be applied to costs as well.

¹⁶ Note that value transfer can also be used to assess different changes (than originally assessed) at the same site.

(the difference between the transferred values and the 'actual' values which a particular site or change generates).

Value transfer methods embrace a variety of techniques varying from the simple transfer of adjusted mean values (univariate transfer) to sophisticated applications of *value functions* (multivariate transfer) specifically developed for transfer purposes. Simple value transfer (e.g. Muthke and Holm-Mueller 2004) takes values from primary valuation study sites (or a pool of such studies) and transfers these to the focal policy site(s). The values transferred are either (adjusted) means or unit values. It is important to note here that the adjustment of transferred values needs to be executed carefully - poor or incomplete adjustments can exhibit bigger errors than simple mean transfers (Brouwer and Bateman 2005).

The value function transfer approach (e.g. Bateman et al. 2011b), in contrast, employs statistical methods to estimate a relationship between the site characteristics and values estimated. The derived function is then used to predict values for the policy site(s), using the data from the new site(s). This method explicitly incorporates the difference between the sites, as the actual characteristics of the policy site determine the final value obtained from the function. Clearly, identifying the most appropriate variables and specifications for such value functions becomes a central issue. Potential approaches include meta-analyses of the extant valuation literature (e.g. Brander et al. 2006) or simply using statistical methods to identify relevant variables. However, Bateman et al (2011b) argue that for value transfer purposes it is preferable to use function specifications which conform to economic theory (and possibly omit some context specific variables) rather than simply rely upon the best statistical fit (see Bateman et al 2011b for further details). This is because, while best fit models may incorporate site specific

variables unique to study sites, theoretically derived functions are more likely to incorporate generic variables, applicable to both study and policy sites.

Bateman et al (2011b) also provides a performance comparison between the univariate and multivariate approaches to value transfer in a controlled multi-site experiment. Results show that for heterogeneous set of sites, value function transfer exhibit lower transfer errors than using mean value transfer. In contrast, when transferring between similar sets of sites, mean value transfer performed better.

Recent methodological developments in value transfer include the adoption of the GIS techniques (see e.g. Bateman et al. 2002b, 2006; Troy and Wilson 2006; Brander et al. 2012). This trend has a promise of further methodological refinement for value transfer, reflecting the – in many cases crucial - spatial dimension of ecosystem service and related goods provision (see section 2). Incorporating GIS into valuation studies facilitates the construction of spatially explicit value functions. Sen et al. (2013), for example, provide a novel methodology and application for spatially sensitive prediction of outdoor recreation visits and values for different ecosystems. Using data on recreation trips in the UK from over 40,000 households, geographical and environmental data and meta-analysis of recreation values the authors derive spatially explicit estimation of visit numbers and recreation values under present and potential future land use in the UK.

Despite its promise, value transfer techniques are necessarily constrained by the availability of primary valuation studies. A range of initiatives aim to collate already existing valuation evidence and organise it in searchable database form - e.g. Environmental Valuation Reference Inventory (EVRI)¹⁷.

¹⁷ www.evri.ca

Such databases can be a useful tool for initial screening for value transfer purposes and helps to systematize the available evidence. However, coherent coverage of valuation studies is far from complete both in terms of the quality and quantity of applications. In particular, there is need for a systematic effort to build a primary valuation base from which value transfer can be conducted. In effect, this requires coordination on a transnational level and – ideally - specification of globally agreed ‘valuation guidelines’ to ensure development of a comprehensive set of primary studies (varying in terms of geography, value types, techniques used and ecosystems assessed) which would form such base on a global level. Indeed, this essentially calls for a large scale medium-to-long term research programme which might also include commissioning a set of studies solely for the value transfer purposes.

3.5 Decision support tools and integrated ecosystem analyses:

Most recently, a range of decision support tools aiming to support systematic ecosystem assessments have emerged. Being broad in the scope of their focus, these tools integrate ecology, economics and geography, with some employing ecosystem valuation tools (see Bagstad et al. 2013 for an overview of 17 of such ecosystem service tools). A number of these rely on an integrated set of models, providing a comprehensive scenario modelling, which can support policy and decision making with a spatially explicit advice (see e.g. InVEST, Tallis et al. 2013; ARIES, Bagstad et al. 2011; or TIM, Bateman et al. 2014). These tools vary in their ability to address differing scales (spatial and temporal) as well as data and computational constraints, but most seek to inform decision making by mapping the impacts of say climate and land use change on the provision of ecosystem related goods and services. Employing the Integrated Valuation of Environmental Services and Tradeoffs (InVEST) tool, Goldstein et al. (2012) analyse seven land use scenarios for a private land

development in Hawaii with an aim to balance both private and public interests on a local level, while taking into account carbon storage, water quality and financial return over a 50 years horizon. Lawler et al. (2014) project land use change for period 2001-2051 in the United States under different policy scenarios. The study analyses resulting effects on ecosystem related benefits, in terms of carbon storage, food production, timber production and habitats provision for selected species, and examines the effects of incentive and land use regulation policies. Bateman et al (2014) in UK NEA (2014) provide analysis of optimal land use policy in Great Britain for period of next 50 years in terms of spatial explicit advice for forestry policy under different optimising rules¹⁸. The Integrated Model (TIM), developed for this study, takes into account monetary estimates of agriculture and timber production, recreation, and agricultural and forestry Green House Gasses emissions, and non-monetary measures of water quality and bird diversity.

The integrated modelling tools, such as the ones listed above, make use of different combinations of valuation methods (as outlined throughout this section) and represent an integration of knowledge and models across disciplines and often require several years of development and data harmonisation. Indeed, they have their limitations and further improvement is required, though they represent an ample example of how ecosystem science and valuation research can support decisions on different levels and show promise of further development of ecosystem valuation research.

¹⁸ The integrated model can optimise the land use under given optimisation rule. The rules explored in the case study included maximising market value, social value (i.e. taking into account full spectrum of ecosystem services in the model) or partial social values (taking into account some of the ecosystem services).

4 Conclusions: The future of valuation research

“Reflecting the value of nature” is an increasingly common phrase in policy debates related to sustainability and conservation. This is good news for conservation and nature in general, but also for people. Omitting the value of nature from decisions at any scale is bound to lead to suboptimal decisions through inefficient resource allocation. This flaw in our decision making frameworks not only inhibits better decisions, but is also a major driver of extensive environmental degradation. Economics and its tools for valuing ecosystem related goods and services, are well positioned to help to correct this flaw by identifying, quantifying and, when possible, expressing the value which nature generates for human societies in monetary terms. The use of monetary values is not always necessary, nor it is always robust (e.g. in case of biodiversity). However it does possess the great advantages of being readily understood by stakeholders at all levels, reflecting diverse biophysical impacts in a common unit and being directly comparable with other areas of decision making.

Failure to include the value of nature in decisions can lead to inefficient resources allocation and further environmental degradation. Carefully applied, economics and monetary valuation can fill this gap. It provides information that is understood by stakeholders at all levels and facilitates comparative assessment across different policy areas. Increasingly, the value of ecosystem related goods and services has become a central issue in environmental policy debates. Such attention is appropriate given the importance of economic analysis for understanding and addressing many of trade-offs underlying environmental decision making. Valuation research can play a major role in transforming the way we manage and interact with nature, particularly by bringing nature into decision making across scales. However,

the policy demands on valuation research are immense. Current EU and CBD biodiversity targets aim to account for and incorporate the values of ecosystem services and biodiversity in decision making frameworks by 2020. While initial progress towards these targets has been made (SCBD 2014), actually delivering on these objectives still requires urgent and extensive action - in terms of knowledge generation, but also through further mainstreaming across society.

Further work is required to strengthen the scientific knowledge base surrounding ecosystem services assessments and accompanying economic analyses. This concerns both work within and across the involved scientific disciplines. Both economics and natural science have a number of areas urgently needing further researched in order to strengthen the ecosystem assessments required for policy use. As economists continue developing valuation methods, they must improve their understanding of preference formation and dynamics, integrating advances from behavioural and experimental economics as well as subjective wellbeing research and contributions from other disciplines such as psychology or cognitive science. In turn, natural science's further understanding of ecosystem functioning, its relation to biodiversity and the role and occurrence of thresholds and non-linearities are a key basis for further advancement of ecosystem assessment research. The challenge for ecosystem science is indeed of sizeable proportions, requiring "a new kind of interdisciplinary science ... to build understanding of social-ecological systems" to support decisions with robust advice (Carpenter et al. 2009, p. 1309).

Further, valuation of ecosystem related goods and services require building up a strong and robust evidence base. A co-ordinated effort is crucial to develop a sufficient basis to support broad-scale value transfer, as well as

to build set of best practice examples for replication in different contexts. Importantly, rigorous monitoring systems for mapping the outcomes of policy interventions are crucial for understanding of valuation-based (and broadly speaking ecosystem related) policies, their improvement and further refinement. Environmental economics, as any other research with policy implications and on-the-ground results, needs evaluation programmes similar to ones present in development studies which evaluate interventions, their results and effectiveness. In the case of valuation research and applications, this can take form of follow up studies, monitoring how the results of analyses influenced on-the-ground reality, as well as how the values estimated transpire in reality. This can have benefits for both knowledge base building and for further understanding of what works on the ground and what not. In addition, such evaluation can help us understand the dynamics behind formation of values and their change through time.

Co-ordination of related policy processes and research initiatives is crucial in order to exploit synergies and strengthen the evidence base. Environmental valuation exercises can inform numerous policy and research processes, including those related to climate change, mitigation and adaptation (e.g. including reduced emissions from deforestation and forest degradation); moving beyond GDP and natural capital accounting; water and agricultural policies; and natural hazard policies. Moreover, co-operation across sectors can highlight pressing problems and identify priority research needs. Importantly, valuation research is well positioned for a productive co-operation with the private sector, which can play a key role in both changing the impact businesses have on environment and driving consumer behaviour change. While numerous areas exist where businesses can build on valuation research (e.g. TEEB 2012), there are still only isolated cases of cooperation between valuation researchers and private sector.

Valuation research and ecosystem research more broadly has gathered momentum and holds potential to change the way nature has been disregarded in our decisions throughout the past century, causing a major environmental destruction. Valuation research can contribute to ecosystem related research and further to broader 'sustainability science'. However, bridging across disciplines and sectors is vital for this information to turn into changes in decisions. This remains a major challenge, indeed, '[s]uch a massive effort in social–ecological science is unprecedented in human history, yet it is commensurate with the problems we face and with the potential of sustainability science.' (Carpenter et al. 2009, p. 1311).

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CHAPTER 2: OUR BIODIVERSITY IS BETTER THAN YOURS - SPATIAL EFFECTS ON PREFERENCES FOR SEAGRASS RESTORATION IN SOUTH EAST AUSTRALIA

Tomas Badura*¹, Silvia Ferrini^{1,2}, Ian Bateman³, Amy Binner³ and Michael Burton⁴

Contributions: The lead author (**TB**) lead all the work related to this chapter, with invaluable inputs from the supervisory team (**MB, IB, SF, AB**). In particular **TB, MB** and **IB** designed the study, **TB** with inputs from **MB** and **SF** analysed the data and **TB** with inputs from the whole supervisory team wrote the chapter. The study was designed in 2014.

Affiliations:

1. Centre for Social and Economic Research on the Global Environment (CSERGE), School of Environmental Sciences, University of East Anglia, Norwich Research Park, Norwich, NR4
2. Department of Political and International Sciences, University of Siena, Siena Italy.
3. Land, Environment, Economics and Policy Institute (LEEP), University of Exeter Business School, Exeter, UK.
4. School of Agricultural and Resource Economics, The University of Western Australia (M089), 35 Stirling Highway, CRAWLEY WA 6009, Australia

1 Introduction

1.1 The Context

Alongside harbouring a variety of species of marine fauna and flora, the marine environment also provides a set of services supporting both local communities and global societies (Barbier 2017). These benefits include the provision of fish stock, recreation opportunities through tourism and recreational fishing, protection from natural hazards (e.g. coastal protection provided by mangrove forests and coral reefs) and oceanic climate regulation (ibid). For example, in 2013 it was estimated that fish accounted for around 17 percent of the global population's intake of animal protein consumed, providing more than 3.1 billion people with almost 20 percent of their intake of animal protein (FAO 2016). Similarly, oceans play an important role in the global carbon cycle of the earth and hence influence the impacts of climate change (e.g. Falkowski et al. 2000).

Globally, the marine environment is under ever increasing pressure from human activities that negatively impact on marine biodiversity as well as on the services provided to human populations (e.g. Worm et al. 2006; MA 2005). Most of these impacts are related to overexploitation of biotic resources, climate change effects (ocean warming, acidification and hypoxia/anoxia – i.e. high/low levels of oxygen) and pollution, all of which threaten the oceans' health, which in some (local) cases can lead to ecosystem collapses¹⁹.

Australia, which provides the setting for the case study focus of this research chapter, has a rich and globally important marine biodiversity,

¹⁹ See e.g. Special Issue of Marine Pollution Bulletin, Volume 74, Issue 2, 2013, The Global State of the Ocean; Interactions Between Stresses, Impacts and Some Potential Solutions. Synthesis papers from the International Programme on the State of the Ocean 2011 and 2012 Workshops.

exhibiting one the highest levels of species endemism worldwide. The Australian coastline (of length of around 36,000 km) is globally unique as it covers tropical, temperate and polar waters. It has one of the most diverse ranges of marine life in the world, with estimated 11% of all known marine species in the world present in Australian's waters, and harbours over 5,000 species of fish and around 30% of world's sharks and rays (Australian Bureau of statistics, undated).

In Australia, the impacts of human activities on marine environment follow global trends. Although the Australian marine environment is assessed as being in generally good condition, this is mostly due to the good condition status of the offshore waters and the areas where the pressure are the lowest (State of the Environment Committee 2011). In contrast, in the inshore waters near to coast of the south west, east and south east of Australia the conditions of ecosystems are poor (ibid). This is predominantly a result of the altered inflow of nutrients, pollutants and high level of pesticides found in many areas adjacent to intensive agricultural production and major conurbations (ibid).

Biodiversity protection could be achieved through number ways. One side lies internalisation of externalities and associated financing schemes, such as payments for ecosystem services in the terrestrial context. These approaches have proved difficult in the marine context, due to difficulties in implementing and enforcing property rights that are essential for this (Coase 1960) Therefore, another useful approach for biodiversity protection in the marine context is direct regulation. Indeed, the main conservation tool for addressing the global decline in biodiversity and the provision of ecosystem services has traditionally been regulatory tools with a prominence of protected areas designation. However, the global efforts for conserving biodiversity are unevenly distributed between terrestrial and marine biodiversity, with the

protection status of the marine environment lacking behind the progress made on the land. While 17% of the land is projected to be protected by 2020 and is heading toward meeting the associated Convention of Biological Diversity's (CBD) Aichi target²⁰ of 20%, the 10% target for the marine area protection is not expected to be met (Tittensor et al., 2014).

Biodiversity protection is underfunded for meeting the global commitments and this is particularly pronounced in Australia (Waldron et al. 2013). The lack of protection coupled with accelerating degradation of marine environment calls for targeted interventions and effective implementations of protected areas networks. Ecological economic analyses can help to understand how to achieve the best socio-economic and ecologic results for the least costs. Information from non-market valuation, in particular can, help to understand the benefits that environmental interventions bring to society and weigh them against their costs (see Chapter 1 for overview of valuation approaches and argument for their use in decision making). This can help to select policies which show the highest "returns" in terms of the broader welfare goals pursued, as well as conservation/ecological objectives when complemented by natural science input.²¹ In addition, further understanding the preferences people hold towards the environmental protection can help to inform the design of policies and facilitate broader political support from relevant populations.

²⁰ <https://www.cbd.int/sp/targets/>

²¹ While we acknowledge that natural science input is a crucial and essential input into full analyses of environmental policies (see also chapter 1 for further information), the present research study solely focuses on preferences for environmental policies.

1.2 Focus of this chapter

We report here the results of an economic valuation study of preferences regarding location and accessibility of sea grass restoration projects in South-East Australia (SEA). This research specifically focuses on how spatial location influences preferences for environmental interventions that are designed to influence the provision of non-market goods and services. We designed a choice experiment that enable the examination of preferences and a hypothesized community association effect. This effect, we hypothesise, is a phenomenon that leads to the clustering of respondents (and their preferences) due to association with a particular (spatially relevant) community – in our case state-based communities in South East Australia (South Australia, Victoria and New South Wales). We show that, while significantly heterogeneous across the sample population, respondents exhibited on average preferences for sites within their own states in our study. These findings are particularly interesting in Australia – a country where the state divisions are mostly of administrative nature²², with same language and no border crossing between the states and territories. The presence of community association effect in Australia might imply that such effects could be expected to be more significant in, for example, Europe or other parts of the world where language, cultural and physical borders exist. Indeed, nationalism has been more strongly present in past two years in politics (e.g. election of US president, UK’s decision to leave the EU)²³.

We also provide some evidence of specific characteristics of location that matter to respondents. In particular, in our sample we could see a strong negative preference for restoration locations that are in close proximity to any

²² See e.g. <http://www.australia.gov.au/about-government/how-government-works>

²³ See e.g. [The Economist \(30th July 2016\) The new political divide](#)

of the major cities in the region. This effect is present when we control for distance and across all three cities in the region, suggesting that it is the specific characteristics of being close to a city that influences sample's preferences.

1.3 Research context

The past two decades have seen an increased research interest in the spatial dimensions of non-market values (e.g. Sutherland and Walsh 1985, Bateman et al. 2006, Brouwer et al. 2010, Campbell et al. 2008, 2009, Czajkowski et al. 2016, Johnston et al. 2002, 2016, Johnston and Ramachandran 2014, Schaafsma et al. 2012, 2013, Schaafsma and Brouwer 2013), however our understanding of how space influences environmental values is still in its infancy.²⁴ This understanding, though, is precisely needed for designing optimal environmental policies that have – often – heterogeneous spatial and temporal impacts (e.g. Rodriguez et al. 2006, Bateman et al. 2013).

The present study contributes to the field of spatial stated preferences by looking at how location impacts on elicited preferences for environmental improvements. Employing the discrete choice modelling methodology, the study looks at what aspects of the location of the site of environmental intervention matter and uses maps to portray the location of such interventions to respondents. Maps have increasingly been used in choice experiments to portray the relative locations of intervention sites and/or to support valuation narratives (see Chapter 3 for further discussion of the use of maps in Choice Experiments).

The study focuses on three aspects of space. Following standard practice in this field, the study accounts for an effect whereby some environmental

²⁴ See Chapter 3 for a more detailed overview of the literature in spatial stated preference research.

values decrease with distance - known in the literature as distance decay effect (e.g. Loomis 2000, Bateman et al. 2006, Schaafsma et al. 2012, 2013, Liekens et al. 2013). Second, the present study aims to shed some light on the importance of political boundaries in valuation studies, through accounting for state borders between the three states of South East Australia where the hypothetical interventions would take place. This research has been in part motivated by a recent increase in nationalistic interests globally²⁵ and by the recent valuation studies that document preferences for provision of environmentally related goods in the country of provision that respondents resides in (see the next section for further details). Thirdly, the study looks at how specific locations in the region impacts on preferences for environmental interventions, aiming to test whether any specific non-structural spatial factors influence non-market values. In particular, we look at the locations that are in close proximity to capital cities in the region.

To our knowledge, these three combined aspects of how space might influence preferences for environmental intervention have not been addressed simultaneously in the literature before. We hypothesise a community association effect whereby respondents' preferences are clustered due to particular spatial delineations (e.g. state borders or proximity to major cities) and that this might impact on how respondents feel about environmental intervention across space.

The study results show that people have a preference for restoration activities to take place in their state (other things held constant) and show strong negative preferences for sites located close to major cities in the region both within their own state and particularly located in other states. At the

²⁵ See e.g. <https://www.economist.com/news/international/21710276-all-around-world-nationalists-are-gaining-ground-why-league-nationalists>

same time the analysis shows that these effects are heterogeneous across respondents and that there might be specific aspects of space (and/or respondents) influencing non-market preferences that warrant further investigation.

1.4 Preferences for politically delineated areas

Beyond physical space, people has been shown to have heterogeneous preferences across political boundaries, for example, exhibiting “premiums” for environmental goods and services delivered in the respondent's own country of residence (e.g. Dallimer et al. 2014; Rogers and Burton 2017; Bakhtiari et al. 2018). While this effect might be intuitively expected, it has been documented in only a limited fashion in the literature – for example in Dallimer et al. (2014) where respondents in the valuation study exhibited a “patriotic premium” (ibid) for locally delivered goods in comparison to goods delivered in other countries. Similar result has been observed in study looking at the preferences for biodiversity offsets in Australia (Rogers and Burton 2017) an in Hoyos et al. (2009) who document an increase in mean WTP for protection of natural resources in Basque country for respondents that claim their cultural identity to be Basque. Most relevantly, Bakhtiari et al. (2018) controlled simultaneously for both country of provision and distance, as we do in this and the following chapter, to show the separate effect of country of provision next to the one by distance to the concerned site. Further, in contrast to other studies which predominantly concerned relatively different countries (culturally or language wise) which were generally far from each other, Bakhtiary et al. (2018) similarly to this and the next chapter looked at countries (or states) that are close and broadly similar.

The above literature evidences existence of preferences for provision of environmentally related goods and services within country where the

respondents live. Such 'cultural' preferences for the provision of environmental goods and services related to administrative, political and/or other cultural boundaries were shown to be motivated by inter alia sense of ownership, cultural identity or ethical concerns (e.g. Hoyos et al. 2009; Ressurreição et al. 2012; Dallimer et al. 2014; Dallimer and Strange 2015; Daw et al. 2015; Faccioli et al. 2018). We look at the impact of political boundaries on preferences for environmental goods in the context of South East Australia and (in the next chapter) Great Britain where borders are of relatively little practical significance. Such 'country' effects would hence be of significance if still present despite the small role the political boundaries play in these areas in contrast to other regions where environmental policies are implemented in a transboundary manner (e.g. European Union). Please note that while the research presented in this thesis does not focus on the reasons why people might feel association with the country/state where they live²⁶, but only explore evidence of such preferences, it is of further research interest to explore these topics in the future.

The next section will present the spatial choice experiment, including the survey and experimental design. The next two sections set out the empirical approach chosen for this study. The fourth section presents the results, while fifth section discusses these results and what do they imply. Final summarises main findings and discusses their implications.

²⁶ Another area of research that might inform these questions would be e.g. human geography, psychology, sociology or political sciences.

2 Survey instrument design

2.1 Survey instrument²⁷

The valuation survey was implemented online following expert consultation and pilot testing. The survey instrument was developed over a four month period in 2014. This involved first understanding the case study region, including the ecological and policy context. Second, a realistic valuation scenario was developed and experimental design for the survey was prepared, followed by the development of the questionnaire and online survey tool.

2.1.1 The geographical, ecological and policy context

The study area is located along the Australian south-east coastline, incorporating three states – South Australia, Victoria and New South Wales. This region is the most populated in Australia, and is characterised by a major agricultural and mining industry presence. The region is inhabited by around 15.6 million people with three major Australian cities, the capitals of each of the three states: Adelaide (1.32 million inhabitants), Melbourne (4.73 million inhabitants) and Sydney (5.03 million inhabitants), with remaining 4.5 million people living outside these three cities²⁸.

The coastal waters of Australia are naturally low on nutrients and the diverse species and ecosystems in these waters are very sensitive to additions of any (land- or ocean-derived) nutrients. However, the high level of economic activities associated with the regions' populations create an immense pressure on the marine ecosystems. Indeed, the nutrient and sediment inputs in the

²⁷ See Accompanied CD for a “paper” version of the survey, titled “CD_Chapter2_Survey_Version B printout.pdf”

²⁸ See Australian Bureau of Statistics - <http://www.abs.gov.au/ausstats/abs@.nsf/mf/3218.0>

coastal waters have been heavily altered; high levels of pesticides have been found in waters near areas of intense agriculture and mining (State of the Environment Committee 2011). The South East region has been assessed as being in the worst condition out of the whole of Australia in terms of ecosystem health (State of the Environment Committee 2011). It is expected that further degradation can be expected in the next 50 years (ibid).

Parts of the coastal and deep sea waters of the South East are protected through Australian's Marine park network. The protection of Australian marine biodiversity has been legislated through the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act). This cornerstone of environmental legislation provides a legal framework for the protection and management of important flora, fauna and ecological communities in Australia²⁹.

Figure 1: Weedy (left) and Leafy (right) sea dragons



Two species of sea dragons which are protected under the EPBC Act - leafy and weedy sea dragons (Phycodurus equus and Phyllopteryx taeniolatus, respectively, see Figure 1) - were chosen as focal species for the

²⁹ Note that this legislation has been used as a rationale for forced choice, as explained further in this section.

survey. The species populations are specific to the study region and are endemic to Australia, hence representing species of international conservation interest. Although their cryptic nature makes it difficult to estimate the current population numbers or population trends for either of these species, both species are protected under the EPBC Act and both are classified as 'near Threatened' under the IUCN Red list of threatened species. The 'near threatened' status of these species in IUCN Red list classification is mainly associated with the loss of their natural habitat resulting from human activities. Given that the natural habitat of both species is sea grass meadows, the study's valued 'good' has been chosen to be sea grass restoration activity. The restoration activity in the narrative presented to the respondents (see the next section) directly specified the protection of both species as its main purpose.

Sea grasses (see Figure 2) are one of the most productive marine ecosystems on earth providing a number of benefits to people (e.g. Waycott et al. 2009). For example, they provide protection and a nursery habitat for young fish, support commercial and subsistence fisheries, trap and store nutrients and sediments, and store high level of carbon (e.g. Waycott et al. 2009, Orth et al. 2006). Recent research show that sea grasses have the ability (similar to land-based plants) to remove pathogens in their waters that might be dangerous to both humans and other organisms (Lamb et al. 2017). However, sea grasses are under pressure from numerous stressors, including climate change (e.g. though increased sea surface temperatures or frequency and intensity of storms) and, most prominently, changes in water quality such as increased levels of nutrients, contaminants and sediments reaching the coastal environments (Orth et al. 2006). Around 30 percent of global sea grass extent has been lost over the past 100 years and the accelerating rate of loss makes them one of the most threatened ecosystems on Earth (Waycott et al. 2009).

Restoration of sea grass ecosystems is an activity that helps to support and conserve biodiversity, and facilitate provision of ecosystem services, including recreation opportunities.

Figure 2: Sea grass ecosystems



2.1.2 Valuation Scenario

A realistic scenario was developed in order to satisfy one of the key requirements to elicit relevant and realistic values from the respondents (Johnston et al. 2017). Respondents were faced with the following survey narrative. First, the survey briefly described both sea dragon species, their population and the threatened status of their populations, as well as the role of sea grasses in supporting marine life and sea dragons. The text also described that the two species of sea dragons are protected under the Australian legislation (EPBC Act, see previous text). Second, the survey described a degradation of sea grass area of 50ha due to agricultural pollution that would lead to an expected loss of around 500 sea dragons³⁰. Third, a sea grass restoration activity aiming to support the sea dragons' population of same magnitude was described, and respondents were presented with a choice of four alternative locations for restoration. Fourth, the restoration itself

³⁰ This figure was based on a literature review and was calculated using the estimations used in IUCN red list assessment information. See <http://www.iucnredlist.org/details/17177/0> and <http://www.iucnredlist.org/details/17096/0> Browne et al (undated)

was described as comprising of replanting of seagrasses and discontinuing the agricultural pollution by investment in improving the quality of water that enters the sea from towns and rivers. Fifth, two sets of six choice questions were presented to each respondent, one with sites accessible for recreation and one without access to sites (whether access or non-access choice sets were presented first was randomly determined).³¹

2.1.3 Attribute selection

Each respondent was faced with the narrative described above explaining four location alternatives available for sea grass restoration activities (see Figure 3 below). The choice set was designed in order to present alternatives with a number of attributes as explained below.

The study employed simple experimental design and attribute selection to focus on the issues associated with location specifics and effect of community association. While only price attribute has been used in the generation of experimental design (see below), in the data analysis we focused on another two attributes that were implicit in our use of maps. These were related to the community association effect we aimed to analyse – that is related to location of restoration activities. First was related to state where restoration would take place and second to proximity to capital cities in each state. The study was designed in such a way that it was clear to respondents – through use of maps with state border delineations - whether the restoration activities were to take place in their home state or other state; and within each state, whether it was close to each state’s capital city or in a site close to border with another state (see next section). The research idea underpinning this research design was that people might prefer seagrass restoration activity that

³¹ The step four was consequently framed in either access or non-access narrative.

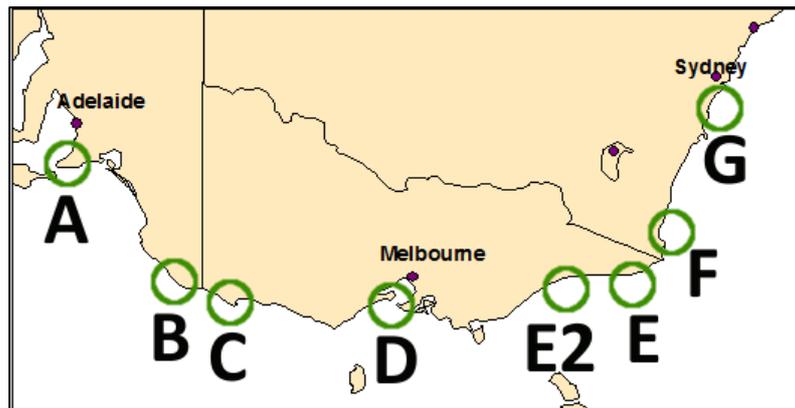
takes place in respondents' state in contrast to when such activity is located other state.³² Such findings have been previously documented in the literature (Dallimer et al 2013 and particularly Rogers and Burton 2017). Similarly we were interested to see whether close proximity to major city or in more remote area elicit any specific preferences, building on common dislike of capital cities in countries around the world.

2.1.3.1 Locations

Eight different sites alongside the south east coastline were selected as potential locations for sea grass restoration (see Figure 3); two locations in South Australia (SA in the following text), four locations in Victoria (VIC in the following text) and two locations in New South Wales (NSW in the following text). Out of these, two pairs of locations were located very close to borders between the three states, and three locations were chosen close to cities (where most of the population resides) and additional location was also selected in VIC (E2; see further below). Four locations were presented to each respondent, these were either locations {A, B, C, D}, {D, E, F, G} or {D, E2, E, G} from Figure 3 below and each respondent saw only one of these quartets of sites.

³² Note that in context of Australia, the state feeling is likely to be expected less strongly felt than, for example, in the context of Germany and France.

Figure 3: Map of a region with all sites available for restoration



Within the four potential locations for restoration activity (see Figure 4), the alternatives always were as follows:

- one close to the main city in the state where initial impact occurred;
- one location where the negative environmental impact took place located close to the border;
- one location across the border to the impact location; and
- one location close to the main city in the neighbouring state.

One of the sites close to border was always a site where original negative environmental impact occurred. The impact site stayed the same for the whole survey (i.e. each respondent faced only one impact location)³³.

Five versions of the survey were developed (labelled B, C, E, F and F2, according to where in Figure 3 the initial negative environmental impact occurred). In each of the versions the respondents were faced with the same impact locations and associated choice set of four sites to choose from for

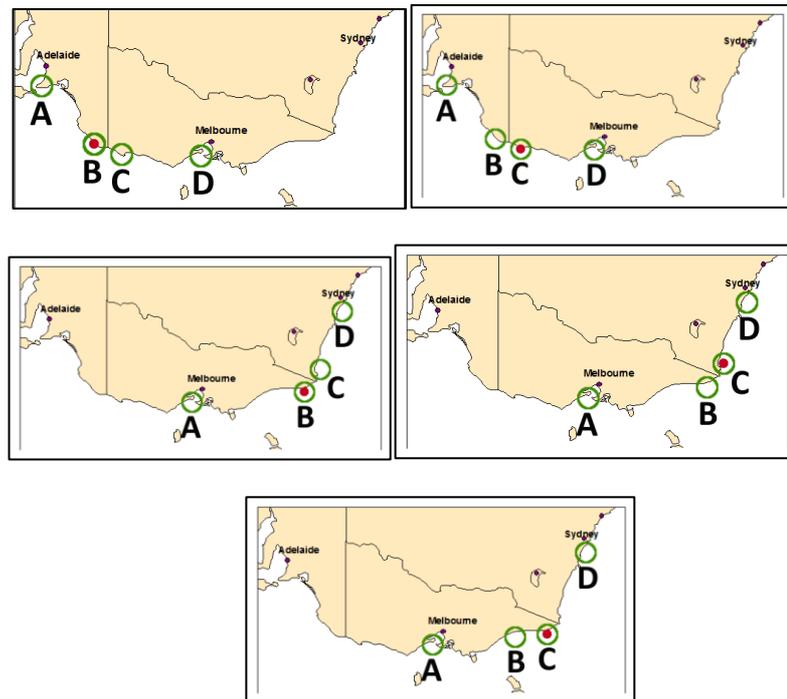
³³ Note that while varying the impact location would be interesting and useful for our research, it was unfortunately technically impossible to do so within the used survey platform.

restoration. For example, in version 4B (see top left choice set map in Figure 4) respondents always faced a scenario in which the environmental degradation occurs at location B and their choice set for locations for sea grass restoration comprised of locations A B C and D.

In addition to first four versions of the survey (called B, C, E, F throughout the text) initially created, a fifth version was added. This version (F2) was created in order to have a scenario where the two 'middle sites' were not located at the borders, but within the state of Victoria only (see last choice set map in Figure 4).³⁴

³⁴ Due to the technical necessity of having each respondent facing only one impact location throughout the survey, it could be expected that the state community association effect (i.e. preference of "my state over other state") would be confounded with the effect of initial impact (as can be seen in results section, respondents had a strong preference for choosing the original site where environmental impact took place). The F2 version hence allowed looking at the effect that negative environmental impact has on the preference for site restoration in relation to another close site without being confounded with the border effect. This version is not the focus in this chapter, however some of the results from this version are reported in the Appendix.

Figure 4: All versions of the choice set maps (left to right: B,C,E,F,F2; Red dot represented original environmental impact)



We took advantage of the physical symmetry of the region regarding the distances between each major cities and borders in between alongside the coast. More specifically, the distances between the three capital cities and borders between the region's states are around 450-550 km when going alongside the coast (Adelaide to border between SA and VIC; this border to Melbourne; Melbourne to border between VIC and NSW; and finally this border to Sydney). This physical symmetry of the region was helpful for the experimental design (see further below).

2.1.3.2 Costs

Each of the seagrass restoration locations had associated costs of the intervention to the respondent in terms of a one off increase in annual tax bill, varying between 5 and 100 AUD (derived from the wider literature and previous valuation research in the area). The payment vehicle was framed using the narrative that it is a governmental responsibility to protect the sea

dragon species (with legal underpinnings stemming from the EPBC Act) and hence no status quo (i.e. no payment) was offered in the choice set. In addition, a question on how feasible respondents consider the payment vehicle to be was included after the choice questions. The study was soft-launched to a pilot of 50 respondents and analysed in order to assess the feasibility of the cost parameters chosen.

2.1.3.3 *Distance*

We used a Stata module together with an input of a postcode from respondents to generate travel distances to each of the sites from each respondents' home location. While we use distance in the analysis it was not displayed to respondents, since the technical capabilities of the survey software did not allow to do so.

2.1.3.4 *Access and non-access*

Two sets of six choice questions with locations were presented - one with the regenerated sites being freely accessible for recreational use and one between locations that were not possible to be accessed by population. This was done in randomised order – i.e. some respondents faced first six non-access choice sets and then six access choices and vice versa. The survey narrative was developed either versions.

2.1.3.5 *State and city attributes*

Both state and the proximity to capital cities were implicit each particular site's location which was used for the analysis reported later in this chapter. While these two attributes were not made explicit in the choice set text, it was clearly visible from the supplied maps on which respondents could consult their choices. These maps hence visually portrayed both the proximity to any of the three capital cities in the region as well as whether the site was in any of the three sites.

2.1.3.6 *No Status Quo question*

In this research we have decided not to include a status quo option. This was justified in the survey narrative, which explained that due to legislative status of the sea dragon protection the Australian government is bound to act on endangerment of sea dragon species. Since the interest of this research was only in preference difference and not WTP values *per se*, the omission of a status quo option did not pose problems for any WTP values estimates.

2.1.4 **Experimental design**³⁵

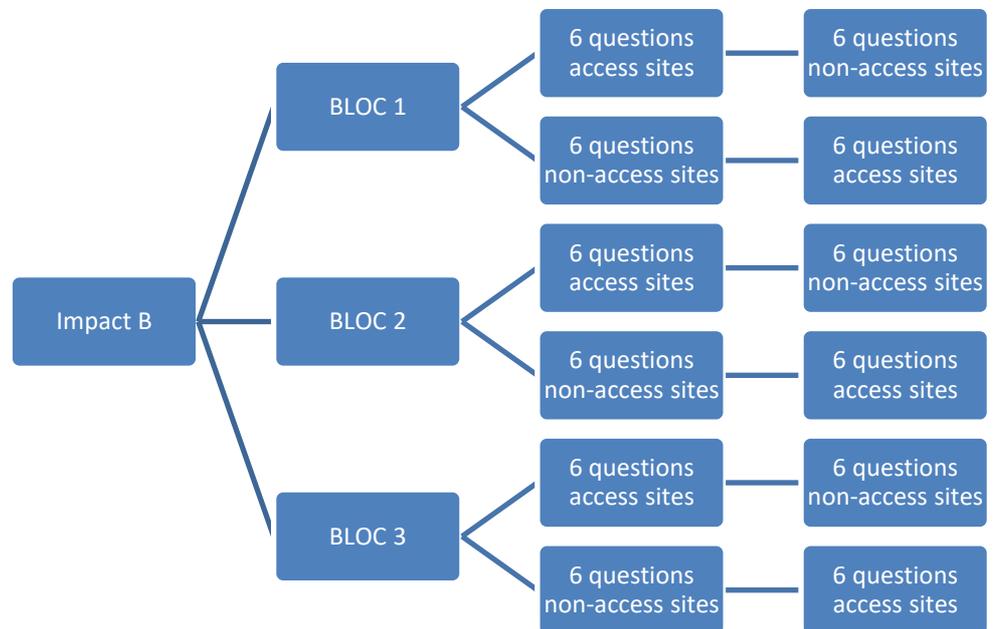
The experimental design was devised using Ngene software³⁶ to create a Bayesian efficient design with ten cost levels. Efficient designs have been increasingly used in the Choice Experiments and aim to provide way how to construct experimental designs with as low as possible standard errors, building on some prior knowledge about the attributes' effects on utility. When such knowledge is present it has been shown that particularly D-efficient designs can be useful for designing choice experiments (Ferrini and Scarpa 2007). Bayesian designs take into account the uncertainty surrounding the "true" value of parameters to be estimated, by assuming a probability distribution in the design generation. In our case, three blocks of six questions were specified and model averaging (Rose et al. 2009) between three utility models with individual utility functions was used to generate a D-efficient design for each version of the survey. The three utility models were prepared for respondents from each of the three states of the South East Australia and these were averaged together to generate an efficient design (Rose et al. 2009). Initial theoretical expectations were used for generating initial design in

³⁵ Note that further background and more detail on experimental design is given in the next chapter.

³⁶ <http://www.choice-metrics.com/features.html>

Ngene. Estimated coefficients from the pilot wave were then used for generating final design in Ngene in a sequential manner (Scarpa et al. 2007). For each version of the survey respondents were randomly assigned to one of the 6 possible “branches” of the survey – see Figure 5 below (please note that this is an example of B version of the survey). The same structure of the design tree was implemented for all five versions of the survey.

Figure 5: Survey versions design – version B (i.e. initial impact occurs at point B; note that the same structure was applied for each of the survey versions B, C, E, F and F2)



2.1.5 Follow up questions

A number of follow up questions were added to the survey (see Accompanied CD for a “paper” version of the survey³⁷). Demographic characteristics, such as household size, education levels, sector of employment and income were recorded. A number of questions further explored the respondents’ association with the location of their residence (i.e. ‘sense of place’), including number of years lived in each of the states, favourite footie (Australian football) and rugby teams. Further, questions on recreational activities, such as past and likely future visits to locations presented in the survey were asked, as well as conditional questions for respondents who selected the same answer 11 or 12 times (in order to identify protest voters). Finally, a number of reflective open ended question on the survey were presented alongside standard questions relating to how certain the respondents were about their answers, which factor was most important for their choices and opinions of the full survey.

2.1.6 Data collection

The survey was developed and consequently programmed into a web-based/online form using Qualtrics software. This was done by the researcher on the online platform and involved development of a number of versions of the surveys and logical functioning that, for example, randomised which survey version was given to a respondent, randomised the order of choice questions, and developing a functionality to ask respondents specific control questions based on their answers in the survey. The resulting survey was then administered by an external company to collect the data from the relevant sample population.

³⁷ The file is titled “CD_Chapter2_Survey_Version B printout.pdf”

The survey was initially pre-piloted with colleagues and few people out of the field in order to identify any obvious problems. Then the survey was piloted on a small number of respondents (n=50) in order to ensure that the chosen cost attribute levels were adequate, to check the adequacy of cost attribute levels, explore potential problems with the survey design and wording, and refine the final experimental design through updating of the Bayesian priors. Following the pre-pilot and pilot versions the survey was fully launched and data were collected over the course of 3 weeks.

2.1.6.1 *Protest voters*

We identified respondents that acted as protest voters and these were removed from the sample in order to focus the data analysis on valid responses (see e.g. Freeman et al. 2014, Powe and Bateman 2004, Brouwer and Martín- Ortega 2012). Identification of protest voters was undertaken by examining the time taken to answer the survey, stated WTP values and a number of follow up questions investigating the motivations of the respondents. 8.7 percent (or 176) respondents were dropped from originally collected 2020 responses (resulting in n=1844), due to overly quick time of survey completion (7 minutes and less) and unrealistically high WTP responses in the questions asking for a direct WTP for a seagrass restoration activity.

2.1.7 Descriptive statistics

Table 1 reports the number of respondents and the proportions of respondents living in each state and major cities in the regions (percentage of population from the region in the state; percentage of state population in the major city; and the same for the survey data). We incurred the location of the respondents from the reported postcode of their residence location. From the

publicly available data we also incurred whether the respondents are living in the metropolitan areas of Adelaide, Melbourne or Sydney.

As we can see, our sample broadly follows the distribution of population in the region, in terms of relative proportions of state populations and the percentage of associated populations living in the three states' capital cities. We can see slightly higher proportion of the respondents from SA (4% higher) relative to populations for the three states and consequently slightly lower proportion (5% lower) of respondents from NSW. We see some differences in terms of the proportions of state populations living in the cities in our sample in comparison to regions statistics (5% higher in SA; 14% lower in VIC; and 14% lower in NSW).

Table 1 : Representativeness of the South East region

	SA	VIC	NSW	Total
State population	1,713,054	6,179,249	7,739,274	15,631,577
Relative proportions	11%	39%	50%	100%
Population in capital cities	1,324,279	4,725,316	5,029,768	10,400,000
% of state population in capital city	77%	76%	65%	
State population (survey)	279	740	823	1,842
Relative proportions (survey)	15%	40%	45%	1
Population in capital cities (survey)	228	458	423	1,109
% state population in capital city (survey)	82%	62%	51%	

Source: Australian Bureau of Statistics¹ and survey data

The summary of socio-economic variables, with mean and associated standard deviation values for each state, is reported in Table 2. While there are similar statistics in terms of gender, age and number of children, minor differences could be seen in terms of education and income between SA and VIC and NSW. This difference reflects the fact that VIC and NSW contain the two biggest cities in Australia, which are more likely to attract more educated

residents and migrants. At the same time, performing Kruskal-Wallis equality-of-populations rank test showed the only significant difference between the three subsample to be in terms of gender ($p = 0.0044$).

Table 2: Summary statistics per state

Variable	Mean SA	St.Dev SA	Mean VIC	St.Dev VIC	Mean NSW	St.Dev NSW	Kruskal-Wallis P-value
gender	1.50	0.50	1.54	0.50	1.60	0.49	0.0044
age	50.89	16.21	48.36	17.30	49.96	17.28	0.3134
Age group	46.45	16.24	44.05	17.13	45.71	17.33	0.5503
education	4.05	2.06	4.39	2.06	4.38	2.11	0.0819
income	4.96	2.59	5.61	2.73	5.49	2.71	0.0673

3 Empirical approach

This section will outline the empirical approach chosen for the analysis of the data. This includes a short overview of choice experiments before setting up details of the statistical model used for the analysis of the data.

Discrete choice models (DCM) or choice experiments (CEs) (e.g. Louviere and Hensher 1983, Louviere and Woodworth 1983) are used to evaluate decision makers' choices among finite number of alternatives. DCMs statistically relate the choices made by the decision makers to decision maker's characteristics and to the attributes of the alternatives available. The models estimate the probability that a decision maker chooses particular option from a set of alternatives. By including costs of alternative as one of the alternatives' attributes, it is possible to estimate an implicit willingness to pay for the remaining, non-cost, attributes. These values, in turn, can be used to compute estimates of the economic implications of policies that could be broadly described in the chosen attributes.

3.1 Choice set

DCMs are based on a concept of the choice set which represent a set of alternatives from which a decision maker in a choice situation can pick one. Within the DCM the choice set needs to have following three characteristics (e.g. Train 2009):

1. The set of alternatives needs to be exhaustive (i.e. it includes all possible alternatives³⁸);
2. The alternatives in the choice set must be mutually exclusive (i.e. choosing one alternative means not choosing another); and

³⁸ Note that by a correct definition (e.g. option "none of the other alternatives") all possible alternatives can be always defined

3. The choice set must contain a finite set of alternatives.

While the first two characteristics are not restrictive and can be satisfied by a correct definition of the alternatives in the choice set, the third one is restrictive. In fact, this condition is a defining characteristic of discrete choice models and distinguishes them from regression models which permit the researcher to examine continuous (and hence infinite) variable(s) (Train 2009).

3.2 Random utility maximisation and modelling strategy³⁹

Discrete choice models are grounded in Random Utility Maximisation theory (McFadden 1973). This assumes that, in considering alternatives, a respondent chooses, with error, that option which is perceived to offer the highest level of utility (U). The utility U for a given alternative is composed of a deterministic part of utility observed by the researcher (V), which is in turn a function of an individual set of preference parameters (β) for observable attributes (x) of the alternative, and a random part of utility (ε) as follows:

$$U = V(x, \beta) + \varepsilon \quad (1).$$

This formulation and assumptions about the distribution of the error term allows researchers to make probability statements about the choice of an alternative over a given set of other options (Train 2009). We adopt modelling approach that is suitable to the panel structure of the data.

The mixed logit (or random coefficients multinomial logit) model (McFadden and Train, 2000) can closely approximate a very broad class of Random Utility models (*ibid.*). It is used to account for the panel structure of CE data and for preference heterogeneity across the sample. The k^{th} -

³⁹ Note that this section is taken from the submitted paper for publication and also used in the next chapter, as it represents my best formulation of the theoretical underpinnings

respondent's utility from choosing alternative i in the j^{th} choice situation can be represented by:

$$U_{ijk} = V_{ijk} + \varepsilon_{ijk} = \beta'_k X_{ijk} + \varepsilon_{ijk} \quad (2)$$

, where X_{ijn} is the set of explanatory variables observed by the researcher (including the attributes of the alternatives and the respondent's characteristics), and ε_{ijn} is an error term that it can be assumed to be iid Gumbel distributed. β'_n represents vector of preference parameters which are individually specific and can be assumed to be either fixed or randomly distributed across respondents. One approach for selecting the appropriate "mixing" (i.e. which parameters should be modelled as random and which as fixed) is to use the Lagrange multiplier test by McFadden and Train (2000) that we used for model specification. Another approach, adopted in this chapter is to treat all variables as randomly distributed to model the preference heterogeneity across population.

3.3 Community association effect:

The primary motivation of this research project was to analyse whether there is a general preference for environmental interventions to occur in the same states that respondents live in, all thing holding constant. This is what we have termed community association effect (working definition). We define community association as clustering of preferences that adhere to spatially relevant boundaries such as state borders, major cities or regions. We test the existence of such an effect in Australia, where such an effect is less likely to occur than, for example, in Europe. Next, this research aims to test whether preferences for sites that are portrayed in the survey as originally impacted will be preferred over others. This expectation is based on previous work of one of the supervisors of this project (see Rogers and Burton). Finally, this projects also aimed to examine whether any specific preferences regarding

major cities in the region will be present in our survey. This was motivated by an observation from living in numerous cities across Europe that there is particular dislike for capital cities in most of countries.

We have a number of research questions that we want to analyse through estimation of the mixed logit models, as described above. Our research interests are as follows:

- 1) Do respondents, holding all things constant, prefer sites that are in the same state as they are? (community association effect)
- 2) Do respondents have any particular preference for sites where original negative environmental impact occurred? (impact effect)
- 3) Do respondents hold any particular preferences for other cities in the region? (city effect)

The following section will present results related to these research questions.

4 Results & empirical strategy

This chapter presents two different views on how to analyse the data. First we analysed the data using a pooled model, redefining the variables in such a way that we estimate a single model (and variants thereof) for the preferences for all residents of South East Australia in the sample. This aimed to examine any general, common, preferences regarding our survey scenario. Second, we also looked at the data from a perspective of each of the three states in the South East. This involved estimating three individual models for respondents from each of the three states. We will report results for each of the approaches in the next sections. Please also note that an extensive exploratory analysis was performed first using conditional logistic regression (McFadden 1973).⁴⁰

4.1 “One model” approach

The first approach adopted to analyse the CE data from this research project is a general one – we use one model for all respondents. Since our intention is to analyse the data from a community association effect point of view, we needed to re-define the collected data in a way that would reflect attributes that relates to whether the sites are in the same state in the region as respondents and likewise for the major cities in the choice set. We hence define attributes for analysis in a way that they *describe the location of the restoration site in relation to each respondent’s state and city*. In other words, the attribute of locations that were implicit in the choice set (i.e. whether the restoration site was in the same state as respondent and whether it was close to a border or

⁴⁰ Available on request from the author.

close to a city) were made explicit for the analysis. This re-definition was done as follows:

- **homestate** = 1 if the restoration site is in respondent's state; = 0 otherwise
- **homestateimpact** (an interaction term) = 1 if the site is in respondent's state and was also originally damaged/impacted; =0 otherwise
- **homestatecity** (an interaction term) = 1 if the site is in respondent's state is also in proximity to the state's capital city; = 0 otherwise
- **otherstate** = 1 if the restoration site is in other than respondent's state; = 0 otherwise
- **otherstateimpact** (an interaction term) = 1 if the site is in respondent's state and was also originally damaged/impacted; =0 otherwise
- **otherstatecity** (an interaction term) = 1 if is in proximity to one of the capital of other than respondent's state ; = 0 otherwise
- **ldist** = log (distance)
- **cost** – cost associated with restoration at particular site, ranging between 5 and 100 AUD

This re-definition allows us testing the research questions set out in the previous part of this chapter. In the new definition of the attributes we highlight three possible characteristics of a site:

- 1) Whether the site is in the same **state** as the respondent, which allows us to test for positive community association with the state of residence;
- 2) Whether the site was originally where the negative environmental **impact** took place; and
- 3) Whether the site is in a close proximity to any of the three major cities in the region, which facilitated testing for positive or negative community association with the capital cities in the region.

Beyond these three characteristics, we control for distance by including a logarithm⁴¹ of distance which was calculated using Stata module that uses Google maps for calculating travel time and distance. Note that, while controlling for it, the effect of distance was not a primary focus of this research. This is due to the fact that the variation in the distance attribute was only present across respondents, due to the way the study was designed (i.e. recall that respondents always saw the same four locations; however note the difference in design with Chapter 3 below).

4.1.1 Mixed logit results

Table 3 below reports the results of a mixed logit model that has been estimated on the full dataset. All parameters are assumed to be randomly distributed and the model allowed for correlation across the parameters. The model was estimated in Stata 13 using 3000 Halton draws for the simulation. In the first half of the table, we can see the estimated mean coefficients for each parameter, while the second part of the table reports the estimated standard deviations of each of the parameters (all assumed to be normally distributed).

⁴¹ We tested multiple specification of the distance attribute with Log yielding the lowest values of AIC and BIC criteria.

What is being estimated is a preference distribution for each of the attributes that represents the preference heterogeneity for each of the attributes across the population.

All of the coefficients reported in the below are highly significant and have the expected signs. Looking at the first half of the table we can see that having a site from a choice set in your state is, on average, preferable for respondents in comparison to having the same site in another state (please note that “otherstate” is used as a base in this model – and the estimated coefficients need to be interpreted relative to it). Note however that this is the mean of a distribution for the parameter – as discussed also below this means

Table 3: Mixed Logit Model

(Pooled sample; All parameters are estimated at the * p<0.001 level)**

	Mean		Standard Deviation	
	Coef.	SE	Coef.	SE
<i>homestate</i>	0.57	0.14	1.39	0.21
<i>homestatecity</i>	-1.50	0.23	5.26	0.33
<i>homestateimpact</i>	0.94	0.15	1.82	0.20
<i>otherstatecity</i>	-1.77	0.12	3.34	0.13
<i>otherstateimpact</i>	0.82	0.10	2.48	0.10
<i>ldist</i>	-0.65	0.14	3.00	0.15
<i>cost</i>	-0.06	0.00	0.07	0.00
N (respondents)	1833			
n (observations)	87,984			
Log-likelihood	-16595			
Halton draws	3000			

that the mean of the normal distribution (with a standard deviation estimated in the second part of the table) is positive and hence shows that more than half of people do prefer sites to be located in his/her sites than not. This we term as a **positive state association effect** for further discussion. We can also see that that initial negative environmental impact matters, as both of the mean coefficients on the interaction terms with impact (*homestateimpact* and *otherstateimpact*) are highly significant and positive. Please note that we

interpret the variables in an additive way. As such, to compare the preferences for sites where impact occurred across the state dimension we need to compare $(\beta_{homestate} + \beta_{homestateimpact})$ against $(\beta_{otherstate} + \beta_{otherstateimpact})$.⁴² The fact that both impact interaction terms show the relatively highest estimated coefficients tells us that, on average, respondents had the strongest preferences for the restoration to take place at the location of the original impact, and this preference was most strongly demonstrated for the impact occurring in the home state. We term this an **impact effect** for further discussion. We also see that the city interaction term is highly significant and negative for both homestate and otherstate. This suggests that on average our sample greatly disliked to restore the sea grass ecosystems in close proximity to major cities in the region, being either in respondent's or in another state of the South East. We term this a **city effect**. We can see that the dislike of cities was more pronounced for "otherstate" cities, as we need to add both $\beta_{homestate}$ and $\beta_{otherstate}$ to each interaction term if we want to compare the two sites being restored one being in respondent's state and one in another state. This means that respondents, on average, most disliked for seagrass restoration to happen in a close proximity to state's capitals, particularly when the state was different to the one the respondent resides in. Finally, the mean cost coefficient and log of distance coefficients are both negative as expected.

In the second half of the table we can see the estimated standard deviation for each parameter. The results suggest that preferences for seagrass restoration in the South East of Australia are highly heterogeneous, with wide distributions for all coefficients. What this means is that while we see the mean

⁴² Note that $\beta_{otherstate}$ is equal to zero in the model reported in this chapter, as this coefficient is used as a base in the model estimation.

effect reported in the previous paragraph, these effects vary widely across the sample population. This can be seen from the magnitude of the estimated standard deviations for each of the random coefficients which are in all cases greater in size than each associated mean value. In particular, notable is a relatively wide distribution for the price attribute, suggesting that for some people positive price was not negatively impacting the probability of choosing a site.

4.2 Three-models approach

Another approach to the data analysis was to analyse the preference data from perspective of each state. The full data set collected from the five versions of the survey was divided into three 'state models' with one model for all respondents from each of the three Australian states – South Australia (SA), Victoria (VIC) and New South Wales (NSW). The same definition of attributes were chosen as in the previous section. This follows a rationale that we might see a difference between the preferences related to restoration of seagrasses in South East Australia (SEA) between the respondents in each state, particularly in relation to the community association effect that we are interested in.

Table 4 below reports the results of the estimated pooled mixed logit model from the previous section and three further models for each of the three states (SA | VIC | NSW). Please note the following: 1) in the top part of the table, the mean coefficients are estimated, with standard errors in parentheses and in red colour are highlighted parameters that were not estimated to be significantly different from zero at a $p = 0.05$ level; 2) in the bottom part of the table, the estimated standard deviations are reported; all but one (in red) standard deviation were estimated to be significantly different from zero at a $p = 0.001$ level.

Table 4: Pooled & States Mixlogit model with correlated coeffs (**RED estimates are not significant a p=0.05 level**).

Pooled & States Mixlogit model with correlated coeffs.				
	POOLED	SA	VIC	NSW
Estimated Mean values (st.error)				
homestate	0.5699*** (0.1372)	1.3212* (0.5318)	0.6014* (0.2432)	0.4074 (0.2792)
homestatecity	-1.5018*** (0.2277)	1.0631 (0.8479)	-1.8650*** (0.3271)	-1.3416** (0.4846)
homestateimpact	0.9359*** (0.1535)	-0.0931 (0.6019)	0.8922*** (0.2092)	1.4133*** (0.3374)
otherstatecity	-1.7674*** (0.1177)	-2.5979*** (0.3013)	-1.1891*** (0.2305)	-1.7321*** (0.1779)
otherstateimpact	0.8225*** (0.1029)	1.2151*** (0.2168)	0.8887** (0.2799)	0.8309*** (0.1337)
ldist	-0.6510*** (0.1425)	0.6036 (0.4409)	-1.1684*** (0.2387)	-0.4943 (0.2599)
cost	-0.0642*** (0.0021)	-0.0767*** (0.0062)	-0.0613*** (0.0031)	-0.0609*** (0.0032)
Estimated Standard.Deviation values (all significant on a *** p<0.001 level, but SD estimate in RED)				
SD (homestate)	1.39	0.58	1.79	3.47
SD (homestatecity)	5.26	8.06	3.60	7.08
SD (homestateimpact)	1.82	4.47	2.43	1.28
SD (otherstatecity)	3.34	4.17	2.64	3.35
SD (otherstateimpact)	2.48	2.56	2.61	2.52
SD (ldist)	3.00	4.87	1.40	4.01
SD (cost)	0.07	0.09	0.07	0.08
n (people)	1833	274	736	823
N (choice sit.)	87984	13152	35328	39504
ll	-16595	-2339	-6639	-7532
aic	33259	4749	13348	15134
bic	33588	5011	13645	15435

*** p<0.001 | ** p<0.01 | * p<0.05

All estimates in RED are not significant on a p=0.05 level

The first observation we can make from the table below is the fact that looking at the separate, state, models shows preference heterogeneity across the three states. While estimated parameters for VIC and NSW are in the same

direction and broadly in same relative magnitudes, the SA respondents differ more significantly from the other two states. This difference is most notable for mean estimated coefficients for *homestatecity*, *homestateimpact* and *ldist*, all of which have been estimated as not significantly different from zero for SA in contrast to other two states. A majority of SA respondents show relatively highest preferences for sites being restored in their state – showing strong positive state association effect – in contrast to other two states’ respondents. This mean effect is less pronounced for VIC and is not statistically different from zero for NSW. While this might suggest that there is no state association effect for NSW respondents, notable preference difference (in favour of homestate) are present in both interaction terms (city and impact interaction terms). Consistently to the pooled model we see that across all three states a relatively highest mean negative preference is present for cities located in other states, however note that this is most pronounced for SA respondents. Similarly also with the pooled model, we can see that where original impact occurs matters, being either in the respondent’s state or elsewhere (for SA the impact site is on par with any site in SA). Notably, only VIC respondents seem to conform on average to distance decay expectations in exhibiting mean negative value for the distance attribute. The mean cost coefficient is negative across the three models, conforming to theoretical expectations.

While the results of the mean estimated coefficients reported above suggest a presence of community association effect, with variation, please note that there is significant preference heterogeneity present in our samples that could be seen in the bottom part of the table. Interestingly, the only parameter for which heterogeneity across the sample was not identified is the homestate attribute for the SA model. This suggests, in line with the mean results, that the SA respondents show the most pronounced community association effect

in contrast to respondents from the other two states. The wide estimated distribution for our other attribute suggests that while the effects discussed above are present on the mean results, the preference heterogeneity is significant. This is not surprising given our simple experimental design and the fact that state associations might not be as pronounced in Australia given little factual importance of state borders for its citizens.

5 Discussion

5.1 Community association effect

Across the results presented in the previous section we provide evidence of an existence of community association effect. At the same time this effect is very heterogeneous across the sample population, exemplified by wider distributions for each of the estimated coefficients. In this section we discuss the evidence in support of the existence of this effect and what this might imply for further research and policy. We will discuss this in turn following the two approaches chosen for an analysis of the data presented in the previous section.

First we will turn our attention to the one model approach. Overall, we can see that majority of respondents prefer that the seagrass restoration sites are placed in respondents' home state in contrast to sites placed in another state of the South East. This effect holds true for sites with impact (i.e. where original, hypothetical, negative environmental impact occurred), sites without impact and sites that are close to cities – in all three cases when comparing a pair of same sites but being in the same state as a respondent and one not, the majority of respondents would be more likely to choose a site in his/her state. Further, we see that respondents – on average – preferred restoration at sites that have been impacted by an initial damage. This might be related to so called “endowment effect” whereby respondents hold stronger preferences for something they feel entitled to discussed further below. Finally, we also see that a majority of respondents held strong negative preferences for restoring the seagrass meadows in close proximity to any capital cities in the region. This effect is coupled with the state association effect - having a restoration site placed in close proximity to a city in another state was seen as the least preferable by a majority of our sample population.

Second, we look at the approach whereby we looked at the data from a perspective of each individual state of the South East. Here the story differs from the above – as we see that some of the average effects differ significantly when broken down for each state. Most notably we see that VIC residents have no general preference for restoration sites located in their home states (i.e. mean of homestate’s distribution is not significantly different from zero at a 0.05 level), while the majority of the NSW and (in more pronounced was) SA residents have a very strong state association effect. Interestingly we do not see the state association effect for NSW respondents, however even for these respondents we see the preference for home state when comparing impact sites and city sites. Further note that while insignificant the estimated mean coefficients for homestate for NSW is still positive.

As can be seen, the combined results from pooled and separate models provide some evidence of the existence of community association effect related to state⁴³. While we see a great preference heterogeneity across the population (see the large estimated standard deviations, especially for homestate, in Table 4) this effect is present at the mean results. In case of the three states models we see a majority of respondents from South Australia and Victoria preferring sites in their states. Similarly, from the estimated standard deviations we know some respondents feel strongly about sites being in their states (from the wide distribution) and vice versa for sites in other states. It is interesting to see this effect – as heterogeneous as it is - in Australia where state borders have very little factual impact since they do not create difficulties for crossing. They do matter administratively and politically, however less than in any other countries for example in Europe. Finding a strong preference

⁴³ Please note that this could also be due to difference in scale parameters (i.e. people can better evaluate sites in their states, but less precisely other states). We will explore this in the future analysis.

for what one can see as “patriotic values” (Dallimer et al. 2014) or preference for home offsets (Rogers and Burton 2017) might have an important implications for policy design, particularly in areas where borders actually matter or where public feelings regarding secession might be important (e.g. the EU continental member states, Scotland or Catalonia in Europe). After all nature is not bound by state borders and often many environmental policies are implemented across borders.

5.2 “Impact effect”

In both of the approaches that we used for data analysis we see that majority of respondents strongly prefer restoration to take place in the locations where the initial negative environmental impact occurred. This effect hold strongly across state borders and subsamples, similar results to Rogers and Burton (2017) who found in their choice experiment regarding biodiversity offsets that “respondents were strongly against locating the offset at a site other than where the impact occurred.” Our results suggest that people might either feel strongly to “repair what has been damaged”. This supports the concepts of an endowment effect (Kahneman et al 1990), whereby people might prefer the restoration to take place at the place of original impact due a loss they experienced that might be bigger than exactly the same gain at another location.

5.3 “City effect”

Across the pooled and state specific models, the majority of respondents show negative preferences for restoration sites in close proximity of capital cities in the region. What is interesting is that this negative preference is similar between residents from the capital cities and other respondents (while not reported in this chapter, we estimated models for each capital residents, and non-residents). This effect holds despite the fact that we control for distance in

our analyses. This suggest that there is the specific characteristic of the location – i.e. being close to one of the major cities – that is causing the significant negative preferences for restoration in these places. One possible explanation is that respondents realise that the restoration of seagrasses close to capital cities might not be effective – given that it is where the most polluted areas are in South East Australia. Further, in most cases the city effect is heightened by the state association effect, making the city locations outside of a respondent’s resident state the least preferred locations for seagrass restoration. This is most pronounced in the South Australian respondents.

6 Conclusions

In this chapter we reported results of the study that aimed to explore preferences for restoration of seagrass meadows in South East Australia. We developed an experimental design using maps to portray choices for locations for such interventions to assess whether preferences were related to the state that a respondent resides in. We hypothesised that respondents might be inclined towards having environmental activities implemented in their state of residence and that we might also see some effect on preferences for restoration of seagrasses in relation to capital cities in the region. By doing so, we attempted to contribute to the field of spatial stated preference research and uncover some specific effects of space on environmental interventions related to capital cities and political boundaries – both effects being limitedly explored in the literature. The majority of the current research in this field focuses on structural aspects of space-preferences relationship, particularly related to a so-called distance decay effect (e.g. Loomis 2000, Bateman et al. 2006). More recently, further complexities associated with space have been evidenced in the stated preference literature. For example, the importance of the location of substitute goods/sites was shown in the literature (e.g. De Valck et al. 2017, Schaafsma et al. 2012, 2013) as well as that the specifics of locations and habitats matters for eliciting values for environmental change (Interis and Petrolia 2016). One could call this the importance of spatial context – meaning that it is increasingly clear that the spatial surroundings of respondents as well as the surroundings of intervention sites might have an important impact on the stated preferences for environmental interventions. However, most of this research does not incorporate political dimension of space into valuation studies. As we showed in this Chapter (and also in the next chapter), political boundaries might be of importance to people in stated choice context.

The findings of this chapter could be summarized as follows. First, our data suggest that political boundaries might matter. Across our analyses we saw some evidence of the preferences for sites in respondents' states in contrast to states in other states in the region. However the fact that borders matter have been to our knowledge evidenced in the literature only few times (e.g. Dallimer et al. 2013; Rodgers and Burton 2017). We provide some evidence of an importance of this effect also in the next chapter of this thesis. More understanding of this phenomenon might be particularly relevant for places like the EU where environmental policies are implemented across borders between Member States countries (e.g. Habitat and Birds directives that are cornerstones of conservation legislation and are increasingly aiming to incorporate ecosystem services in implementation).

Second, we observe that specific aspect of the location of restoration – its proximity to capital cities in the region – also mattered in the choices made by the respondents. Consistently across our analyses we see that locations that were close to any of the three cities were the least desirable locations for seagrass restoration in choice experiment. Given that we controlled for distance in the analyses, this result suggest that it is the specificity of the location that matters, the fact that it is close to a capital city. This might be in contrast to what one would expect – that these locations might be most favourable.

Thirdly we see in our results that people preferred to restore locations that were originally (in our hypothetical scenario) negatively impacted by pollution. This suggest that our sample might show some evidence of endowment effect by preferring to restore “lost site” in contrast to identical site elsewhere. Similar effects have been shown in study similar to ours (Rogers and Burton 2017).

Finally, this research project greatly helped to develop more complex research design presented in the next, third, chapter. In fact, the many lessons learned in the process of designing, implementing and analysing the spatially relevant choice experiment presented in this chapter lies behind the relative success⁴⁴ of the spatial choice experiment presented next. It is also hoped that the lessons learned in this choice experiment will be employed in my future research that aims to further our understanding of how space impacts on people's preferences in stated preference studies.

This research will be taken forward through further analyses of the preference heterogeneity and the use of number of explanatory variables collected in the survey. We will aim to test using a latent class approach that might help to cluster the respondents into groups that might better explain the preference heterogeneity of the preferences than the mixed logit model. Using a number of explanatory variables designed to detect community association effects (e.g. how long respondents have lived in each of the states in Australia and which sports team they favour etc.) we hope to perhaps uncover what are proxies that might explain why people favour "theirs" sites such as ones located in their state.

⁴⁴ E.g. presentation at leading conferences in the field of environmental economics (EAERE, BioEcon, envecon, International Choice modelling conference).

7 Bibliography

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CHAPTER 3: A NEW APPROACH TO CAPTURING THE SPATIAL DIMENSIONS OF VALUE WITHIN CHOICE EXPERIMENTS

Tomas Badura*¹, Silvia Ferrini^{1,2}, Ian Bateman³, Amy Binner³ and Michael Burton⁴

Contributions: The lead author (**TB**) led all the work related to this chapter, however with invaluable inputs from the supervisory team (**SF, IB, AB**) and co-authors (**MB**). Specifically, **TB** led research design, including survey and experimental design, data analysis and project management of the survey implementation, all with inputs from the supervisory team. **SF** and **MB** helped with advice on data analysis. **TB** researched and wrote the chapter with inputs from the team. The study was initiated in 2015 with data collected in summer 2016.

Publication status: A paper based on the research presented in this chapter has been submitted for Journal Environmental and Resource Economics. Some parts of the text in this Chapter are directly taken from this manuscript since they presented my best effort to formulate the research findings. Further, work based on this chapter have been presented in various forms at four conferences in 2017 (Envecon, International Choice Modelling Conference, EAERE, Bioecon) in paper or presentation form, available on request.

Note: The survey presented in this chapter also collected additional data that is not reported in this thesis. This is particularly related to public opinions related to UK's recent decision to withdraw from the European Union. This data is available upon request.

Affiliations:

1. Centre for Social and Economic Research on the Global Environment (CSERGE), School of Environmental Sciences, University of East Anglia, Norwich Research Park, Norwich, NR4
2. Department of Political and International Sciences, University of Siena, Siena Italy.
3. Land, Environment, Economics and Policy Institute (LEEP), University of Exeter Business School, Exeter, UK.
4. School of Agricultural and Resource Economics, The University of Western Australia (M089), 35 Stirling Highway, CRAWLEY WA 6009, Australia

Abstract:

Both theoretical expectations and empirical regularities indicate that individuals' preferences for spatially located goods vary with distance to those goods. Most obvious is the well documented 'distance decay' effect whereby willingness to pay for certain use value goods declines with increasing distance. Understanding how the value of environmental goods and services is influenced by their location relative to respondents can help identify the optimal spatial distribution of conservation interventions across landscapes. However, capturing these spatial relationships within the confines of a stated preference study has proved a challenge. We propose and implement a novel approach to bringing space into the design and presentation of stated preference choice experiments (CE). Using an investigation of preferences concerning land use change in Great Britain (GB), CE scenarios are presented through individually generated maps, tailored to each respondent's home location. Each choice situation is underpinned by spatially optimal experimental designs relevant to the individual's spatial context and current British land uses. To the best of our knowledge, this represents the first case of a CE that integrates space into both the design and presentation of options. We test the effect of our map format for presenting spatial attributes against a commonly applied tabular approach, finding that the former yields both significantly different and more robust preference estimates than the latter. At a time of growing public use of mapping software and applications (e.g. Google/Apple maps) this approach appears to significantly enhance respondents' understanding of key spatial dimensions of goods while also providing analysts with an enhanced basis for the spatial transfer of valuation results and their incorporation within decision making support systems.

1 Introduction and literature review

Increasingly choice experiments (CE; Louviere and Hensher 1982; Louviere and Woodworth 1983) have been used to inform decision makers about preferences for and values regarding potential environmental change (e.g. Johnston et al. 2017). At the same time researchers have stressed the importance of ensuring the validity and reliability of elicited welfare estimates for analyses that assist policy making (Johnston et al. 2017; Desvousges et al 2016). Substantial progress has been made to develop informative and valid CE for environmental decision making. For example, the effect of attributes, levels, number of choice cards and order has been extensively tested (De Shazo and Fermo 2002; Hensher et al. 2005; Day and Pinto Prades 2010; Day et al. 2012, Meyerhoff et al. 2015, Ribeiro et al. 2017), as well as the effect of information (Czajkowski et al. 2016b), sensitivity to scope (Czajkowski and Hanley 2009), design dimensionality and propensity to choose status quo option (Boxall et al. 2009, Oehlmann et al. 2017) or attribute non-attendance (Hussen Alemu et al. 2013, Scarpa et al. 2013).

Spatial considerations are also expected to play an important role in CE concerning the environment as environmental related goods and services are inherently spatial. The physical characteristics of ecosystems vary across space, this affects both their capacity to produce ecosystem services, the potential for these services to generate benefits and value, and the cost of service provision (Fisher et al. 2011, Bateman et al. 2011). For example, water purification or flood protection services only benefit downstream populations while the economic benefits of pollination are confined to nearby agricultural production. Similarly, proximity to human settlements can be a major determinant of certain values - e.g. green space located close to residential areas has the potential to generate large recreational benefits, whereas a

physically similar green space located in remote and/or inaccessible areas provides limited recreational benefits (Parsons 2017). Environmental policies are likely to generate spatial and temporal trade-offs in the provision of ecosystem related goods and services (Rodriguez et al. 2006; Fisher et al. 2009; Fisher and Turner 2008). Understanding of spatial trade-offs between competing land uses and preferences for (potential) changes is crucial for targeting policy interventions with greatest welfare returns for the society (e.g. Bateman et al. 2013). Spatial considerations are clearly fundamental in ecosystem service assessments which is reflected in a growing body of valuation research in this area (e.g. Bateman et al. 2011; Fisher et al. 2011; Johnston et al. 2016).

1.1 Space in choice experiments

The incorporation of space in the modelling of CE is growing fast (e.g. Adamowicz et al. 1994; Brouwer et al. 2010; Campbell et al. 2008; 2009; Czajkowski et al. 2016a; Johnston et al. 2002; 2016; Johnston and Ramachandran 2014; Meyerhoff 2013; Schaafsma et al. 2012, 2013; Schaafsma and Brouwer 2013). The initial interest in spatial aspects of preferences for environmental changes in stated preference valuation studies was predominantly motivated by the definition of relevant markets for aggregation of estimated values (e.g. Sutherland and Walsh 1985, Pate and Loomis 1997). The relationship between distance and value is the underlying driver of travel cost revealed preference modelling (e.g. Bockstael and McConnell 2007). In the context of CEs, initial studies included distance to the valued site, good and/or service in the attribute description (e.g. Adamowicz et al. 1994). The “distance decay” effect (Bateman et al. 2000; Loomis 2000) - the phenomenon of decreasing magnitude of elicited values with increasing distance of a beneficiary to a valued site and/or good/service - has been

documented by an expanding body of literature (e.g. Bateman et al. 2006, Schaafsma et al. 2012, 2013, Liekens et al. 2013). Distance decay has been shown to vary across different goods and sites, users and non-users (e.g. Sutherland and Walsh 1985, Bateman et al. 2000, 2006; Schaafsma et al. 2013) and is influenced by the availability and proximity of substitutes (e.g. Schaafsma et al. 2012, De Valck et al. 2017), with significant implications for value aggregation. Distance decay has been also shown to be influenced by perceptions and characteristics of the good being valued (Andrews et al. 2017). As discussed in Chapter 2, Section 1.4 (“Preferences for politically delineated areas”), recent valuation research has documented differences in preferences for provision of environmental goods across political boundaries. Such difference has been also shown to occur in results presented in Chapter 2 of this thesis.

Spatial context of choices has been increasingly portrayed through maps and graphics (e.g. Schaafsma et al. 2012, 2013; Johnston et al. 2016; Holland and Johnston 2017). Maps and spatial information in general have been shown to influence spatial and non-spatial policy attributes as well as WTP estimates in SP studies (Johnston et al. 2002, 2016). Furthermore, due to practical difficulties in making each map in SP individual to each respondent, only rarely are maps spatially tailored to respondents or show one’s location on a given map (Johnston et al. 2016). Such “individualisation”, however has been shown to be important for respondents to locate themselves on the maps and as having impact on preferences and WTP estimates for hypothetical environmental interventions (Johnston et al. 2016).

1.2 Spatially tailored choice experiment

Building on the above research this chapter presents a novel methodology for incorporating spatial complexity in CE. It incorporates two

spatial attributes in experimental design, one concerning physical distance and the other a political dimension of space. The methodology combines a spatially explicit experimental design that utilises a detailed spatial database of the study region to generate choice set maps tailored to the respondent's location. These maps reflect the actual availability of sites subject to proposed changes. Each map is individualised to the respondent – showing both the locations of hypothetical change and also the respondent's home location. By automating these operations to work in real time tailored spatial information can be generated for each respondent within the survey. Taken together, this approach (to our knowledge) presents the most complete incorporation of space in the design of a spatial stated preference survey to date and one that could be replicated for different contexts. Preferences elicited through this approach are contrasted with those obtained from a separate sample facing the same choices but presented through the more commonly used tabular approach. Estimating mixed logit models in WTP space on data collected under these two modes clearly suggests that portraying spatial choices via a tailored map approach enhances respondent comprehension of options, yielding results which conform more strongly to theoretic expectations.

The chapter has the following structure. The following section provides an overview of the CE survey instrument design and explains the functionality developed for presenting locations in the CE. The third section provides a description of the empirical approach to examine the effects of different presentation modes of choice sets. The fourth section provides results of mixed logit models estimated in WTP space. The final two sections discuss implications of the results, possible extensions and shortcomings and presents conclusions.

This chapter has the following structure. The second section will provide an overview of the CE survey instrument design and explain the novel functionality developed for this research. The third section overviews the data while fourth provides a description of the empirical approach to examine the effects of the novel presentation of choice sets and means for deriving the value transfer function for conservation interventions in agricultural landscapes of GB. The fifth section provides results of initial analysis and sixth of mixed logit models in WTP space. The final section discussed implications of the results for the CE research field and concludes.

2 Methodology: Survey instrument development and experimental design

The application element of the study focuses on valuing conservation changes to high intensity agriculture in Great Britain that has positive impacts on biodiversity. In the United Kingdom, agricultural landscapes cover around 17.1 million hectares or over 70% of its land area (DEFRA et al. 2016) and its intensification over the past four decades⁴⁵ has been the major driver of biodiversity loss in the UK (Hayhow et al. 2016, Burns et al. 2016). The interventions valued in the survey are broadly relevant to agri-environmental schemes which were introduced in the EU and UK under reforms of the EU Common Agricultural Policy to address the increased concern about the negative environmental impacts of agriculture.

The web based CE survey⁴⁶ provides means for assessing the preferences for such interventions by exposing respondents to 12 choices between three locations for intervention sites with varying characteristics and associated costs, with the opt-out option to leave the status of intensive agriculture practice unchanged. The decisions made regarding dimensionality of the choice experiment (i.e. number of choice tasks, number of alternatives, opt-out options etc.) were based on the available guidance and recent literature in the field (e.g. Meyerhoff et al. 2015, Oehlmann et al 2017). The inclusion of an opt-out option is important for deriving valid responses and WTP values and creating incentive compatible choice situations (e.g. Johnston et al. 2017). The web based survey instrument was piloted and refined over the course of 7

⁴⁵ Through dramatic changes in farming practices, including increased use of chemicals, UK agriculture has almost doubled milk and wheat yields since the 1970s (Hayhow et al. 2016).

⁴⁶ Choice of online survey, by itself, over other forms regards attention. For a (rare) overview of the considerations regarding choice of online survey mode see Menegaki et al. (2016)

months – in terms of the survey wording, presentation and experimental design. Four pilot stages of the survey were implemented before reaching the final iteration that was used to collect the data presented in this chapter (i.e. excluding the responses to previous iterations of the survey). The piloting allowed testing the approach, in terms of compressibility and functionality as well as allowing updating our experimental design (see further below). The initial stage of piloting involved in-person implementation of the survey in order to ensure comprehension of the survey and avoid any missing information for the respondents in the described scenarios. In its final iteration, the survey is presented in 10 treatment formats (see Section *Versions of the survey* below) which provides the means for testing the impact of the novel choice set presentation, ordering of questions, the stability of preferences and the impact of how spatial information is presented to respondents. A generous sample of over two thousand respondents allowed to design the survey in multiple versions for future data analysis. This Chapter focuses on the impact of choice set presentation.

In the rest of this section, the development of the choice experiment will be detailed as follows. First, it outlines the valuation scenario and the process of attribute selection. Next, the considerations for experimental design are discussed and then this section details the different stages of survey development.

2.1 Valuation Scenario

The valuation scenario concentrated on two types of land use change from the status quo of intensive agricultural landscape which is a very common land use in the study area. First an explanation of the current British agricultural landscape was given (see Figure 6 below), noting that the majority of this land is high intensive agriculture. The main characteristic of intensive

agriculture was given, its environmental impacts, but also the potential benefits (i.e. high agricultural productivity) were described. Half of the choice questions for each respondent concerned interventions aiming to transform the agricultural landscape into a less intensive one (scenario: AGRI, see Figure 7 below), while the other half described measures to plant new woodlands in the area (scenario: WOOD, see Figure 8 further below). Starting from a common initial land use, the potential change then portrayed two alternative end uses that are substantially different both in terms of their character and the ecosystem services they yield. It is expected that the values that respondents elicit will differ for these different end points. However, while the difference between preferences for the two scenarios is also of (future) research interest, this chapter focuses predominantly on the general impact of spatial information presentation on preferences for environmental interventions. As we believe that these spatial aspect will be present in any spatially-relevant choices, the majority of the analysis presented below considers two scenarios as one, general, change from status quo of high intensity agriculture

Figure 6: presentation of status quo (high intensity

Agriculture covers around 70% of land in the UK. Of this, most is what is termed 'high intensity' farming. The following pictures and table summarise high intensity farming (scroll over the pictures to enlarge them).



High intensity farming	
Amount of food grown	High levels of food produced per acre
Number of farm animals	High numbers of animals per acre
Levels of inputs used	High levels of fertilisers, pesticides and machinery used
Effects on water quality in rivers	Likely to pollute nearby rivers
Effects on wildlife	Main cause of declines in UK wildlife, e.g. significant loss of certain bird species over the past 40 years

The description of the status quo (intensive agriculture), AGRI and WOOD scenarios used similar characteristics and visual support. Each figure

could be inspected in more detail by incorporating zooming ability for each picture presented to respondents whereby hovering over any picture in the survey enlarged it for further inspection by the respondent. The main dimensions of the changes were selected to reflect the major environmental impacts of the proposed changes. This resulted in five main dimensions of change being selected as follows:

- 1) **Amount of food produced** on the site aimed to capture the dimension of agricultural productivity and the main benefit of the status quo – high agricultural yields.
- 2) **Number of farm animals** on the site was a second agriculture productivity indicator and indicator of animal welfare.
- 3) **Levels of inputs used** described the major distinguishing factor of high intensity agricultural landscapes (i.e. higher use of fertilisers, pesticides and machinery) which has negative environmental impact on biodiversity.
- 4) **Effects on water quality in rivers** was used to indicate both a major environmental impacts of high intensity agriculture (related to point 4) above) and also to provide a link to the payment vehicle used in the survey (annual water bills, see Section *Cost Attribute (PRICE)* below for further details)
- 5) **Effects on wildlife** was selected as a second indicator of major environmental impacts of high intensity agriculture and to provide an indicator of the biodiversity benefits of change (see further details in Section *Biodiversity attribute (BIRDS)*).

Each change/intervention scenario was presented as a table with comparison against the status quo across the above five dimensions of change (see Figure 7 and Figure 8). While it was acknowledged that these five axes of

change simplify the complex environmental-agriculture impacts of proposed changes, the initial and further piloting of the survey provided evidence that the respondents understood these descriptions and found them adequate in conveying the changes summarised⁴⁷. This indicated that the scenarios are both adequate and credible, essential requirements for any survey based valuation research. Further, in order to increase the credibility of the survey and its consequentiality, a screen detailing possible use of the results of this study for the policy in the UK in terms of informing the implementation of 25 year plan for the environment in the UK was provided.

Following this initial piloting, significant time was dedicated to selecting adequate figures to represent changes. Consultation with experts was used for the selection of the visual support (with the zooming functionality as explained above). While there might be some effects of framing, any effects were held constant across all respondents.

Figure 7: presentation of low intensity agriculture scenario (AGRI)

One way to reduce the negative environmental impact of high intensity farming is to change some of these areas into **low intensity farming** (see pictures on the right hand column - scroll over the pictures to enlarge them).

In these areas this would lead to the effects described in the table below. These changes take a number of years to have their full effects.

		
	High intensity farming	Change to low intensity farming
Amount of food grown	High levels of food produced per acre	Lower levels of food produced per acre
Number of farm animals	High numbers of animals per acre	Lower numbers of animals per acre
Levels of inputs used	High levels of fertilisers, pesticides and machinery used	Lower levels of fertilisers, pesticides and machinery used
Effects on water quality in rivers	Likely to pollute nearby rivers	Unlikely to pollute nearby rivers
Effects on wildlife	Main cause of declines in UK wildlife, e.g. significant loss of certain bird species over the past 40 years	Increase in wildlife, e.g. greater numbers and varieties of birds

⁴⁷ The interviewer explicitly asked whether there is anything missing in the descriptions of the scenario that respondent find important.

Figure 8: presentation of new woodland scenario (WOOD)

One way to reduce the negative environmental impact of high intensity farming is to plant some of these areas with **new woodlands** (see pictures on the right hand column - scroll over the pictures to enlarge them).

In these areas this would lead to the effects described in the table below. These changes take a number of years to have their full effects.

		
	High intensity farming	Change to Woodland
Amount of food grown	High levels of food produced per acre	No food produced. Timber produced
Number of farm animals	High numbers of farm animals per acre	No farmed animals raised
Levels of inputs used	High levels of fertilisers, pesticides and machinery used	No chemical inputs used, little machinery used
Effects on water quality in rivers	Likely to pollute nearby rivers	No pollution to nearby rivers
Effects on wildlife	Main cause of declines in UK wildlife, e.g. significant loss of certain bird species over the past 40 years	Increase in wildlife, e.g. greater numbers and varieties of birds and wildlife

Each respondent was faced by both scenarios of possible change and associated sextet of choice questions, with randomised order of what came first (i.e. some saw the AGRI scenario first, while others saw the WOOD scenario first). Following each change scenario, respondents were faced with a simple quiz aiming to capture the differences between the status quo and each scenario (see Figure 9 below). This was undertaken for two reasons: 1) to ensure that the respondents understood the main difference between the status quo and each scenario; and 2) to use the number of attempts to answer this question as an indicator of comprehension in subsequent analyses. If a respondent was unable to answer the quiz correctly they were asked to re-review the prior information and answer the quiz again, until the quiz was correctly answered. While this might have caused irritation for the respondent, we saw this simple quiz as essential for further progress in the survey. It was expected that if the respondent read the description provided he/she should answer these quiz questions on the first attempt (with 1/8 probability that the respondent answered the quiz at a first attempt by a chance). Majority of the respondents passed this quiz on a first attempt and

only a small fraction of respondents required high number of attempts to pass it.⁴⁸

Figure 9: quiz question ensuring comprehension of the change scenarios

For each of the statements below please indicate if they best describe high or low intensi

Show more info

Lower level of chemical inputs	High intensity farming	Low intensity farming
Worse water quality	High intensity farming	Low intensity farming
More bird species	High intensity farming	Low intensity farming

2.2 Attribute selection

A combination of literature review, current policy context, consultation with experts⁴⁹ and piloting, was employed as inputs and motivation for the attribute selection procedure. The process was intended to provide three categories of attributes capturing the relevant characteristics of the land use interventions valued in the survey: 1) locational attributes 2) site characteristic attributes and 3) cost attribute. Table 5 summarises the chosen attributes. These are discussed further below while Figure 12 in the Appendix details how the attributes were presented.

⁴⁸ Respondents were allowed to continue with the survey only after the correct answer was selected aiming to ensure the comprehension of the main dimension of change presented in the survey. An option to revise previously given information was available for answering the quiz.

⁴⁹ Ecology/ornithology – Gavin Siriwardena, Environmental Economics and stated preference valuation – prof. Ian Bateman, dr. Amy Binner, Choice modelling and stated preference valuation – Dr. Silvia Ferrini, Prof. Michael Burton.

Table 5: attribute levels

Spatial attributes (binary)

Country	(= 1) if site located in the same country as respondent (= 0) if site located in other country than respondent
Distance	Distance (in miles) from the respondent's home location to the proposed intervention site location

Site characteristic attributes (binary, continuous and categorical)

Access (to site)	(= 1) if site will be accessible for recreation (= 0) if site is closed to the public
Size (of site)	7 ha (small) 100 ha (medium) 400 ha (large)
Birds0	Little or no increase in the number of birds and wildlife already present in the area of the site
Birds1	Some increase in the number of birds and wildlife already present in the area of the site
Birds2	Substantial increase in the number of birds and wildlife already present in the area of the site
Birds3	Substantial increase in the number of birds and wildlife already present; Some increase in the number of species in the area of the site
Birds4	Substantial increase in the number of birds and wildlife already present; Substantial increase in the number of species in the area of the site

Cost attribute (continuous)

Price	Increase in annual water bill per household (£15, £30, £70, £100, £150 or £200)
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2.2.1 Locational attributes (DISTANCE, COUNTRY)

The binary variables, DISTANCE and COUNTRY were selected as relevant location attributes which were used to reflect the spatial dimension of value in the choice experiment. While the first – distance – was meant to capture the physical dimension of space, the second attribute – country – aimed at capturing its political dimension.

A standard measure of distance (given in miles for UK presentation; note that this was converted to km for the purposes of analysis, thereby enhancing the international transferability of findings) was chosen – the Euclidian distance from a respondent's home to each site in each choice situation⁵⁰. This

⁵⁰ While some consideration was given to using travel distance, the computational and logical demands for the underpinning functionality precluded such an option (see details further in the thesis).

attribute captured distance decay (Loomis 2000; Bateman et al. 2000) effects on function for estimated values that can be then used in a value function for scenario evaluation.

A spatial attribute reflecting policy-relevant spatial political units was also included. Given the historical and current context⁵¹, as well as practical dimension of policy making in the UK⁵², the country level attribute was chosen. The attribute chosen was limited to binary “home country” (country=1) or “other country” (country=0), however the data allowed for accounting for each specific country in Great Britain, should it be needed. This attribute was used to capture the potential preference for interventions located within the respondent’s country which has been shown in the CE literature to potentially occur (Rogers and Burton 2017, Dallimer et al. 2014; see Chapter 2, Section 1.4 “Preferences for politically delineated areas” for further details).

2.2.2 Site characteristic attributes

Following a survey of the relevant valuation literature for the project and policy context of the change under consideration, three attributes associated in turn with the biodiversity impact, scale and accessibility to the site of intervention were chosen for inclusion in the analysis.

2.2.2.1 Biodiversity attribute (BIRDS)

Selection of the attribute capturing the biodiversity impact of the proposed changes was motivated by three issues. First, the attribute needed to be relevant and applicable to both of the land use change scenarios under consideration. Second, the chosen attribute needed to capture the broad

⁵¹ Devolution is a relevant topic in the UK. Following a No result of Scottish independence referendum in 2014, the discussions have been renewed reflecting the UK’s vote to leave the European Union.

⁵² See e.g. https://en.wikipedia.org/wiki/Separation_of_powers_in_the_United_Kingdom

changes in wildlife and biodiversity in the area. Third, the design of attributes had to permit both combined and separate assessment of the use and non-use values related to land use change. This latter dimension of value was examined by looking at how the value of biodiversity changes when use is prohibited by making the site inaccessible (see below).

Motivated by a vast literature on the use of birds as indicators of biodiversity (e.g. Gregory et al. 2003; Bateman et al. 2013; Harrison et al. 2014) as well as the very clear strength of preferences regarding birds in Great Britain (for example, the UK Royal Society for the Protection of Birds has well over one million members; RSPB, 2016; similarly UK has one of the most well run volunteer breeding bird survey, see Chapter 5 for further details), bird species were chosen as the major taxa used in the description of biodiversity impacts of proposed changes. This choice was also motivated by the fact that, for both farmland and woodland birds, biodiversity indices currently exist which are widely used in research and monitoring. These indices are also used for reporting on the biodiversity strategy and related targets globally⁵³, EU-wide⁵⁴ and nationally in the UK⁵⁵ (see for further details Chapter 4). Due to difficulties in meaningfully portraying composite indicators in the surveys, we abstained from portraying biodiversity change in the scenario by the indicators *per se*.⁵⁶ Instead, we have decided to describe that the change in bird

⁵³ [CBD Strategic plan for Biodiversity 2011-2020](#) which is built on the set of Aichi Biodiversity Targets ([see here](#))

⁵⁴ Biodiversity Strategy to 2020 (see e.g. [version for the public here](#) or [official legislative decision here](#))

⁵⁵ [UK Biodiversity 2020 policy](#)

⁵⁶ While we did not use the indicators as attributes for portraying the interventions, we designed the survey in a way that the indicators could be used as input to a potentially derived value function. Indeed, as highlighted elsewhere in the thesis, one of the aims of the project was to integrate the economic analysis with the ecological analysis provided in Chapter 4.

populations is likely to indicate broader changes in wildlife. This lead to the textual description of the biodiversity attribute to be worded as the “change in bird populations and wildlife present in the site”.

In order to capture the broad possible changes which the specified interventions might generate, we consulted at length (including an extended period of co-working) with Gavin Siriwardena, ornithology expert at the British Trust for Ornithology, Thetford⁵⁷ (see for example, Siriwardena et al. 2008, 2007; Plummer et al. 2015). Two dimensions of change were selected; (i) the abundance and (ii) the richness of bird species present on site. While abundance relates to the total number of birds (of any or all species) in the area, species richness relates to the number of bird species in the area. It was envisioned that it is likely that the change will occur first on the abundance dimension of existing species while only with more significantly conservation-oriented intervention is the diversity of species likely to change. Since the researcher was of the strong opinion that neither the actual number of percentage of species provide clearly understandable information to most respondents, we opted for a textual description distinguishing between “some” and “substantial” change in “existing” and “new” species present in the area. The some/substantial change in existing species is related to species abundance while the description of some/substantial change in new species relates to species richness. The pictograms used for depicting these levels reflected both the abundance and richness dimensions of change (see Figure 12 in the Appendix).

Please see last section of the chapter for initial thoughts on how such integration would happen, using the policy indicators related to British birds.

⁵⁷ For details please see <https://www.bto.org/about-bto/our-staff/gavin-siriwardena>

In total five categorical levels were used for the biodiversity attribute (see Table 5 above). While not used initially, a “no change” or 0 level of biodiversity change was included in the later stage of survey development. Such a level can be thought of as a very poorly implemented agri-environmental or conservation intervention.

2.2.2.2 *Scope of intervention attribute (SIZE)*

It is a commonly recommended practice to include a parameter related to the scale/scope of the intervention in stated preference valuation studies (e.g. Johnston et al 2017, Carson and Mitchell 1993, Czajkowski and Hanley 2009). This allows testing of whether respondents’ stated values are sensitive to the scope of the intervention which in turn tests a key theoretical expectation and hence indicator of study validity. The scope parameter also enters the estimated value function allowing ready transferral to a range of potential future policy scenarios.

Three levels of project scope were chosen for the intervention description with associated values of 7ha, 150ha and 400ha, respectively. The choice of these levels were partly motivated by the size of the underpinning database used for site selection for the choice sets (see Section *Underpinning spatial database* below) and to reflect the different size of interventions in terms of agri-environmental schemes. In order to ease the comprehension of these relative sizes for respondents within the choice descriptions, these sizes were translated into an equivalent number of football (soccer in US) pitches alongside pictograms of circles of relative sizes⁵⁸. In our subsequent analyses these are labelled as the “small”, “medium” and “large” interventions.

⁵⁸ Small (7 hectares, about 10 football pitches), Medium (100 hectares, about 150 football pitches), Large (400 hectares, about 550 football pitches)

2.2.2.3 Accessibility to public (ACCESS)

Biodiversity (and broadly environment) related goods and services deliver both use and non-use values (e.g. see Chapter 1). The non-use value component could be broadly motivated by either interest in biodiversity existence per se (i.e. existence value), interest in the biodiversity change for current (i.e. altruistic value) or future generations (i.e. bequest value), without any intention of respondent to use the site in question ever in his/her life. In order to disentangle the use and non-use components of value a binary attribute related to accessibility of the sites to public was used in the description of the proposed change options (1=accessible; 0=not accessible). The situation of non-accessibility is credible given some of the designation sites in the UK as well as property rights which frequently prevent the public from entering private property.

2.2.3 Cost attribute (PRICE)

A *cost* attribute was included as standard practice for deriving marginal WTP values for each attribute. The payment vehicle for this survey was chosen to be annual water bills. This was motivated by the literature review showing long term use of this payment vehicle (Bateman et al., 1995; Ferrini et al., 2014), the fact that all residents in the UK are faced by water bills and that water bills are already used as a vehicle for paying for environmental improvements under the Ofwat 2019 Price Review process (Ofwat 2017)⁵⁹.

An initial cost range from £15 to £75 was used for the pre-pilot stage. Analysis of the results indicated that this range was too restricted to avoid a fat tails problem (Bateman et al. 2002). Accordingly six attribute levels were chosen to be in the final survey, varying between £15 and £200 per annum.

⁵⁹ In order to increase the credibility of the payment vehicle an explicit link to water quality was made in the description of the portrayed interventions.

2.3 Experimental design

2.3.1 Background / Theory

This study employed an innovative extension to choice modelling experimental design to incorporate space in three ways: 1) explicit inclusion of locational/spatial attributes in the design; 2) use of model averaging in the experimental design to address geographical realities needed to be accounted for in the design; and 3) use of a sequential approach to experimental design development.

Experimental design is an approach to allocate the values of the attribute levels to the choice sets presented to respondents. Even in simple choice experiments the number of possible combinations of attribute levels are significant. Experimental design theory helps to guide these choices (Scarpa and Rose, 2008). How this allocation is done has an influence on the sample size and/or the expected estimation efficiency (Ibid).

An important paper by Huber and Zwerina (1996) linked the statistical properties of choice experimental data with the econometric models used for estimation on these data. They show that if there are any reasonable nonzero priors for expected coefficients, then these can be used for generating more efficient choice designs.

2.3.2 Experimental design construction

This research used the Ngene software⁶⁰ for generation of the experimental design. Ngene provides a relatively simple coding environment which, after specification of the required parameters of the design (optimisation measure, model/s used for estimation, utility function specification, including attributes and levels used, etc.), generates the

⁶⁰ See <http://www.choice-metrics.com/features.html>

experimental design for the stated choice experiments which populates the choice tasks with attribute levels according to chosen optimisation measure. This research employed the Bayesian D-error prior measure⁶¹ optimised using the iterative, sequential approach considered by Scarpa et al., (2007). This Bayesian approach takes into account that priors are never likely to be known fixed values and are instead treated as random variables with associated, assumed distributions (in our case normal and log-normal distributions). In this way the design is constructed with a controlled uncertainty – i.e. with knowledge of approximate distribution of the parameters as estimated from the various waves of piloting that are undertaken. The sequential approach to design in each step of the piloting takes new estimated parameters and their distributions and includes them in the next phase of the design.

The first version of the survey built on a simple priors D-efficient design with only the expected sign of the attributes' used as priors as follows: country – positive; distance – negative; access – positive; birds – positive; size – positive; price – negative; status quo (sq) - unknown. The country, access and all five bird attributes (no change + 4 levels) were treated as dummy variables in the design, with birds0 (no change) used as a base. Distance, size and price were treated as continuous variables. The design was manually checked and modified to avoid unrealistic choice situations and dominated choices.

As discussed further below, there were in total 4 pilot versions of the survey leading up to the final survey instrument. In an iterative manner each stage of piloting was used as an input for the next version of the experimental design. While the pre-pilot design was guided only by the expected signs of

⁶¹ Although we considered employing the so-called C-efficient design (concentrating on the efficient estimation of WTP values), it was decided to use a D measure. This was due to the fact that the WTP values obtained from this study was not the sole purpose of the research.

effects on utility and literature review, all the further designs employed Bayesian D-efficient designs with priors obtained from the estimates obtained at each previous stage of piloting as new updated priors, in a sequential manner (Scarpa et al., 2007).

The inclusion of locational attributes in the design required a novel approach to that design in order to reflect combinations of distance and country attributes which were feasible to individual respondents. This was done as follows. First, the distance attribute, although treated as continuous, was included as having two levels. These levels were⁶²: 1) “close”, defined as a distance of below 60 miles (roughly 100km) to a specified site, ensuring a good representation of such close distances to parametrise distances over which preferences might shift rapidly, and 2) “far” defined as over 60 miles to a specified site. This “treatment” of distance allowed the functionality for choice set generation as explained below (for each distance bound a random cell was chosen). While theoretically possible, inclusion of more than two levels of distance was felt to be too complex for this application given the other attributes under consideration.

An extension of the model averaging approach to experimental design (Rose et al. 2009) was used to explicitly incorporate both spatial attributes (country and distance) into the experimental design. As mentioned previously, during the design phase the distance between the respondent’s home and a site was included as a simple two level attribute, being either more or less than 60 miles (chosen as a rough limiting distance for what might be considered as a ‘nearby’ site). The country attribute was also treated as having just two levels; whether the site was inside or outside the home country of the

⁶² The actual distance from a respondent to a site was proxied as the average distance for each of these bounds in the experimental design.

respondent. This results in four possible combinations of the distance and country attributes. However, only respondents located 'nearby' the two borders in GB (that between England and Wales, and the border between England and Scotland) can feasibly face all four of these combinations; the combination of 'other country' and 'nearby' being an infeasible choice in other areas of GB. All four possible combinations (alongside other attributes) were considered in deriving the CE design. Two pivot designs were created – one for respondents living 'nearby' borders and the second for living elsewhere - and they were jointly optimised by minimising the Bayesian D-error through the Ngene model averaging capability.

As will be explained in the next section, the design also built on the vast natural science database. This database reflected real world situation across the British landscape in order to provide individuals with realistic options for change.

2.3.3 Underpinning spatial database

The survey was underpinned by a large 2km grid square data set of over 55,000 cells encompassing all of the GB land area and detailing its most recent land use characteristics. This dataset, originally obtained from the Edina Digimap database (<https://digimap.edina.ac.uk/>), was used to classify the current land use in Great Britain in order to identify sites which could be realistically converted through interventions described in the survey. The selection of these sites was done using Geographical Information Systems (GIS) to analyse land use and environmental variables in the dataset that was previously used for the analyses provided in Bateman et al. (2011b, 2013, 2014) and more broadly in UK NEA (2011, 2014) and Natural Capital Committee (2014). Imposing internally discussed assessment criteria in terms of cropland classification and the estimated density of livestock in each cell the final subset

classified as high intensity agriculture identified around 35,000 cells or roughly 63% of the total area. The relatively high occurrence of high intensity agriculture reflects the fact that the UK has around 70% of its land in agricultural use (DEFRA et al. 2016). This database was used to calculate distance from sites to respondents, using the postcodes given by the latter as part of the survey, and to generate the actual choice sets as explained in the next section.

2.3.4 Spatial choice set generation functionality

A crucial functionality and contribution of this valuation survey is how each individually tailored choice set was generated in real-time and how it was presented in a spatially explicit format. This functionality was underpinned by input of respondent's home location, the land use database discussed above and a spatially optimised experimental design. Each choice set was generated in choosing three cells from the spatial database as follows:

- First, the respondent's home location determined whether she received choice sets from the restricted or non-restricted design, respondents within 60 miles of a country border received sets from the unrestricted experimental design (see Section *Experimental design construction* above).
- Second, the "country" attribute limits the available cells from the database for land use scenarios to those either within or outside respondent's country (see Section *Locational attributes* above).
- Third, the cells are further limited by empirical information on the real world location of high intensity agriculture (see Section *Underpinning spatial database* above).

- Fourth, a cell is randomly chosen from the remaining subset of cells from the land use database. The actual distance from this cell to respondent's home is calculated. When this distance satisfies the experimental design levels (i.e. either more than 60miles or, alternatively, less than 60 miles level) the first option of choice card is created and distance displayed. Contrary to this, if the randomly allocated cell fails to satisfy the distance level given by the design the fourth step is repeated until satisfactory result is reached.
- Fifth, the same procedure is repeated for all three land use options displayed in the choice cards (see Figure 10 below).

The above process runs in real time when an individual takes the survey and the requirement for this to happen in a fraction of a second posed a number of programming difficulties which were overcome with a help of professional programmers. The design hence incorporates distance in a two-step approach. In the first step, we allocate adequate design to respondents clustered according to their location. In the second, the locations of alternative intervention sites for each choice set were randomly allocated within each of the two given distance attribute categories⁶³.

The choice cards were visually represented on a map together with respondent's home location (see Figure 10 below). Attribute levels (all but country) of choice experiment options were displayed in tables next to the map. Option specific labels (letters) and colour coding were used⁶⁴ to help

⁶³ The random component in selecting the sites effectively ensured a sufficient variability in the distance variables across the alternatives.

⁶⁴ Colours were chosen in a manner to avoid any common colour-blindness problems. Thanks to Dr. Amy Binner for pointing this out in the design stage of this survey.

respondents more easily associate an option and its attribute levels to its location on the map. Further, to keep respondents reminded of the current intervention scenario (and to distinguish the choice sets across the two scenarios of change), a picture previously used in the scenario description was also displayed alongside each table (with status quo - no change option - displaying the picture used to describe high intensity agriculture). To further assist choices, an information link was also included to provide respondents with the opportunity to consult previously displayed information about each scenario or attribute levels. Each map displayed the names of each country in Great Britain, its capital city and the border between each country.⁶⁵

⁶⁵ While we would see benefit from additional spatial information (e.g. other major cities or satellite type of map) through more detailed spatial information we believed that in the current format this would clutter the choice set image and hence was not included.

Figure 10 – Choice set example

Looking at the map below and tables on the right, please select from LOCATION A, B, C or NO CHANGE, which one you prefer. (please scroll down to see all four options)

Show instructions What do the icons mean? Prefer not scroll to answer the question? Click here.

Choose a location below to select as your answer:

LOCATION A	35 miles	Future land use:
NO RECREATION No Change No Change No Change 10 FOOTBALL PITCHES WILDLIFE PRICE £30		
RECREATION No Change No Change No Change 150 FOOTBALL PITCHES WILDLIFE PRICE £100		
RECREATION No Change No Change No Change 550 FOOTBALL PITCHES WILDLIFE PRICE £100		
NO RECREATION No Change No Change No Change NO SIZE WILDLIFE PRICE £0		

2.3.5 Testing the effect of using maps in the survey

Our expectation is that, by mapping the spatial location of the new site options and their home, respondents will gain a superior understanding of the choice set than that provided by the standard ‘tabular’ approach to CE choice set description provided on the right hand side of Figure 10. Aside from the fact that online software, mobile phones and other navigation aids have greatly increased everyday usage of maps, this approach to presentation contrasts with typical approaches to CE research in three ways. First, in a significant number of valuation studies the spatial context is presented in a somewhat abstract tabular format (i.e. distance only given in numeric form, with no direction information or visual representation; e.g. Adamowicz et al. 1994, Luisetti et al. 2011, Liekens et al. 2013). Second, in cases where maps are

used in CE studies they are rarely individualised by making each respondent's location explicit (see Johnston et al. 2016 for individualised maps; see e.g. Schaafsma et al. 2013 for generic map). Third, due to practical difficulties, only rarely is the choice set generated in real-time (see Bateman et al. 2016 for a revealed preference example).

To assess the effect of this novel approach to choice set presentation, we conduct a split sample comparison of this "Map" based mode of display with a separate "Tabular" treatment identical in all respects (including experimental design and functionality) but from which the map display was omitted, leaving just the standard tabular information used in CE studies (i.e. only the tables and photographs shown on the right side of Figure 10, with the addition of a "country" attribute). Further, at the end of the choice blocks presented to the Tabular subsample, two further choice questions were added. These questions were identical to choices they had already faced but now presented using the "Map" format (i.e. by simply adding the relevant map shown on the left hand side of Figure 10). This subsample were also asked some additional questions regarding their opinions of the Map vs Tabular approach to presentation.

2.4 Survey development and survey versions – iterative revision process

This section provides details on the iterative process in which the survey was developed. In order to fully describe this process, an initial overview of the final survey instrument is provided first, then the description of the different versions of the survey and finally the piloting process.

2.4.1 Overview of the final survey⁶⁶

The final, web-based, survey was piloted a number of times and was consequentially refined over the course of 7 months – both in terms of the survey wording, but also in terms of the experimental design. In order to better understand the steps implemented over this process which are explained further below, first the structure of the final version of the survey is given below.

1. Screening questions (postcode collection) needed for the functionality of the survey;
2. Status quo (i.e. intensive agriculture) description;
3. Scenario 1 of change (randomly assigned WOOD or AGRI) & quiz of comprehension;
4. Detailed step-by-step choice set example build up (to ensure comprehension of the novel format of the choice set and how to answer);
5. Scenario 1 – 6 choice questions (+ Preference stability Test, see e.g. Day et al. 2012; for one third of respondents)
6. Scenario 2 of change (randomly assigned WOOD or AGRI) & quiz of comprehension;
7. Scenario 2 – 6 choice questions (+ Preference stability Test for one third of respondents)
8. Question regarding preferred land use change and reasons for this choice;
9. Follow up survey-related questions (e.g. importance of attributes; protest voter questions etc.);
10. Socio-economic and other follow up questions.

2.4.2 Versions of the survey

Further to the primary aims of this research, the survey was meant to provide a varied dataset to test a range of hypotheses. For this reason 10

⁶⁶ See accompanied CD for a simple “paper” version of the survey titled “CD_Chapter3_Survey_wording_basic.pdf”

versions of the survey were designed (see Table 6 below). First, in a randomised manner respondents were faced with each of the two land use change scenario first to provide possibility to test whether the order of the scenario presentation had any effects on choices. Second, preference stability tests were performed for a subset of the survey versions (Test in Table 6 below). Specifically, one third of the respondents were faced with a repeated first question from the first block of questions (i.e. Scenario 1) at the end of this block. Second third of the respondents had the same test implemented at the end of the second block of choice questions. Finally, a subset of the sample was presented with the exactly same version of the survey and with the same functionality, however with the choice set presented in a traditional Tabular format. This version of the survey also had additional two questions at the end of the choice questions, however now presented in a map format. Also half of the respondents without maps were also faced with the preference stability test as for the maps version. Only the final version of the survey was implemented over all ten versions listed in the table below.

Table 6: versions of the survey

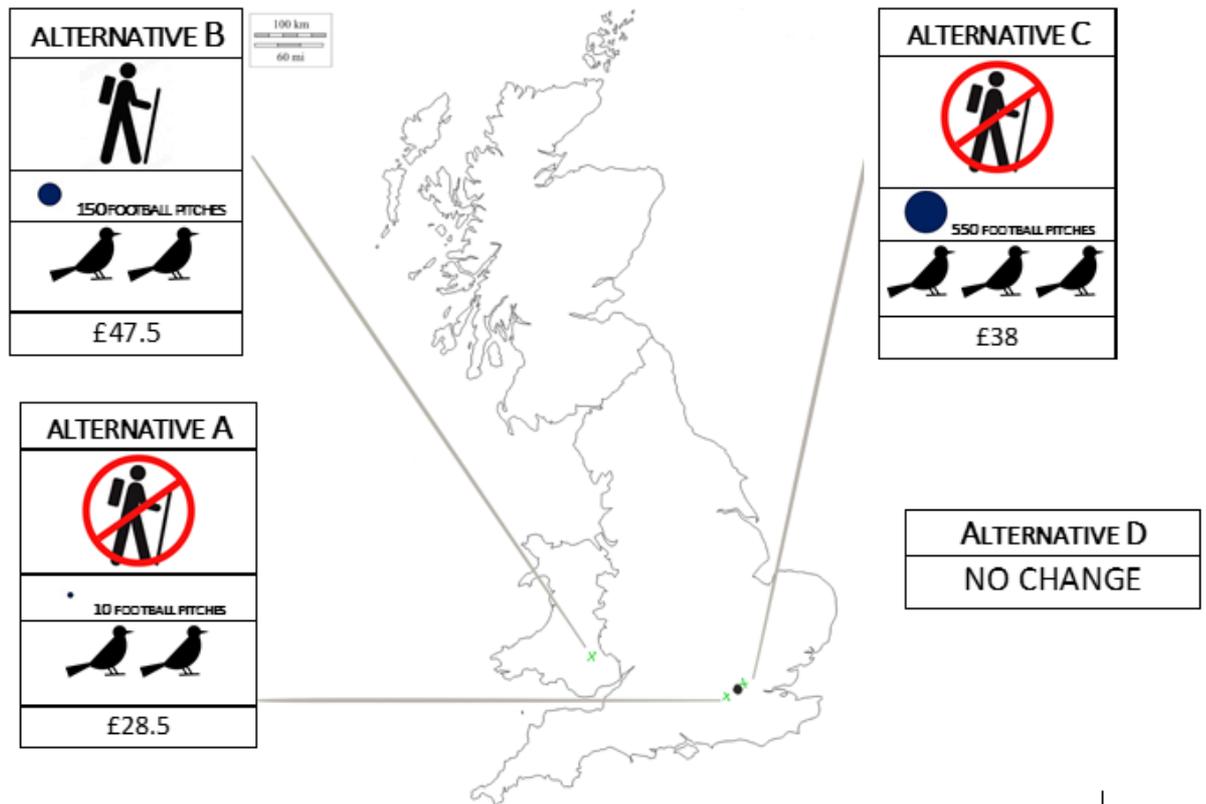
	First block of 6 questions	Test	Second block of 6 questions	Test	Other
MAPS 85% sample	Low intensity Agriculture	n/a	Woodland	n/a	n/a
	Low intensity Agriculture	1st question repeated	Woodland	n/a	n/a
	Low intensity Agriculture	n/a	Woodland	7th question repeated	n/a
	Woodland	n/a	Low intensity Agriculture	n/a	n/a
	Woodland	1st question repeated	Low intensity Agriculture	n/a	n/a
	Woodland	n/a	Low intensity Agriculture	7th question repeated	n/a
NO MAPS 15% sample	Low intensity Agriculture	n/a	Woodland	7th question repeated with MAP	1st question repeated with MAP
	Low intensity Agriculture	1st question repeated	Woodland	7th question repeated with MAP	1st question repeated with MAP
	Woodland	n/a	Low intensity Agriculture	7th question repeated with MAP	1st question repeated with MAP
	Woodland	1st question repeated	Low intensity Agriculture	7th question repeated with MAP	1st question repeated with MAP

2.4.3 Sequential/iterative survey instrument development

The first version of the survey was prepared manually, including the choice set (see Figure 11 below) and was implemented over the course of 25 minutes, on average. This initial version (**PRE-PILOT**) was piloted in person in Norwich and London and allowed an understanding of the respondents' (n=15) comprehension of the survey and estimation of the first model for priors to use in the next phase of survey piloting. In particular, the focus of the PRE-PILOT has been threefold: 1) to verify the survey comprehension; 2) to explore respondents' understanding of the biodiversity/birds attribute; 3) to

explore whether any essential attribute of proposed changes is missing; and 4) to derive priors for the next pilot phase.

Figure 11: Choice set for the first version of the survey



Building on the insights from the PRE-PILOT phase and working with a professional programmer an internet version of the survey was developed over the course of next few months. In particular, the main work over these months have been on the development of the choice set presentation functionality which posed a number of programming challenges (see Section 2.4.3 above). This version (**1ST_PILOT**) was implemented online (n=103) and had the following goals: 1) to test the survey functionality, amount of time spent on answering it and initial feedback from the respondents; 2) to explore potential problems with choice set presentation and general survey functionality; and 3) to derive new set of priors for the next phase of the survey piloting.

Further pilot (**2ND_PILOT**) was implemented (n=507) with the same goals in mind, however adding preference stability testing versions of the survey and refinement of the survey wording.

Final pilot (**3RD_PILOT**) was implemented due to realisation that an additional change to the attributes and experimental design is needed. This involved adding another – “no change” (birds0) – level to the biodiversity attribute and to derive prior for this attribute for the final version of the survey instrument. This additional attribute level was meant to separate the preference for change from status quo per se from change that has some effects on biodiversity.

The final version of the survey instrument was implemented over all 10 versions of the survey (see previous section), including version without map presentation. Further, given the Brexit vote in Britain which happened few days before launching the final version of the survey for full data collection, additional questions related to the future agri-environmental policies in the UK were added to the final version of the survey. This was intended to provide 1) policy relevant data on public opinions related to these topics and 2) to support/strengthen possible policy relevance aimed for this research.

3 Data

A representative sample was recruited by a professional panel provider. In order to collect sufficient sample responses to estimate the Country parameter a fixed quota for responses from England, Scotland and Wales was set at 68%, 16%, 16%, respectively; somewhat oversampling the latter two countries in order to ensure adequate sample sizes. The responses to the final version of the survey were collected online during August and September 2016. The sample was cleared from protest responses (see e.g. Brouwer and Martín-Ortega 2012) using standard approaches designed to avoid problems of scenario credibility failure (Powe and Bateman 2004), as detailed below.

3.1 Sample quality control and pre-analysis consideration

The full sample of collected data (n=2610) was analysed to identify two potential sources of invalid responses – protest voters and respondents who were likely to pay little attention to the survey (distracted respondents). Typically stated preference methods suffer from protesters effects and common guidelines were applied for identification.

3.1.1 Protest voters

The questionnaire included a set of control variables to account for *protest voters* – i.e. respondents which were likely to not believe in either the scenario itself or governmental intervention per se (e.g. Powe and Bateman 2004). The three selected variables (see Table 7) for identification of protest voters were:

- Respondents that stated that the survey was unrealistic in the questionnaire's debriefing questions
- Respondents who only selected the Status quo option and selected that they pay already enough in taxes and charges.
- Open ended answers suggesting a protest voter.

In total 261 (10%) responses were identified as protest voters – see table below.

Table 7: Identification of protest voters

Variable / Reason	# of observations	% of full sample
"Survey is unrealistic"	138	5.29%
"I already pay enough in taxes and charges"	110	4.21%
"I should not have to pay for any changes to high intensity agriculture"	84	3.22%
Open ended answers indicating protest votes	22	0.84%
TOTAL of protest voters (accounting for multiple indication of protest vote from above)	261	10%

3.1.2 Inattentive respondents

Respondents were identified as being *inattentive respondents* using the length of interview time. Based on numerous attempts to answer the survey “as quickly as possible” by the researcher and a number of “test subjects”, a decision was made to exclude the fastest 5% of respondents⁶⁷ as potentially inattentive. This identified 129 respondents fell into this category and were removed from the sample.

3.1.3 Overall exclusions from the sample

In total 372 respondents (14.25%) were removed from the full sample prior to analysis due to either being protest voters, inattentive respondents or

Table 8: Characteristics of removed and retained sample

	full sample	protest	distract	dropped	retained
observations	2610	261	129	448	2238
gender	0.44	0.31	0.43	0.35	0.46
age	49.43	53.62	36.25	48.28	49.62

Gender (1=female; 0=male)

⁶⁷ Those who completed the survey in under 6 minutes 45 seconds.

both. Parametric and non-parametric tests were performed to test the differences between the two subsets of excluded respondents (protest voters & inattentive respondents) and the rest of the sample ([see details online here](#); the log file is available to inspect as Annex 1⁶⁸ on an accompanied CD). Protest voters were statistically significantly different from the non-excluded sample in terms of gender, age and socio-economic group status, and (as expected from the definition of protest voters) by how many times they chose status quo. Inattentive respondents were statistically significantly different from the wider sample in terms of age and income. Table 8 reports differences in terms of the mean age and gender for each group of respondents. On average, protester respondents were more likely to be male and older than respondents in the sample to be kept, while distracted respondents tended to be younger.

3.2 Sample for analysis

Table 9 reports selected descriptive statistics for the two treatments subsamples alongside responses to a number of key questions concerning the survey. Using the two sided ttest, we found no evidence of differences in means for all variables, apart from the percentage of respondents that overall found the survey difficult (p-value 0.047).

⁶⁸ File is titled "CD_Appendix1_CEUK__LOG_final_wave-02summ04b-kwallis-ttest-dropped-only-sign"

Table 9: Sample statistics

	Maps	Tabular	Pooled	Variable description
Sample size:	1911	327	2238	
<i>Socio-economic variables</i>				
Gender	0.45	0.49	0.46	Portion of sample that is female
Age	49.3	51.1	49.6	Respondent age (in 10 year categories)
Income (£/m)	3003	3087	3016	Total monthly household income before tax
<i>Follow up questions</i>				
Policy impact	5.6	5.5	5.6	How likely is this survey to influence policy? (10 = most likely)
Nature visits	2.5	2.6	2.5	How often you make trips to outdoor for recreation (5=every day)
Choices difficult	2.6	2.6	2.6	How difficult it was to make CE choices? (5=very difficult)
Survey long	6.6%	4.3%	6.3%	Portion of respondents finding survey overall too long
Survey difficult	16.9%	12.8%	16.3%	Portion of respondents finding survey overall too difficult
<i>Survey-related variables</i>				
# SQ choices	1.9	1.65	1.86	Number of times respondent chose Status Quo (out of 12 q's)
Scenario quiz	1.61	1.47	1.59	Number of attempts to answer the scenario comprehension quiz
Choice time	21.9	22.2	21.9	Average seconds taken to answer each choice question
Survey time	22.55	21.5	22.4	How many minutes it took to answer the survey

4 Modelling responses: Theory and methods

4.1 Random utility maximisation and modelling strategy

Discrete choice models are grounded in Random Utility Maximisation theory (McFadden 1973). This assumes that, in considering alternatives, a respondent chooses, with error, that option which is perceived to offer the highest level of utility (U). The utility U for a given alternative is composed of a deterministic part of utility observed by the researcher (V), which is in turn a function of an individual set of preference parameters (β) for observable attributes (x) of the alternative, and a random part of utility (ε) as follows:

$$U = V(x, \beta) + \varepsilon \quad (1).$$

This formulation and assumptions about the distribution of the error term allows researchers to make probability statements about the choice of an alternative over a given set of other options (Train 2009). We adopt modelling approach that is suitable to the panel structure of the data.

The mixed logit (or random coefficients multinomial logit) model (McFadden and Train, 2000) can closely approximate a very broad class of Random Utility models (*ibid.*). It is used to account for the panel structure of CE data and for preference heterogeneity across the sample. The k^{th} -respondent's utility from choosing alternative i in the j^{th} choice situation can be represented by:

$$U_{ijk} = V_{ijk} + \varepsilon_{ijk} = \beta'_k X_{ijk} + \varepsilon_{ijk} \quad (2),$$

where X_{ijn} is the set of explanatory variables observed by the researcher (including the attributes of the alternatives and the respondent's characteristics), and ε_{ijn} is an error term that can be assumed to be iid Gumbel distributed. β'_n represents vector of preference parameters which are individually specific and can be assumed to be either fixed or randomly

distributed across respondents. One approach for selecting the appropriate “mixing” (i.e. which parameters should be modelled as random and which as fixed) is to use the Lagrange multiplier test by McFadden and Train (2000) that we used for model specification.

Income heterogeneity across the population often leads to the price parameter being modelled as a random variable. This poses difficulties in deriving WTP values from mixed logit models estimated directly in preference space. An alternative is to derive consistent WTP values as proposed by Train and Weeks (2005). Here, the price parameter is assumed as log normally distributed and the objective is to directly estimate WTP values and their distributions. To achieve this, equation (2) is re-parametrized to derive the WTPs as per (3):

$$U_{ijn} = -\beta_k^{price} \left(\frac{\beta_n^{non-price}}{\beta_k^{price}} X_{ijk}^{non-price} - X_{ijk}^{price} \right) + \varepsilon_{ijn} = -\beta_k^{price} \left(\gamma_k' X_{ijk}^{non-price} - X_{ijk}^{price} \right) + \varepsilon_{ijk} \quad (3)$$

where β_k^{price} is the parameter associated with the price attribute, $\beta_k^{non-price}$ is a vector of parameters for the non-price attributes, ε_{ijk} the error term, and γ_k' the vector of WTPs for every non-monetary attribute. This model is referred to as the mixlogit in WTP-space and is estimated using simulation methods (Train 2009). This is a convenient specification to allow us to compare mean WTP values and their distributions across the Tabular and Map presentation format.

4.2 Assessing impact of spatial information

In order to assess the impact of spatial information on our model, we assume that the deterministic part of utility can be represented by two types of attribute; those that either are (denoted x^S) or are not (denoted x^{NS}) influenced by spatial information regarding the location of sites. Examples of

x^S variables are likely to include both spatial attributes, while price might be a x^{NS} variable. The utility from any spatially located alternative can then be represented as per (2):

$$U = \beta_0 sq + \beta_{ns} x^{NS} + \beta_s x^S + \varepsilon \quad (4)$$

where β_{ns} and β_s represent associated vectors of preference parameters, sq is a dummy parameter that is equal to 1 when the status quo is chosen by a respondent and β_0 captures preferences for the status quo option.

We hypothesise that using different approaches to representing spatial information in a CE may have various effects upon choices. Here we have two spatial representation modes; the Maps and Tabular treatments. These could have different (or no) effect on parameters for each of the sq , x^S and x^{NS} variables. To assess these effects we combine our theoretical representation of impact that spatial information has on utility (4) with the mixed logit modelling approach estimated in WTP space (3). Assuming, now for clarity, that the price attribute is of a x^{NS} type, we arrive at the following re-parametrisation (5):

$$U_{ijk} = -\beta_k^{price} \left(\gamma_k^0 sq_{ijk} + \gamma_k^{NS} X_{ijk}^{NS-price} + \gamma_k^S X_{ijk}^S - X_{ijk}^{price} \right) + \varepsilon_{ijk} \quad (5)$$

In the results section we compare the WTP estimates ($\gamma_k^0, \gamma_k^{NS-price}, \gamma_k^S$) and the price parameter (β_k^{price}) across the Tabular and Map based presentation subsamples. This provides evidence on which attributes from (4) above are influenced (x^S) by an addition of an (individualised) Map to Tabular presentation only, and those that are not (x^{NS}). Also of interest is whether there is a difference in preferences for the change from Status Quo per se (sq) across the two treatments.

5 Results: Initial analysis (Conditional logistic regression)

The following section reports the results of the estimated models. Table 10 reviews definition of each attribute variable used in the analysis. Following an exploratory analysis of different specification for the distance variable a natural log form (denoted *ldist*) was found to provide the best fit to the data.

Table 10 Variables descriptions (see also Table 5)

Variable	Description & Coding
country	1 = home country; 0 = other country
Log(dist)	Logarithm of point distance to sites from home location in kilometres
access	1 = access; 0 = no access
size	Size of site; Small=7ha, Medium=150ha, Large=400ha
birds0¹	1 = Little or no increase in the number of birds and wildlife already present in the area; 0 = otherwise
birds1	1 = Some increase in the number of birds and wildlife already present in the area; 0 = otherwise
birds2	1 = Substantial increase in the number of birds and wildlife already present in the area; 0 = otherwise
birds3	1 = Substantial increase in the number of birds and wildlife already present, Some increase in the number of species in the area ; 0 = otherwise
birds4	1 = Substantial increase in the number of birds and wildlife already present, Substantial increase in the number of species in the area; 0 = otherwise
price	The annual increase in water bills for the household (cost of improvements)
sq	1 = if choice is Status Quo; 0 = otherwise

Initial exploratory analyses were, in accordance with the literature and common practice (e.g. Hensher et al. 2005), prepared using the conditional (fixed-effects) logistic regression (e.g. McFadden 1974). The conditional logit model is well suited to analyse the probability of choice among alternatives as a function of the characteristics of the alternatives. However, it does not take into account the panel structure of the choice data (i.e. repeated choice) nor group the responses per each individual answering the choice questions. It is

also based on a strong independence of irrelevant alternatives assumption⁶⁹, an assumptions which will be later relaxed with further steps in the analysis. Nevertheless, this model still provides valuable insights particularly in terms of an initial understanding of the structure of the choice data and helps to guide further steps in the analysis.

Table 11 provides the results of the full conditional logit model with all parameters in their original forms, including distance specified in km. The conditional logit estimates provide the relative influence of each parameters on the probability of choosing an alternative, holding all other parameters constant. From another perspective they can also be interpreted as contributions to utility relative to other parameters. These contributions to probability (or utility) needs to be interpreted relative to a base, which in our case was chosen to be *birds0* or “little or no change in number of birds and other wildlife present in the area. The *sq* or “status quo” parameter captures the contribution to probability of choice (or utility) of choosing Status Quo option (“no change”).

All estimated parameters are highly significant and their signs conform to prior expectations. The positive coefficient on the Country variable shows

Table 11: Conditional (fixed-effects) logistic regression

	Coef.	Std. Err.	P>z
<i>country</i>	0.087	0.020	0.0000
<i>dist_km</i>	-0.00071	0.000061	0.0000
<i>access</i>	0.522	0.016	0.0000
<i>size</i>	0.000495	0.000041	0.0000
<i>birds1</i>	0.421	0.028	0.0000
<i>birds2</i>	0.652	0.028	0.0000
<i>birds3</i>	0.975	0.029	0.0000
<i>birds4</i>	1.124	0.029	0.0000
<i>price</i>	-0.00956	0.000141	0.0000
<i>sq</i>	-0.361	0.037	0.0000
Log likelihood			-31336.8
Number of obs			107424
Number of respondents			2238
LR chi2(10)			11786.95
Prob > chi2			0.0000
Pseudo R2			0.1583

⁶⁹ Independence of irrelevant alternatives assumes that removing any alternative that was not chosen from a choice will not impact on the chosen alternative.

that sites in the respondents' home country are favoured over those located in other countries. This seems to suggest that the political attribute (country) in our analysis of the pooled sample matters. The negative coefficient on the distance variable confirms that the further a site is from a respondent the less likely they are to choose it as a preferred option. Positive signs on the Access and Size variables confirm that open access and larger sites are preferred over those with no access and/or of smaller extent. Estimated coefficients on the Birds variables show strong positive preferences for sites with more abundant and diverse wildlife, with clear significance across the levels of these variables showing that this is not a simple discrete addition to utility but rather an effect which is sensitive to the scope of the change. The strong negative coefficient on the Price variable confirms to prior theoretical expectations. Finally the negative coefficient on the status quo (*sq*) variable shows the disutility of current intensively farmed landscapes.

5.1 Exploratory hypotheses testing

Using log likelihood ratio tests, the data were examined using the conditional logit model with interaction between distance and country variables. The following initial tests were applied on the data:

- 1) Test1: Farmland vs woodland choices (within respondents test);
- 2) Test2: Order effect - first six vs second six choices (within respondents test);
- 3) Test3: Combined order effect and land use type preferences (across respondents test)
 - a. Order effect on WOOD choices only
 - b. Order effect on AGRI choices only
- 4) Test4: Maps treatment - Map vs no Map survey version (across respondent test)

The core idea of these tests is to compare model coefficients to test the differences in preferences either due to order of questions, type of land use

change scenario or survey framing (maps vs Tabular), with reflection on respondents' socio-economic characteristics. This should indicate the structural differences between preferences (noting again, that this model does not take into account the panel structure of the data). Test 1 and 2 passed with no statistically significant differences between the preferences across land uses and order with further results as follows ([for full details of these tests see online here](#); the log file is available to inspect as Annex 2⁷⁰ on an accompanied CD).

- Test 3 shows that there is a difference in preferences for block1 and block 2 set of Agricultural land use change questions, but no difference for woodland choices. There is no obvious reason and/or expectations for this result.
- Test 4 reports a statistically significant difference between the versions presented with maps and without maps. This is an important result which we see further in the analysis and focus on in the rest of the Chapter.

	1. Farmland vs woodland	2. Order effect	3.a Order effect WOOD	3.b Order effect AGRI	4. Map vs no Map version
Prob > chi2	0.6951	0.1133	0.0887	0.0001	0.0000

When the test reject similarity between the two models two possible interpretations for the difference arises. First, the difference might be due to preference differences. Second, the difference might be due to error variance. In order to test the cause of the differences identified by Likelihood ratio test above, the Heteroscedastic conditional logit model was applied to “Order

⁷⁰ File is titled “CD_Appendix2_CEUK_LOG_final_wave-03analysis01-lrtests-landuse-order-maps”

effect AGRI” and “Map vs no Map version” subsamples. In both cases the independent variable used to model the variance was found significant.

- In case of Maps vs Tabular version the model shows that in the Maps version of the survey present lower error variance. This result is important that it indicates that the noMap version is “noisier” – supporting our expectation that the Maps presentation should elicit better spatial attributes.
- In case of Order effect AGRI subsamples the model shows that there is higher error variance when agriculture related choice questions are presented as a second block of questions (after woodland-related questions) in contrast to when the questions are presented first. This might be due to fatigue, however interestingly this happens only for agriculture related questions.

The differences in preferences between the versions of survey presented with or without map presentation of choice question is highly significant and indicates that the form of presentation of the choice question is influential for respondents’ choices. This result is indicative for further analyses and aimed to be one of the key results of this research. Further analysis of the maps treatment will be provided below.

5.2 Interaction analysis

In order to explore potential sources of preference heterogeneity and, perhaps, justify moves towards more advanced models which are able to account for preference heterogeneity, an interaction analysis with few binary socio-economic and other variables was performed. Selecting a number of binary variables (young vs old, rich vs poor and whether respondents answered both scenario quizzes on a first attempts), the analysis revealed that age and income has influence on some of the parameters (distance, access and

size for age; country, size and sq for income), while the indicator whether respondent answered on the quiz correctly at the first attempt has influence on all but distance parameter. This is likely due to the fact that this variable is likely to proxy attention to the survey. This interaction analysis (Results [available online here](#); the log file is available to inspect as Annex 3⁷¹ on an accompanied CD) informs our understanding about the potential sources of heterogeneity for further analysis and also supports the move towards models which can account for preference heterogeneity.

5.3 Distance specification analysis

The conditional logit model was further used to explore different possible specifications of the distance variable in order to identify the most preferred one. A number of specifications was tested: linear, squared, cubed, log, inverse, inverse squared and tried also step-function (function which takes median values for a selected number of distance categories). The most preferred specification in terms of log likelihood values and commonly used information criteria AIC/BIC was the log specification (the log file is available to inspect as Annex 4⁷² on an accompanied CD).

⁷¹ File is titled "CD_Appendix3_CEUK__LOG_final_wave-03analysis02a-clogits-interactions-binary"

⁷² File is titled "CD_Appendix4_CEUK__LOG_final_wave-04analysis00-distance-specs"

6 Results: testing the impact of Spatial information presentation with Mix Logit Models estimated in WTP space

The below results presents the application of the methodology developed in this Chapter to assess an impact of spatial information presentation in choice situation. As discussed previously in this chapter we contrast commonly used Tabular approach to present spatial choice situation where the choices are presented on our individualised map format. Please note than an extensive exploratory analysis period prior to these results were undertaken, that included estimation of the models in preference space, with numerous interaction terms and with different distance specifications. These analyses informed the final analysis presented in this chapter, however are not included in the thesis.

Using the t-test and Kruskal-Wallis equality-of-populations rank test, no statistically significant differences in socio-economic characteristics could be observed between the Map and Tabular samples. Given this we can compare the estimated WTP coefficients and their distributions across the two treatments.

Table 12 below report results from the mixed logit models estimated in WTP-space for the Map and Tabular treatments. The z-test used in selecting the randomly distributed attributes (McFadden and Train 2000) led us to treat *Country*, *Size* and *SQ* parameters as fixed, while $\log(Dist)^{73}$, *Access*, *Birds1-4* and *Price* were specified as random, suggesting that there is a preference

⁷³ Please recall that following an exploratory analysis of different distance specification of the distance variable, the log (distance) specification was evaluated as best performing for the analysis in terms of AIC and BIC.

heterogeneity for these parameters across the sample population. The models assumed the *Price* parameter to be log-normally distributed and allowed for correlation across the parameters⁷⁴. The models were estimated in Stata 13 using 3000 Halton draws for the simulation.

Table 12: Mixed Logit model of choices in WTP space for Map and Tabular samples (results in GBP per year)

Variable	Maps		Tabular	
	Coef.	SE	Coef.	SE
Mean WTP Values				
<i>Country</i> ^a	2.38	-1.53	20.72***	3.70
<i>Log(distance)</i>	-4.23***	-1.07	6.11 *	-2.2
<i>Access</i>	50.01***	-2.4	73.98***	-5.56
<i>Birds1</i>	66.49***	-3.47	65.16***	-7.51
<i>Birds2</i>	92.82***	-3.84	116.11***	-8.27
<i>Birds3</i>	126.50***	-4.65	165.04***	-10.98
<i>Birds4</i>	138.95***	-4.95	174.27***	-12.14
<i>Size</i> ^a	0.03***	0.000	0.03***	0.000
<i>SQ</i> ^a	-81.84***	-5.48	-20.13	-12.45
<i>Price</i>	-4.25***	-0.03	-4.20***	-0.07
S.D.				
<i>Log(distance)</i>	18.05 ***	-0.81	14.25***	-1.52
<i>Access</i>	49.19***	-2.1	60.95***	-4.68
<i>Birds1</i>	45.96***	-2.92	44.85***	-7.06
<i>Birds2</i>	71.93***	-3.06	79.65***	-6.46
<i>Birds3</i>	108.22***	-3.49	131.85***	-9.13
<i>Birds4</i>	119.16***	-3.62	152.49***	-10.03
<i>Price</i>	1.12***	-0.04	1.16***	-0.08
N (respondents)	1911		327	
n (observations)	91728		15696	
Log-likelihood	-23324		-3762	

*** p<0.001 | ** p<0.01 | * p<0.05

WTP values in GBP per year (1 GBP = 1.13 EUR = 1.34 USD)

^a parameters assumed to be fixed, remaining parameters assumed to be randomly distributed

⁷⁴ The means and standard deviations of the price attribute were re-calculated from the original Stata output, following Hole (2007). For the coefficient covariance matrices for each model please contact the lead author.

The table reports both similarities and differences across the two treatments. Considering first the similarities; consistent with empirical regularities in the literature we find mean WTP values for all biodiversity coefficients (Birds1-4) are positive and increasing with clear scope sensitivity between levels. Also as expected, mean WTP for site Access is positive, suggesting some role of use value in preferences for the land use interventions. The magnitudes of the estimated standard deviations for all Bird and Access coefficients suggest high degrees of preference heterogeneity across both subsamples with slightly wider distributions in the Tabular sample. Mean WTP for the Size coefficient is positive and similar size across the two treatments, suggesting that the change from a small (7ha) to medium (100) and large (400) site sizes portrayed in the survey equate to mean WTP of £2.8 and £11.8 per annum, respectively.

Accepting the above similarities, our results also highlights a number of clear differences between the two treatments. First is the general preference in the two subsamples for change from status quo. Tabular presentation sample has fixed WTP coefficient for SQ estimated as not significantly different from zero, but when respondents are faced with Maps the WTP value is negative. On average, then, Maps treatment respondents have a greater preference for change *per se* from current situation of high intensity agriculture than what we would expect from attribute levels alone in comparison to the Tabular sample. The major differences between the WTP results obtained from the two treatments are, as expected, concerning the two spatial attributes; *Country* and *Log(Dist)*. When faced with Maps the majority of our sample tend to care more about the distance attribute and on average disregard whether it is in the same country as they are. The opposite is true for the Tabular presentation – on average the respondents prefer the site being in the country they reside in and less pay attention to distance in contrast to Maps treatment. In fact, at the mean

of WTP distributions only the Maps treatment shows distance decay (i.e. negative sign of the mean WTP estimate) which is expected from theory and the literature. The mean of the WTP distribution for distance in the Tabular sample is positive. This is contrary to expectations however note large estimated standard deviation suggest that a large number of respondents exhibit distance decay.

Similar effects of the Maps treatment are mirrored in two additional analyses of choices within and across respondents. First, we applied mixed logit models estimated in WTP space on two sets of questions that were repeated among respondents who were in the Tabular sample. These respondent saw two of the choice sets presented in tabular form repeated (after all tabular choice sets) with addition of Maps. The results can be seen in Annex Table A1 and broadly suggest similar effects on WTP estimates as in the subsamples comparison presented in Table 12. The decision models suggest that majority of respondents provided with Maps exhibited distance decay (and strong preferences for change from Status Quo *per se*), while paying less attention to whether the site was located in their country of residence in comparison to the Tabular treatment. In fact, the two Tabular choices per respondents were driven by the Birds, access and price attributes only, with remaining attributes statistically indistinguishable from zero. Comparable inference to the above could be taken from the second analysis (see Annex Table A2) where we further compared the preferences elicited in the single choice question that both Tabular and Map sample respondents saw first.

We asked Tabular subsample that faced both modes of presentations a number of control questions regarding their thoughts on the difference between the two. 71% percent of the respondents faced with both formats indicated that they would prefer to see questions with Maps and only 10.5%

respondents preferred choices in Tabular format only (see Annex Table A3⁷⁵). In a multiple check box question with statements about the two formats 67% of respondents indicated that maps helped them better understand where the sites were and 49% what the distance to the site was. 45% of respondents indicated that maps made the choices more realistic and 40% claimed that the maps made the choices easier in contrast to the tabular approach. Around 25% of respondents suggested that they were likely to choose sites in a specific region and the same proportion claimed that maps made them choose sites that were closer (which seem to be supported in our results). One quarter of respondents said that the Maps did not influence their choices.

⁷⁵ The exact phrasing of the question was as follows: “If you were to have to answer more questions like those about the sites for land use changes, would you prefer them to have maps or not have maps?”

7 Discussion and conclusions

7.1 Impact of choice set presentation mode

The application of the methodology presented in this chapter allowed us to uncover some effects that presenting spatial information in CE (and perhaps SP) have on preferences for environmental interventions. We compared a traditional Tabular approach against a Maps treatment where additional individualised maps are presented to a respondent alongside the attribute tables. Maps have the most notable effects on the two spatial attributes in our study and also seem to induce more frequently choices away from status quo. While the distance coefficients were estimated with wide distributions, the majority of respondents presented with Maps conform more to the theoretical expectation of distance decay effects; this was not the case for the majority of the Tabular subsample. At the same time, the wide WTP distributions for the distance attribute suggest that across both subsamples a large number of respondents tend to prefer sites further away from their residence. This might be related to unobservable motivations (e.g. belief that biodiversity should be restored away from major populations) and requires further attention. Interestingly, the Tabular format seemed to make people choose more on the basis of whether the site was in the country they live in, a similar observation shown in some existing - note tabular-based - CE research (e.g. Dallimer et al. 2014; Rogers and Burton 2017). This observation suggests that an exploration of incorporation of different political boundaries, reflecting how the spatial information is presented, would be a welcome avenue for further research. An interesting extension would also reflect recent literature regarding distance decay heterogeneity across users and goods and in relation to substitutes (e.g. Bateman et al. 2006; Schaafsma et al. 2013), as well as more sophisticated

incorporation of distance in the analysis (e.g. Andrews et al. 2017, Holland and Johnston 2017).

Use value might also play a role in how presentation of spatial information on individualised maps relative to Tabular format. The Tabular sample showed higher Mean WTP (and larger standard deviation) estimates for the access attribute and the two high levels of biodiversity. It is likely that these might be related to potential use value of biodiversity through, for example, recreation. Our results in this sense seem to be broadly in line with research on the impact of individualised map information by Johnston et al. (2016). We hypothesise similarly to this study that the individualised maps presentation might make the respondents aware that the distances to sites might be larger than speculated without maps and therefore decrease with more detailed spatial information. The role of use motivation could be also support that the mean WTP estimates and WTP coefficient estimated for Price and Size attributes were are not different across the treatments. Since even the smallest of the sites presented in the scenarios (7ha) are sufficient for recreation, it is unlikely that spatial information would impact on WTP if the use motivation play a role. We expected price to be invariant to spatial information.

We believe that presenting choice sets on individualised maps is a very relevant way to portray choice tasks for (implicitly) spatial goods. Maps are now in common use in online software and mobile phones (e.g. Google Maps, Apple maps). In our comparison, presenting CE on Maps demonstrate more theoretically consistent results than in a Tabular format only. Additional control questions related to respondents' opinion on the (difference between) two presentation modes are in favour of Mapped format too. However, it seems that the map format might be perceived more difficult in overall which

might be the necessary price to pay for more realistic choices that are likely to elicit robust policy-relevant information.

7.2 Novel Survey design for spatial stated preferences

This chapter presented a novel approach for spatially-relevant choice experiments applied to a case study of assessing preferences for land use change interventions in high intensity agricultural landscapes in Great Britain. It presents a survey instrument that incorporated spatial dimensions of choice in different stages of the CE survey development. It explicitly included physical spatial and political attributes in the experimental design. Choice set options were generated by further taking into account current land use and the respondent's location. Choice sets were presented to respondents through the real-time generation of maps encompassing all of the above spatial information including where each site is as well as where the respondent is located on a map. To our knowledge this is the first time that a CE study has incorporated these multiple dimensions into the design and display of choice sets and thereby into derived values. This approach could be adopted to other contexts to elicit spatially relevant preferences (including Contingent Valuation research), derive value transfer functions and test different modes of presenting spatial information to respondents. Indeed, given that the underlying functionality is independent from the way choice set is presented, this methodology is particularly well positioned to test the impacts of spatial information on WTP estimates as demonstrated in this chapter.

The methodology has potential to be expanded to consider a wider array of spatial complexity. More sophisticated incorporation of distance⁷⁶ and

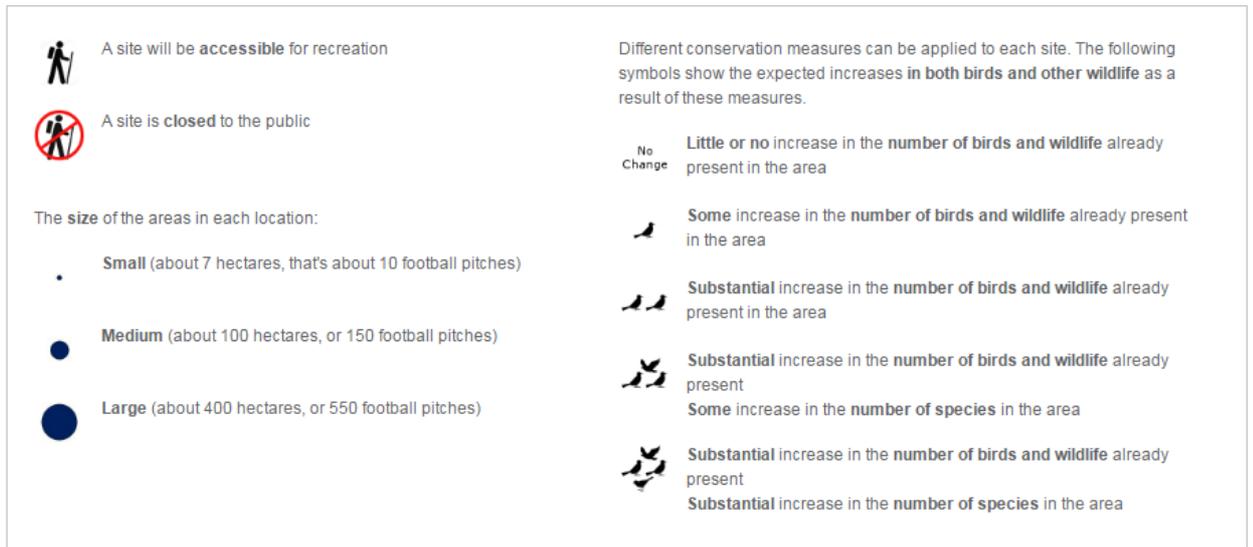
⁷⁶ Note, however, that while the two levels of the distance attribute was featured in the experimental design, respondents were faced with a high variability in the actual distance to intervention sites.

political jurisdictions in the experimental design and treatment of distance in the data analysis (e.g. Andrews et al. 2017, Holland and Johnston 2017) could readily be envisaged. Similarly the resolution of the case study map, while acceptable for national level study, remains too coarse for local level value elicitation. Switching between national and local levels via a mapping interface incorporating a zooming functionality would further enhance the usefulness of this approach. More fundamentally, given the clear impact on preferences of substitutes (e.g. De Valck et al. 2015, Schaafsma et al. 2013, Schaafsma and Brouwer 2013) their availability needs to be incorporated both within the experimental design, and in the presentation of options and analysis of the choice data. Incorporating respondent (and choice alternative) proximity to relevant substitutes is an ongoing focus of our research. Furthermore, just as the nature of new sites could be more rigorously explored in studies, so the spatial distribution of respondents, their sampling and their characteristics (e.g. income) within optimal designs requires further consideration.

Research concerning the incorporation of spatial and related aspects of the environment within CE exercises is growing and is aiming to be of increasing policy relevance and use. The present study examines how these aspects can be both incorporated within study designs and more effectively presented to survey respondents. In a world of increased environmental pressures (e.g. MA 2005, Rockström et al 2009, Butchart et al. 2010, Pimm et al. 2014, Lenzen et al. 2012), relevant and reliable valuation research is increasingly required for ecosystem management and investment decision making. The more that such research can incorporate the realities and complexities of the natural environment the better it will be able to contribute to improvements in such decisions.

8 Appendix

Figure 12: presentation of attributes levels in the survey



The two models in Table A1 below were estimated on repeated choices made by the Tabular subsample under the two modes of presentation. We estimated Mixed Logit Models in WTP where all but price attributes were assumed fixed. This was due to low number of choices that posed difficulties for convergence of the model when randomly assumed parameters were assumed. We used 3000 Halton draws.

Table A1: Mixed Logit model of two sets of repeated choices made under Tabular and Maps formats for the Tabular subsample estimated in WTP space (results in GBP per year)

	Maps		Tabular	
	Coef.	SE	Coef.	SE
Mean WTP Values				
<i>Country</i>	-5.66	16.72	23.55	14.79
<i>Log(distance)</i>	-52.77***	10.12	-12.22	8.15
<i>Access</i>	87.71***	19.27	69.02***	14.92
<i>Birds1</i>	95.07***	26.56	82.03***	22.27
<i>Birds2</i>	124.93***	27.59	105.36***	22.14
<i>Birds3</i>	121.21***	26.58	151.14***	22.97
<i>Birds4</i>	177.38***	28.45	161.75***	23.57
<i>Size</i>	0.09**	0.04	0.05	0.03
<i>SQ</i>	-266.92***	56.71	-73.00	51.25
<i>Price</i>	-4.92***	0.14	-4.74***	0.10
N (respondents)	327		327	
n (observations)	2616		2616	
Log-likelihood	-743		-730	

*** p<0.001 | ** p<0.01 | * p<0.05

WTP values in GBP per year (1 GBP = 1.13 EUR = 1.34 USD)

all parameters assumed to be fixed, apart from price

The two models in Table A2 below were estimated using two conditional logit models (McFadden 1973), on first choices only, for Tabular and Maps subsamples. Note that given the randomisation of task orders, the models were estimated on full design and hence present valid results for our analysis. Further, the first question in a CE exercise has a number of convenient properties, including stronger incentive compatibility (Scheufele and Bennett 2013).

Table A2: Conditional Logit Model estimated on first choices only

Variable	Maps		Tabular	
	Coef.	SE	Coef.	SE
<i>Country</i>	0.15*	(0.07)	0.34*	(0.17)
<i>Log(distance)</i>	-0.15***	(0.04)	0.01	(0.10)
<i>Access</i>	0.53***	(0.06)	0.58***	(0.14)
<i>Birds1</i>	0.40***	(0.10)	0.54*	(0.26)
<i>Birds2</i>	0.63***	(0.10)	0.71**	(0.26)
<i>Birds3</i>	0.77***	(0.10)	1.01***	(0.26)
<i>Birds4</i>	1.00***	(0.10)	1.17***	(0.26)
<i>Size</i>	0.00***	(0.00)	0.00	(0.00)
<i>SQ</i>	-1.09***	(0.25)	-0.05	(0.61)
<i>Price</i>	-0.01***	(0.00)	-0.01***	(0.00)
N (respondents)	1911		327	
n (observations)	7644		1308	
Log-likelihood	-2249.15		-383.434	

*** p<0.001 | ** p<0.01 | * p<0.05

Table A3: Respondents' opinion on Map versus Tabular presentation

Opinion on maps presentation	Number of respondents	Percent
I would strongly prefer questions with maps	134	41 %
...	55	17 %
...	43	13 %
Neither prefer with or without maps	61	19 %
...	15	5 %
...	14	4 %
I would strongly prefer questions without maps	5	1.53 %
Total	327	100%

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CHAPTER 4: LAND USE DETERMINANTS OF FARMLAND BIRD ABUNDANCE IN GREAT BRITAIN

Tomas Badura*¹, Gavin Siriwardena², Silvia Ferrini^{1,3},
Ian Bateman⁴ and Amy Binner⁴

Contributions: The lead author (**TB**) lead all the work related to this chapter, with a crucial supervision from **GS** and **SF**, and further inputs from **IB** and **AB**. Since the author is not of ecological education the patient supervision of **GS** is particularly appreciated. The research for this project was conducted mostly in 2016/2017.

Affiliations:

1. Centre for Social and Economic Research on the Global Environment (CSERGE), School of Environmental Sciences, University of East Anglia, Norwich Research Park, Norwich, NR4
2. British Trust for Ornithology, BTO, The Nunnery, Thetford, Norfolk IP24 2PU
3. Department of Political and International Sciences, University of Siena, Siena Italy.
4. Land, Environment, Economics and Policy Institute (LEEP), University of Exeter Business School, Exeter, UK.

1 Introduction

In face of changing natural environment it is important to understand the impact of human activities on the environment and, in turn, the impact of changing environment on human well-being. Biodiversity - the diversity of living organisms, ecosystems and genes - is of a particular interest, as it underpins ecosystems function and how they provide goods and services for human benefit (MA 2005, Mace et al. 2012). This is both due to anthropocentric reasons (i.e. biodiversity's role in supporting human wellbeing), but also due to intrinsic and/or moral considerations to protect nature for its own sake⁷⁷. By itself, biodiversity is of value to people visiting nature reserves in search of sights of rare, unique and/or charismatic species such as birds. It also plays some, yet not fully understood, role in ecosystems' ability to provide benefits to people (Mace et al. 2012) - for example to support provision of food and timber or ecosystems' ability to purify water and regulate floods and air quality. It hence plays a role in regulating ecosystem processes but also can be a final⁷⁸ ecosystem service itself (Mace et al. 2012). Further, it is becoming clear that biodiversity underpins how ecosystems can absorb external shocks and changes, such as weather extremes associated with climate change (e.g. Isbell et al. 2015, Duffy et al. 2016), and continue their functioning despite these shocks. As such even despite our not-that-complete understanding of its role in ecosystem functioning it is no surprise that biodiversity protection remains one of the key conservation strategy worldwide.

Biodiversity is, however, being lost at a startling rate. Recent authoritative scientific literature is not shy of claiming that the world due to

⁷⁷ Please note that in economic sense, these considerations are categorised under non-use value label and particularly Existence value. See Chapter 1 for further discussion on this topic.

⁷⁸ For further information on what final ecosystem service is see e.g. Fisher et al. (2008, 2009)

human activity entered “sixth mass extinction” (Barnosky et al. 2011) or caused “[a]nthropocene defaunation” (Dirzo et al. 2014). It has been estimated that the “[c]urrent rates of extinction are about 1000 times the likely background rate of extinction” (Pimm et al. 2014). Biodiversity loss has been identified as one of the three crossed planetary boundaries, crossing of which might erode “resilience of major components of Earth-system functioning” (Rockström et al. 2009; Steffen et al. 2015). While the clear urgency of biodiversity protection has recently led to an increased policy response, it has been assessed that the world is unlikely to lead to improved trends in the state of biodiversity (Tittensor et al. 2014).

In the United Kingdom, the trends follow the global (biodiversity loss) steps. Some of the headline messages of a recent State of the Nature report (Hayhow et al. 2016) include the following:

- “Between 1970 and 2013, 56% of species declined, with 40% showing strong or moderate declines. 44% of species increased, with 29% showing strong or moderate increases. Between 2002 and 2013, 53% of species declined and 47% increased. These measures were based on quantitative trends for almost 4,000 terrestrial and freshwater species in the UK.”
- “...we are among the most nature-depleted countries in the world.”
- “The loss of nature in the UK continues.”

The major cause of biodiversity loss in the UK over the past forty years has been policy-driven intensification of agricultural production that led to increasing yields often at the expense of wildlife enabled by the technological advancements (Hayhow et al. 2016; Burns et al. 2016). The significant changes in the management of agricultural landscapes includes, for example, abandonment of mixed farming systems, increased use of pesticides and herbicides or loss of marginal habitats such as hedgerows (Ibid).

1.1 Space & conservation

Given general lack of funding for conservation worldwide (Waldron et al. 2013), an efficient use of available resources is an imperative to halt or reverse the biodiversity loss. Spatial considerations promote effective conservation planning and can boost biodiversity protection in this era of funding austerity. The role of space in understanding values that people hold for environmental policies have been illustrated in the previous chapters of this thesis. Space matters for generating the greatest value from land use change and this is particularly true when considering multiple ecosystem services (Bateman et al. 2013). Spatially targeted conservation measures can offer solutions even in some of the most “hopeless” areas for conservation such as palm oil plantations (Bateman et al. 2015).

The stress on spatial targeting of environmental investments is reflected in UK’s Biodiversity 2020 where the following is stated explicitly:

“A good evidence base is an essential element of delivering the strategy effectively. It will help us make sure we are **doing the right thing in the right place, and using our resources effectively, focusing on action that will have the most impact.**” (Defra 2011)

The recent decision of the UK to leave the European Union has put need for new evidence in a new perspective (e.g. House of Commons, Environmental Audit Committee 2017).

1.2 Use of indicators for biodiversity policy

Given the difficulty of monitoring biodiversity as a whole a common practice is to use composite indicators of species for which data are available. This is particularly helpful for policy context where it is not possible (nor demanded) to evaluate multiple trends in different species at the same time and hence summary indices are useful.

The focus on either single or set of species in a form of targetable indicators is an often-used practice in conservation planning and evaluation. In particular, bird species have been often adopted as relevant indicators for biodiversity monitoring at the national and EU level (see e.g. Gregory et al. 2003, Gregory 2006, Gregory et al. 2008). The relevance of birds for conservation monitoring is partly due to two reasons: first, birds are high in the food chain and hence are likely to integrate the effects of environmental change on a lot of biodiversity “below” them. Second, birds are generally well-monitored and hence data are available to construct better than indices than are possible from data for other groups.

The general topic of use of indicator taxa as biodiversity surrogates is an important and expanding research topic that is key for effective conservation planning and monitoring (see e.g. Garson et al. (2002), Rodrigues et al (2007), Franco et al (2009), Grantham et al. (2010), Leal et al. (2010), Larsen et al. (2012), Breckheimer et al. (2014), Di Minin and Moilanen (2014), Hanson et al (2017), Forest (2017).

1.3 Spatial bird abundance models

This research project develops novel policy-relevant sets of models of biodiversity related bird species hoping to contribute to this body of evidence. Working with the BTO/JNCC/RSPB Breeding Bird Survey (BBS) data, kindly provided by the British Trust of Ornithology (BTO), the models developed within the project focuses on bird species comprising the Farmland bird indicator. The indicator is directly relevant to UK's biodiversity reporting on international and national levels and are used to monitor UK's progress in its Biodiversity 2020 strategy. The project makes use of a rich dataset gathered at the researcher's affiliated institutions. It represents some of the most complete and most integrated sets of relevant land use data assembled to date

previously used for the UK National Ecosystem Assessment and its follow-up stage (UK NEA 2011, 2014). This dataset⁷⁹ is used to provide explanatory variables to model bird abundance numbers obtained from BBS data and other BTO resources. The resulting abundance models provide an opportunity to:

- 1) Test how well the available data can be used to model the selected bird species abundances;
- 2) Assess how these model perform for individual and/or groups of bird species and which variables from our data are driving species abundances;
- 3) Assess how the modelled relationships hold across the two available time periods for which our data is available (2001-2004, 2006-2011);
- 4) Potentially use the models for prediction of bird abundances under different policy and land use scenarios with a possibility to aggregate the outcomes in the associated bird policy indicator.

In line with the rest of the thesis, this chapter is concerning spatial aspects of ecosystem service provision, however this time from an ecological point of view. The previous two chapters concerned the preferences for ecosystem related goods and services and how the potential economic benefits from environmental change are distributed across space. This chapter, in turn, aims to understand how the environmental services – in case of this chapter breeding farmland bird species – are distributed across space in light of land use change in the UK.

This chapter will proceed as follows. The next section provides an overview of the methodology developed for this research project. This

⁷⁹ This dataset was used in Chapter 3 as an underpinning database for generating valuation alternatives in the choice experiment.

includes the context for the research project and initial considerations of the modelling objectives, as well as discussing the data and modelling approach developed for the project. The third section provides an overview of the results, while the fourth section discusses the implications of these results, including the outline of how the models could be integrated with the economic results presented in the previous Chapter. Final section of this chapter summarises the findings and concludes.

2 Research context and methodology development

The first stage of this research project involved articulation of the main objectives of the modelling work. This involved considerations of how to link the resulting models with the valuation results obtained from the survey reported in the previous chapter, assessment of the policy context, consolidation of the available data in order to assess modelling possibilities, and identification of modelling objectives and actual modelling approach. This section will provide an overview of these considerations, including the methodology adopted.

2.1 Integration with economic models from Chapter 3

The objective to integrate the ecological models with valuation study predicated two initial considerations related to scale of modelling and what would be modelled. First, the national scale of the valuation study led us to focus on the same scale in this project. This was possible due to having access to national level dataset in terms of both bird species and explanatory variables related to land use change (see Datasets used below).

Second, the project required a decision on what can be modelled that can be readily integrated with the valuation study. Recalling the valuation study, a hypothetical scenario of changes to high intensity agriculture was portrayed with changes to bird species as one of the main descriptors of such changes. In order to link the two research projects, a focus on changes to bird species as a result of changes to high intensity agriculture was therefore convenient focus. The hypothetical interventions in the valuation study were described in terms of bird species abundance and richness. It was therefore useful to focus on multiple species and their abundances related to agricultural land use change.

These initial considerations provided initial directions for the modelling work. Next, policy context was examined to identify measures of potential interest that this project could focus on.

2.2 Policy context

From the onset this project aimed to derive measure of direct policy relevance. Biodiversity protection is anchored in a linked set of global, regional and national strategies (and associated targets) that underpin most of the current official commitment for conservation. At a global level, world governments have committed to halt the loss of biodiversity as signatories to the Convention of Biological Diversity. Acknowledging that the previous (indeed overly ambitious or perhaps unrealistic) goal to halt the loss of biodiversity by 2010 was not met, the world governments committed in 2010 to [CBD Strategic plan for Biodiversity 2011-2020](#) which is built on the set of Aichi Biodiversity Targets ([see here](#)⁸⁰). At the regional level, the EU built its Biodiversity Strategy to 2020 (see e.g. [version for the public here](#)⁸¹ or [official legislative decision here](#)⁸²) in 2011 in a way that it reflects that commitment made by the EU within the CBD. [UK Biodiversity 2020 policy](#) aims to support these broader international commitments.

UK Biodiversity Strategy 2020 (Defra 2011) aims “to halt overall biodiversity loss, support healthy well-functioning ecosystems and establish coherent ecological networks, with more and better places for nature for the benefit of wildlife and people” and stresses the importance of evidence and

⁸⁰ <https://www.cbd.int/sp/>

⁸¹

<http://ec.europa.eu/environment/nature/info/pubs/docs/brochures/2020%20Biod%20brochure%20final%20lowres.pdf>

⁸²

<http://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:52011DC0244&from=EN>

delivering the strategy effectively including spatial considerations as highlighted above. Following the release of the strategy, a previously used set of indicators had been refined and fit for purpose for monitoring how the Strategy and UK’s international commitments (EU and CBD) are being delivered. Twenty-four indicators for Biodiversity 2020 were released in 2012 and in the 2014 publication “Biodiversity 2020 - a strategy for England's wildlife and ecosystem services: Indicators” (Defra 2014), some of these were further refined. Within these 24 main indicators a number of sub-indicators focuses on bird species, namely: farmland, woodland, wetland birds, water birds and seabirds (see Figure 13).

Figure 13: UK Biodiversity strategy species indicators (source: from Defra 2017)
(Green circles = increase; yellow = no increase nor decrease; red = decrease)

C5. Birds of the wider countryside and at sea	C5a. Farmland birds	 1970–2014	 2009–2014
	C5b. Woodland birds	 1970–2014	 2009–2014
	C5c. Wetland birds	 1975–2014	 2009–2014
	C5d. Seabirds	Not Assessed	Not Assessed
	C5e. Wintering waterbirds	 1975/76–2013/14	 2008/09–2013/14

“UK & England Wild bird Indicators are produced annually for the following - all-species and farmland, woodland, seabirds, water & wetland birds and wintering waterbirds, as official UK Biodiversity and England Biodiversity Strategy indicators.” ([RSPB website](#))

Given that the focus on this project is on agricultural landscape changes, the relevant bird species indicator was chosen to be breeding farmland birds. Breeding farmland birds form sub indicator C5a, which is a status indicator within Indicator C5 Birds of the wider countryside and at sea. The C5 indicator aims to monitor UK’s biodiversity theme “A more integrated, large-scale approach to conservation on land and at sea” (See Defra 2014). The breeding farmland bird indicator is calculated by the British Trust of Ornithology (BTO)

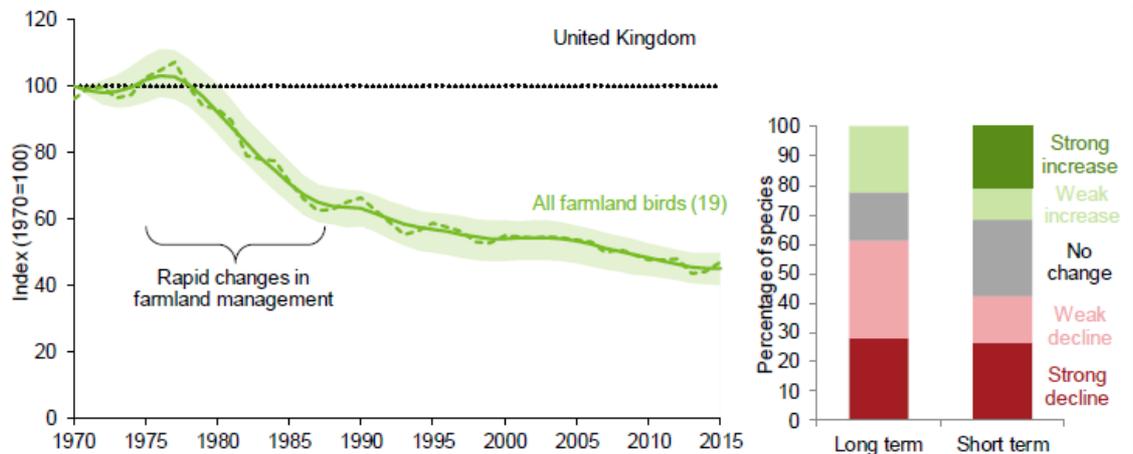
and the Royal Society for the Protection of Birds (RSPB) and is compiled using data from the Common Bird Census (CBC) and Breeding Bird Survey (BBS), provided by BTO, RSPB and the Joint Nature Conservation Committee (JNCC). The indicator is composed of abundance index data for 19 species (see Table 13) and includes birds which are regarded as ‘specialist’ (highly dependent on farmland habitats) and ‘generalist’ (found in a range of habitats, but mostly farmland). Each species is given an equal weight and the index is the geometric mean of the individual species’ annual index values (centred on a common value for the starting year).

Table 13: Breeding Farmland Birds index (source: Defra 2014)

Generalist birds	Specialist birds
greenfinch (<i>Carduelis chloris</i>)	corn bunting (<i>Emberiza calandra</i>)
jackdaw (<i>Corvus monedula</i>)	goldfinch (<i>Carduelis carduelis</i>)
kestrel (<i>Falco tinnunculus</i>)	grey partridge (<i>Perdix perdix</i>)
reed bunting (<i>Emberiza schoeniclus</i>)	lapwing (<i>Vanellus vanellus</i>)
rook (<i>Corvus frugilegus</i>)	linnet (<i>Carduelis cannabina</i>)
woodpigeon (<i>Columba palumbus</i>)	starling (<i>Sturnus vulgaris</i>)
yellow wagtail (<i>Motacilla flava</i>)	stock dove (<i>Columba oenas</i>)
	skylark (<i>Alauda arvensis</i>)
	tree sparrow (<i>Passer montanus</i>)
	turtle dove (<i>Streptopelia turtur</i>)
	whitethroat (<i>Sylvia communis</i>)
	yellowhammer (<i>Emberiza citrinella</i>)

Figure 14 below provides an assessment of the indicator for the past 40 years. The Breeding farmland bird Indicator has been in continuous decline for the past 45 years - in fact, the value of the indicator was half in 2015 of its 1970 value (see left hand side of Figure 14). The colour bars on the right hand side of Figure 14 show the trend assessment of individual species over the short and long term. Indeed, breeding farmland birds overall fared the worst in the UK from all bird species considered in the UK biodiversity indicators and it is therefore a useful and policy relevant indicator for the analysis in this chapter.

Figure 14: Historical status of Farmland breeding birds (source: from Defra 2017)



Notes:

1. The figure in brackets shows the number of species.
2. The line graph shows the unsmoothed trend (dashed line) and smoothed trend (solid line) with its 95% confidence intervals.
3. The bar chart shows the percentage of species within the indicator that have increased, decreased, or shown no change, based on set thresholds of annual change.

Source: British Trust for Ornithology, Defra, Joint Nature Conservation Committee, Royal Society for the Protection of Birds.

2.3 Datasets used

The project will focus on combining two datasets. These comprise a detailed UK dataset to which the researcher has access from his home institutions and the bird abundance data provided by the British Trust of Ornithology. This section will discuss each in turn.

2.3.1 Land use data

Firstly we use a large and comprehensible spatial UK dataset of environmental, atmospheric and land use and land cover data previously employed for UK National Ecosystem Assessment (UK NEA 2011, 2014) and related analyses (see e.g. Bateman et al 2013, 2014). The dataset was constructed from multiple sources of data that were processed and combined together for the purpose of ecosystem assessments related to land use changes

in Great Britain. The dataset was constructed for two target periods 2000-2004 and 2007-2010.

The dataset contains over 55,000 spatially referenced 2x2km squares with variables that describe the portions of different land uses in each cell. The variables are listed in Table 13 below, together with their definitions. For the full detail on the dataset, please refer to Bateman et al. (2014), particularly Annex 1 and Annex 2 (the Annexes taken from the study are available in an accompanied CD⁸³). There is a number of points related to the variables used for the analysis, as follows:

- For the analysis we have only used the land use variables and omitted some of the climate and topographic variables that we had at our disposal. This choice is driven by the fact that most land uses (and especially agricultural) also depend on climate and topographic variables (e.g. some crops are grown only in warmer areas or in areas that are greatly exposed by the sun) and that most climate effects on bird abundance within a limited geographical area like the UK are likely to be mediated by variation in land-use.
- Countries that are known to be dominated by winter cropping were assumed to have only winter cereals and those known to be dominated by spring cropping were assumed to have only spring cereals.

⁸³ See file titled "CD_Bateman_et_al_2014_annexes.pdf"

Table 14: Disaggregated land use definitions (caveats/ restrictions in parentheses)

Name	Description
COAST	coastal margins
FWATER	freshwater
MARINE	sea and estuary
URBAN	urban and other developed land
PERMG	permanent grassland i.e. >5 yrs
TEMPG	temporary grassland i.e. <5 yrs
RGRAZ	rough grazing
GRSNFRM	semi-natural grass or mountains, moors and heaths where NOT used for farming
FWOOD	farm woodland
NFWOOD	woodland NOT used for farming
WHEAT	wheat
WBARLEY	winter barley (England and Scotland only)
SBARLEY	spring barley (England and Scotland only)
OTHCER	other cereals (includes oats and other cereals for combining)
POTS	potatoes
WOSR	winter oilseed rape (where available)
SOSR	spring oilseed rape (where available)
MAIZE	maize (Scotland 2004 is within 'othcrps')
SBEET	sugarbeet
OTHCPS	other crops and bare fallow (includes oilseed rape for Wales; includes maize for Scotland 2004)
HORT	total horticulture
TBARLEY	total barley (Wales only)
TOSR	total oilseed rape (where seasonal data unavailable)
OTHFRM	other farmland e.g. roads, buildings, yards, ponds and, where appropriate, setaside
OCEAN	ocean (area that is not covered by land is given 'ocean' by default)
SHE	Total sheep and lambs
CATT	Total cattle

2.3.2 BTO / bird abundance dataset

The dependent variable – abundance of selected bird species – was derived from two datasets from the British Trust of Ornithology. The counts for the two periods were taken from the BTO/JNCC/RSPB Breeding Bird Survey (BBS) data. BBS is a national volunteer survey that aims to monitor changes in the breeding population of widespread breeding bird species in the UK at the national scale. It is a robust citizen science survey, with developed methodology of sampling and a long time series, as it has run for over 20 years.

In the BBS, each year a random sample (stratified by human population density) of over 3,500 1km squares are visited by large number of skilled

volunteers. In the most recent BBS survey 3,837 squares were collected and covered by 2,796 volunteers (Harris et al. 2017). Birds are recorded in three distance bounds from walks along two 1km transects per each visit. There are two visits to each square one between 1 April and 15 May and a second between 16 May and 30 June. A number of other variables are recorded alongside bird counts, including the dominant habitat or other (mammal) species seen.⁸⁴ The BBS data are widely used for research in the UK⁸⁵ and are used for constructing policy indicators, such as the UK Biodiversity strategy indicators (see Policy context above).

BBS data are collected using a low-intensity, but standardized protocol, designed to provide reliable inference on population changes at large spatial scales. At the scale of the survey square, the sampling method inevitably leads to high stochastic variation in the counts of individual species and the apparent presence of uncommon species. This results from, e.g., weather or the visibility and noise at the time of the survey (limiting potential ability of the volunteer to count some species that are present but not observed at the given time). Hence, for this project, data from multiple adjacent years were combined to estimate “true” counts for a focal, central year. The principle of this approach was that true abundance varies little from year to year (as opposed to over the long term) and that adjacent years can therefore be viewed as repeat sampling events of a particular, stable, local population, such that combining these samples reduces the stochasticity in the final estimate. There is an obvious assumption that there has not been strong change in real

⁸⁴ In order to get familiarized with the data, I took part in a trial BBS data with my supervisor. It was a very useful experience that provided some practical understanding of the data processing explained further below.

⁸⁵ For a list of research articles using the BBS data see here: <https://www.bto.org/volunteer-surveys/bbs/bbs-publications/publications>

abundance over the sampling period, but the sampling approach can also be viewed as estimating background abundance over that period of change; given that large changes in the absence of major changes in local land-use are likely to be due to major extraneous influences, such as weather conditions, local summaries may also be the most appropriate estimate of how abundance responds to land-use and habitat.

In detail, farmland bird BBS data were processed, using the same approach as for the National Ecosystem Assessment (Bateman et al 2014) by extracting counts for all years in which surveys had been conducted in a given square for five-year periods centred on 2005 and 2011. Maximum counts were then calculated per square and year within each period, excluding extreme outliers. The latter was done because, especially for flocking or migratory species, extreme counts can be made on particular occasions because of chance events on migration or late persistence of wintering flocks. Unusually high, outlier bird counts (totals of birds not recorded as in flight) for each square-species combination were identified and excluded as follows for all species: if a species had a ratio of maximum to median count of over 20, taking early and late visit counts into account across the whole BBS dataset, the counts greater than the 99th percentile were flagged. If one of the two counts from a given year were flagged in this way, the other, lower count was used and the flagged value discarded. If both counts were greater than the 99th percentile, then the lower value was used, unless both counts were greater than twice the value of the 99th percentile, in which case no count for that species was included for that square in that year (note that the latter occurrence was extremely rare). This process aimed to exclude records that were unreliable as indices of local breeding densities whilst retaining genuine extreme values that are likely to be informative of bird communities in unusual habitats. After this process, the maximum of the remaining early and late counts for a given square in a

given year was taken as the count for that square and year. The maximum count across years within each period was then used for the modelling reported below. Given the difficulty in observing many of the species of birds and the removal of outliers, the maximum count is likely to represent well the bird abundance (and therefore, across species, richness) in the area.

While zero counts in BBS data do not provide absolute evidence of a species' absence, because detectability can never be 100% across an entire 1km square from a limited transect, zeroes can be interpreted as showing (at most) very low abundance for diurnal species, especially after counts from multiple years are combined as described above. However, from the perspective of modelling habitat selection and land-use impacts, it should be noted that absences from some squares are likely not to be informative because the squares are located outside species' ranges. For example, habitats such as deciduous woodland and arable farmland occur in northern Scotland, but species like nuthatch *Sitta europea* and turtle dove *Streptopelia turtur*, respectively, are absent, presumably because of climatic suitability. Hence, such extra-limital squares would best be excluded from modelling of land-use relationships. To identify such "uninformative zeroes" in the dataset, I have used another dataset, the BTO Bird Atlas 2007-11 data (Balmer et al. 2013). Collecting the Atlas data was a major undertaking, intended to map ranges of bird species across the UK between 2007 and 2011 - that is where the species were present in the UK at the time and where they were not. During the breeding seasons over this four-year period, at least eight of the 25 2x2km squares in every 10x10km square in Britain were surveyed using a standard protocol to provide complete, standardized data on relative breeding abundance for all species at the 10km square scale. This information was used to create actual zeros in the BBS dataset: in BBS squares that had no count for a given species, but which were lying within the geographical range from the

Atlas for that particular species, zero counts were created. In turn, a missing value was confirmed (i.e. squares were not included in the analysis) if no count was recorded but the square lay outside the range indicated by the Atlas data for the species.

2.3.3 Dataset compatibility

The two datasets used in this project differ in their scale – one being on a 2km and second on a 1km scale. In the following remainder of the chapter we take an assumption that the land use proportions from our land use dataset represents the land use proportions at the 1km scale of the BBS data.

2.4 Empirical approach

2.4.1 Literature

This research builds on the previous work of Siriwardena and Hulme for UK NEA UK NEA (2013)⁸⁶. In this work a model of bird species richness was developed using Breeding Bird Survey (BBS) and using same datasets as this project. In the approach presented in this chapter we model individual species with a potential aim to combine these into an indicator. This provides a more sophisticated, complementary analysis, because the individual species making up overall species richness will each respond to environmental variation differently and, hence, relationships between land-use and species richness are more likely to be explained mechanistically by collating species-level models than by modelling derived species richness *per se*.

Modelling bird species takes into account land use (or land cover) variables, climate variables or both. A not-exhaustive selection of these studies include the following. Sohl (2014) model citizen data collected for bird

⁸⁶ Namely Section “3.11 The biodiversity module” in Bateman et al (2014)

populations using in the US, using land cover data from IPCC scenarios, two climate variables and three topographic variables. Modelling the presence of species. Johnston et al. (2013) develop statistical models that link climate to the abundance of internationally important bird populations in north Western Europe. Their spatial climate–abundance models were able to predict 56% of the variation in recent 30-year population trends.

A number of studies used the BBS data to model bird species in the UK. Renwick et al. (2012) uses generalised linear models (GLMs) to model species abundance of two northerly- and two southerly-distributed bird species from BBS as a function of climate in Great Britain. Pickett and Siriwardena (2011) also use GLMs and BBS data to look at the influence of habitat heterogeneity on farmland birds. Siriwardena et al (2011) also uses GLMs to model impact of habitat features on bird abundance of farmland birds. Massimino et al (2015) model spatially explicit species indicators for Farmland and Woodland birds in a two-stage process. First they estimate the likely detectability of each species. In the second stage, they model observed species counts in a Generalised Additive Model (GAM) framework as a function of spatial and habitat variables, using the estimated detectability as the offset.

We aim to contribute to this body of research. In particular, we aim to expand Siriwardena and Hulme’s work for the UK NEA, while aiming to model species that comprise the existing index, similarly to Massimino et al (2015) but not considering detectability.

2.4.2 GLM as Modelling approach:

This project adopted Generalised Linear Models in combination with Information Theory approach to select best models by model averaging techniques. Generalised Linear Models are often used in modelling (bird) species abundance (e.g. Pickett and Siriwardena 2011, Renwick et al. 2012).

Following initial exploration of the data, and identifying presence of over dispersion, negative binomial models, with Log link function, were adopted for the modelling work presented here. In terms of negative binomial models, a random variable Y has a negative binomial distribution as $Y \sim NB(\mu, \kappa)$, with a parametrisation such that $E(Y)=\mu$ and $var(Y)=\mu + \kappa \mu^2$, where $\mu > 0$ and $\kappa > 0$ (e.g. ver Hoef and Boveng 2007). Our modelling approach is formalized as follows: $BBS_{ij} = f(X_i)$, where BBS_{ij} is a count of bird species in cell i of species j , and X_i is a vector of location-specific attributes related to land use change in cell i .

Model averaging is a group of methods based on information theory (IT) that combines predictions from a number of models which has the benefit of accounting for model uncertainty alongside parameter uncertainty. The information theory (IT) approach and model averaging has been increasingly used in the way biologist and ecologists analyse and make inferences from their data (Grueber et al 2011). It forms an alternative to traditional hypotheses testing for model selection (or sometimes referred to as 'frequentist' approach) (*ibid*). In model averaging multiple models are ranked by the Akaike Information Criterion and weighted, reflecting each model's relative standing by its AIC performance. In terms of our project, for each species, models with all possible combination of the variables were estimated and, for each model, the Akaike Information Criterion (AIC)⁸⁷ is calculated. The lowest AIC value across models shows the most parsimonious model that, overall, balances explanatory power of the model against the number of parameters. Based on

⁸⁷ Akaike information criterion, based on work of Hirotugu Akaike, is used as an estimator of relative quality of statistical models and is calculated as $IC = -2 \log \hat{L} + 2k$, where \hat{L} is the maximum value of likelihood function for a given model and k is a number of model's parameters.

the relative ranking in terms of the AIC, the parameter estimates from each model are weighted in calculating an overall, averaged, full model.

More formally (Burnham and Anderson 2002), we consider R models, where each model has θ as the predicted value of interest and each model i allowing an estimate of the parameter θ_i . Model averaging compute weighted estimate of the predicted value across the models, weighting the predictions by the Akaike weights (w_i), as follows:

$$\hat{\theta} = \sum_{i=1}^R w_i \hat{\theta}_i$$

, where $\hat{\theta}$ denotes model averaged estimate of θ from all models $i..R$ (Burnham and Anderson 2002).

This approach hence incorporates model selection uncertainty by relying on multiple models instead of one “best model”. This is useful where no prior expectation or theory would guide variable (& model) selection. Indeed, this is the case of our project where the available data used as explanatory variables (see previous section) and their relatively coarse resolution (4km²) have precluded a strict reliance on ecological theory for model selection. For example, spatial distribution of hedgerows and further classification of the wooded areas in the landscape would be useful (but currently unavailable) data that is likely to have an important role for bird species ecology. At the 2x2km level scale, the data available may work mostly as proxy for other ecologically important determinants of bird abundance in the landscape, instead or as well as directly driving it. For example, the type of crops grown in a particular area is likely to be associated with particular conditions (e.g. landscape features, altitude, etc.) that are having an effect on the abundance of a species. We therefore had no specific expectations about the performance of given explanatory variables in modelling the bird abundances. Indeed, one

objective of the research project was to test how well the land use data support modelling bird species at the national level and to link the findings back to ecological theory.

2.4.3 Assessment of model performance

We estimate our models on the most recent data to maximise inference about future changes. We employed a number of ways to assess the resulting (averaged) models' performance. We estimated the averaged models on 90% of the data from the late period. Further we use the data from the early period to investigate whether the relationships with individual variables change over time. We calculated Pearson and Spearman correlation coefficients between predicted values and actual values for three datasets:

- The data used for training the model (90% for each species considered from late period)
- 10% of the data from the late period set aside
- Full dataset for each species from the early period

We also calculated Root Mean Squared Error (RMSE) and Root Mean Squared Error as a percentage of the actual mean counts (RMSE_PER_MEAN) between predicted and actual BBS counts for each of these dataset.

While the Pearson correlation gives us how well the exact predicted numbers correlated with the actual BBS counts, the Spearman coefficients allows

2.4.4 Modelling protocol

As the initial number of explanatory variable was large, an approach was needed to reduce the variable list to provide computationally possible models (the computational demands rises exponentially for model averaging with increasing number of variables). A protocol was developed that was followed

in modelling the selected bird species. The protocol helped to create an ex ante decision tool that ensured that the same procedure is followed that lead to a coherent modelling approach across all species we focused on. The protocol consisted of the following steps for each species:

- 1) Removal of variables that were correlated (i.e. when $\text{corr}(a,b) > 0.85$)
- 2) Running GLM models with single variables from the remaining variable list
- 3) Removal of variables for which no significant impact (at a 0.1 level) was estimated on the abundance of birds in) above
- 4) Model averaging was then done from all possible combination (of main effects) of variable specification given the variable list following steps 1-3) above.

For each species the above steps were scripted in R, using packages 'MASS', 'glmmADMB', 'MuMIn'.

2.4.5 High performance computing facility

Combination of model averaging, high number of explanatory variables and number of species that this project focused on posed high requirements in terms of computing power. Due to limitation of researchers' own personal computer, an alternative approach was chosen: a High Performance Computing (HPC) cluster facility provided at the University of East Anglia⁸⁸. Since the HPC facility is consisting of 141 compute nodes, providing a total core count of 2,560, it is possible to garner significantly greater computing power than on personal computer.

⁸⁸ For further information see: <https://rscs.uea.ac.uk/new-high-performance-computing-cluster>

The HPC facility allowed us two things in particular. It provided advantage in having ability to run a multiple models in parallel which would otherwise likely be impossible on a personal computer. It also enabled running of models that required a long estimation time (e.g. over a week).

3 Results and discussion

Recalling our initial research interests and given the fact that we estimated and averaged models for 19 species, the results section presented here focuses on overall analysis of the modelling work done across all species. This analysis provides two perspectives, whereby first focuses on the performance measures for averaged models for each species and the second examines what seem to be the variables from the land use dataset that drive our bird abundance models.

3.1 Model performance measures

The Table 15 reports the performance measures of the averaged models for each species considered in our analysis, while Annex provides an example of three of the averaged models for individual species listed in Table 15. Recalling Section 0, we calculated Pearson and Spearman correlation coefficients - $\text{Cor}(\text{pearson}), r$, and $\text{Cor}(\text{spearman}), \rho$, respectively in the Table 15 - between predicted versus actual values and Root Mean Squared Error as a percentage of the actual mean counts (RMSE per mean). The three performance measures are calculated for the following data:

- 1) The “**Training data**” on which the models were estimated (i.e. 90% of late period data)
- 2) “**10% data set aside**” from the late period
- 3) “**Out of time frame (early period)**” (i.e. assessment across the time periods)

The correlation coefficients⁸⁹ give us an indication of how well the predicted values matches the actual counts. In turn the Root Mean Squared Error as a percentage of the actual mean counts (RMSE per mean) gives us indication how big the error was relative to mean value – for example value of 1.3 means that the mean error was approximately 130% size of the mean count.

The first observation from the Table 15 below is that, as expected, the models perform differently across the species and across the three sets of predicted vs actual data on bird abundances in the BBS squares. We will go through the performance measures by the datasets (i.e. top columns). First, in terms of the performance of the averaged models on the training data, the correlation coefficients are all positive and ranging from $r = 0.14$ (Kestrel) to $r = 0.406$ (Woodpigeon) in terms of Pearson Correlation coefficient, and $\rho = 0.150$ (Kestrel) to $\rho = 0.452$ (Woodpigeon) in terms of Spearman correlation coefficients. Crude average correlation values across all species are $r = 0.27$ and $\rho = 0.29$. In terms of the RMSE per mean, the range lies between 73% (Goldfinch) and 143% (Lapwing) of mean value in terms of the average error in prediction.

In terms of the model validation on the 10% of data set aside, we see for some species we get some higher, but also some significantly lower coefficients of correlations than in the training data. Here we see an average correlations coefficients of $r = 0.23$ and $\rho = 0.25$, expectedly lower than in terms of the data on which the models were estimated. In terms of individual species, we can see wider spread in both correlation measures in contrast to the training data. On one side we see relatively high values of $r = 0.55$

⁸⁹ While the Pearson correlation gives us indication of how well the exact predicted numbers correlated with the actual BBS counts, the Spearman (rank) coefficient allows us to assess how well the model predicted relative abundance across the dataset in terms of their rank (e.g. whether the maximum predicted counts are in the same cells as the actual maximum counts).

(Yellowhammer) and $\rho = 0.57$ (Yellowhammer) or $r = 0.48$ (Corn Bunting) and $\rho = 0.52$ (Woodpigeon). For some species, however, the models seem to be unable to predict well with $r = -0.15$ (Turtle dove) and $\rho = -0.03$ (Turtle dove) or similarly poorly performing measures for Tree Sparrow ($r = -0.03$, $\rho = 0.03$).

The results above suggest (expected) variation in the models' ability to predict bird abundance on basis of the land use data we have at our disposal. The difference across species might be due to number of reasons. As acknowledged previously in the chapter, it is likely that the variables that we use for species modelling are too coarse for capturing some of the elements in the landscape that drives specie populations. Indeed, it is likely that there are a number of landscape (and broader) elements that influence bird populations that we cannot model, such as hedges or fine-scale details of habitat (Rhodes et al. 2015). Further, it is likely that counts for some large, non-flocking species such as Kestrel will generally be low (e.g. between 0 and 2) whereas small species like yellowhammer could reach higher counts (e.g. >10) quite often which is likely to provide more variation to model and (probably) lower stochastic variability as a proportion of the mean count. This might have an effect on the maximum model fit.

That being said, averaged models for some species perform relatively similarly to the findings in the literature. For example, Siriwardena et al (2011) using similar modelling approach to this chapter, however considering different explanatory variables (focused on landscape, cropping and boundaries), document correlations between predicted and actual counts ranging between 0.3 and 0.6. This is the case for 6 out of 19 species we considered. While our study uses more coarse variables and hence can be expected to perform worse than in Siriwardena et al (2011) who focused on finer variables that might be driving bird abundances, some species models

reported here could be used for explaining variation related to land use change comparatively well.

We now turn to the assessment of model performance across the time periods (i.e. column "Out of time frame (early period)"). The model performance measures are significantly lower, with average correlation coefficients of $r = 0.05$ and $\rho = 0.06$ and variation between $r = 0.25$ (Greenfinch) and $\rho = 0.29$ (Greenfinch) to $r = -0.12$ (Lapwing) and $\rho = -0.20$ (Turtle Dove). The relatively poor performance of our models across the time periods might be due to number of reasons. First, this might be related to changes in the underpinning biological relationships that are not captured in our data related to, for example, changes in fertiliser use or other changes in micro-level agricultural practice that change the quality of individual habitat types as perceived by the birds. Similarly some of the changes in species population are likely to be driven by other factors than UK land use change, such as influences of weather or environmental variation in wintering areas outside farmland, or disease.

Table 15: Model performance measures for each species

	Training data			10% data set aside			Out of time frame (early period)			Number of data points
	Cor (pearson)	Cor (spearman)	RMSE per mean	Cor (pearson)	Cor (spearman)	RMSE per mean	Cor (pearson)	Cor (spearman)	RMSE per mean	
Corn Bunting	0.31	0.27	1.18	0.48	0.43	1.05	xx ⁹⁰	xx	xx	376
Goldfinch	0.21	0.19	0.73	0.28	0.27	0.69	0.19	0.19	0.86	3512
Greenfinch	0.29	0.34	0.79	0.23	0.26	0.75	0.25	0.29	0.82	3561
Grey Partridge	0.28	0.24	0.96	0.20	0.11	0.90	0.02	0.02	1.13	943
Jackdaw	0.15	0.17	1.06	0.19	0.20	1.09	-0.01	-0.02	1.28	3215
Kestrel	0.14	0.15	0.77	0.10	0.10	0.73	-0.03	-0.02	1.20	2095
Lapwing	0.21	0.22	1.43	0.13	0.20	1.34	-0.12	-0.11	0.96	1722
Linnet	0.28	0.29	1.07	0.19	0.20	1.05	-0.03	-0.01	0.94	2748
Reed Bunting	0.22	0.24	1.15	0.30	0.27	1.02	0.15	0.14	1.30	1484
Rook	0.23	0.25	1.26	0.21	0.17	1.30	0.22	0.20	1.37	2759
Skylark	0.38	0.44	0.91	0.30	0.37	0.93	-0.10	-0.11	1.01	3163
Starling	0.34	0.39	1.16	0.25	0.40	1.42	0.13	0.18	2.21	3220
Stock Dove	0.19	0.15	0.98	0.15	0.12	0.98	0.07	0.14	1.04	2024
Turtledove	0.22	0.22	1.16	-0.15	-0.03	7.77	-0.02	-0.20	1.25	564
Tree Sparrow	0.15	0.21	1.16	-0.03	0.03	1.38	-0.03	-0.03	0.98	494
Whitethroat	0.37	0.42	0.87	0.39	0.48	0.80	0.06	0.12	0.94	3071
Woodpigeon	0.44	0.49	0.77	0.44	0.52	0.78	0.15	0.17	3.66	3936
Yellowhammer	0.41	0.45	0.76	0.55	0.57	0.71	-0.02	0.02	0.98	2459
Yellow Wagtail	0.34	0.27	0.99	0.17	0.06	1.12	xx ⁹¹	xx	xx	483
Average	0.27	0.29	1.01	0.23	0.25	1.36	0.05	0.06	1.29	

⁹⁰ Our models were unable to predict out of time period for this species.

⁹¹ Our models were unable to predict out of time period for this species.

3.2 Relative variable importance

A second set of aggregate results, as presented in Table 16 offers insight into the models in terms of relative importance of each variable in the modelling of all species considered in our analysis. Based on the number of times that the individual models that are being averaged contain certain variable and the Akaike weights per any such model, it is possible to derive the relative importance of each variable used in the final averaged model for each species (see Appendix). The relative variable importance varies from 0, where variable is not present at all in any of the averaged model, to 1 which means that the variable was present in all models that were averaged for a given species, and was essential to explain part of the variation. The table below aggregates these relative variable importance measures across all species and then for Generalist and Specialist species separately, and divides the resulting numbers by number of species in each group. The resulting coefficient that varies also from 0 to 1 gives an indication of how important given variable was in modelling farmland bird species. Or in another words, how often the variable was important in predicting variation for the groups of species, given that we treat all species as having an equal weight.

A first observation from Table 16 is that the most relatively important variables for modelling farmland bird species are ones that do not describe aspects of farmland. This is likely to be due to a negative preference for non-farmland habitat is commonly shared across the species, despite their ecological particularities such as preference for particular cropping in the landscape. Indeed, both coniferous and deciduous woodland have in all but one estimates negative impact on abundance of bird species and urban habitat has predominantly negative impact too which is in line with findings in other literature (e.g. Siriwardena et al. 2011). A second

observation is that the relative variable importance differs between farmland specialist and generalists for some variables. For example generalist species might be less sensitive to the specific crops and land-uses within farmland, but more dependent on gross landscape features like hedges and areas of gardens. This might explain the differences in other farmland, permanent grass, potatoes and spring barley results.

Table 16: Variables' relative importance for modelling farmland bird species

Relative importance of a variable for modelling across species (1 = important for all; 0 = important for none)			
Variable	All	Generalists	Specialists
Coniferous woodland	0.7	0.7	0.7
Urban and other developed land	0.7	0.6	0.7
Deciduous woodland	0.7	0.5	0.7
Maize (Scotland 2004 is within 'othcrps')	0.5	0.5	0.6
permanent grassland i.e. >5 yrs	0.5	0.4	0.6
Other farmland e.g. roads, buildings, yards, ponds and, where appropriate, seaside	0.5	0.7	0.4
Wheat	0.5	0.5	0.4
Potatoes	0.4	0.2	0.6
Spring barley (England and Scotland only)	0.4	0.2	0.6
Rough grazing	0.4	0.4	0.5
Freshwater	0.4	0.4	0.3
Winter barley (England and Scotland only)	0.3	0.2	0.4
Other cereals (includes oats and other cereals for combining)	0.3	0.4	0.3
Temporary grassland i.e. <5 years	0.3	0.4	0.3
Semi-natural grass or mountains, moors and heaths where NOT used for farming	0.3	0.3	0.3
Sugarbeet	0.3	0.3	0.3
Total horticulture	0.2	0.2	0.2
Other crops and bare fallow (includes oilseed rape for Wales)	0.2	0.2	0.2
Sea and estuary	0.2	0.3	0.1
Ocean (area that is not covered by land is given 'ocean' by default)	0.2	0.2	0.2

3.3 Use of models for spatial prediction & integration with economic values

The initial aim of integrating the ecological and economic models from this and third chapter requires further work, going beyond the scope of this thesis, and is foreseen for the post-PhD period. However, this section will provide a short outline of how such integration is expected to be implemented.

First decision will need to be made about selection of the bird species models. Given that the relative model performance varies greatly for our species, careful selection of which models to use for the integration is needed or, alternatively, full set of species could be used with clear caveats about the limitations of the predictions. Indeed, we will need to weight the relative model performance against representativeness of the species for the purpose of modelling broader trends in biodiversity. This will require further consultation with (BTO) experts.

The integration of the two analyses is based on the fact that both the economic and ecological analyses are based on a common land use database and are linked through the biodiversity attribute (recall attribute selection in Chapter 3). The integration is foreseen as follows. Firstly, we will define relevant policy scenario that would be reflected in changes in variables used in the species modelling, but which would also be relevant as a valid scenario that was used for valuing changes in land use in Chapter 3. This decision might be informed by the analysis of relative variables importance for the modelling of the species. The chosen scenario is most likely to be associated with changes to land use that lead to a decrease in intensification of agricultural landscapes in order to retain validity for the Farmland bird species. When such a scenario is selected, a case study region will be chosen

where this scenario could be implemented, most likely being in the areas of high intensity agriculture and on a region-to-national scale, as both modelling projects were estimated on a national level. The selected bird models would then be used to predict a baseline bird abundances and Farmland bird Index will be calculated for each square of the case study region. Next, the selected scenario, in terms of land use variables would be applied in each cell with the changes in the variables leading to changes in species abundances and associated Farmland bird index. This gives us the distribution of potential ecological benefits from an implementation of the given scenario. Finally, we use our spatially explicit value function based on research presented in Chapter 3 and apply the function on the scenario and input from the ecological model. Recalling that one of the key variables in the value function (based on WTP results from Chapter 3) is changes in the bird population in the area, this is where the link between the two research projects occurs. The relative changes from the baseline to the resulting scenario will give us changes in both species abundance and species richness which, in turn, can be used as a variable in the value transfer function. For each cell, we will then have the estimated value that the given scenario generates for the population in the area. Given that the value function takes into account population density and relative distance to each site to estimate the value generated by a given intervention, it is most likely that the distribution of the economic benefits will spatially differ from distribution of the ecological benefits. Indeed this is our aim - to demonstrate that it is most likely that interventions that try to maximise ecological benefits will be differently spatially distributed to interventions that aims to maximise economic value from the same type of interventions.

4 Conclusions

This project had two objectives. First it aimed to test how the land use data that we have at our disposal, and that have been used in the Chapter 3, can be used for model abundance of farmland bird species in Great Britain and which are the most influential variables. We provided evidence that the ability of these models are limited for predicting abundances of some species, however at the same time with number of species models providing similar fit to established literature in the ecological fields. The fact that the models perform worse than we initially expected, while unfortunate, however provides an important result. The land use variables at the scale we used seem to be too crude to explain well the variation in bird species population. This is in line with the previous finding in the literature and underscores the importance of finer scale variables, such as hedges or other fine level habitat features, for modelling bird species (and more broadly biodiversity) at a national scale. Further, our analysis reveals that from the given set of land use variables, the non-farmland variables are expectedly most common drivers of the drivers of the variation in species abundances. However, going beyond these (woodland and urban) variables we also show that farmland land use variables, such as maize, permanent grassland, potatoes or wheat, play an important role for farmland bird species, but this role varies for generalist and specialist farmland bird species.

Our second objective of this project was to derive predictive models that could be used in our future research to integrated ecological and economic models in order to provide an interdisciplinary analysis of land use change in Great Britain. While requiring further work, in terms of careful selection of the species that could be used for this work, as well as

the definition of the relevant scenario that could be used for the integrated analysis, we feel confident that our models could be used this way. This work is foreseen in the next year.

Foremost, however, this project provided an invaluable experience in ecological modelling. Given the crucial role of interdisciplinary cooperation in addressing some of the environmental crises we face today, an understanding of ecological modelling for an environmental economist is a tremendously useful experience for any future research for the researcher involved.

5 Appendix

5.1 Averaged Model - Corn Bunting

Model-averaged coefficients:
(full average)

	Estimate	Std. Error	Adjusted SE	z value	Pr(> z)
(Intercept)	0.6732505	0.6373647	0.6377808	1.056	0.29114
GRSNFRM_10	0.0181241	0.0184353	0.0184585	0.982	0.32616
POTS_10	0.0602287	0.0513123	0.0513809	1.172	0.24112
SBARLEY_10	0.0345582	0.0242244	0.0242572	1.425	0.15426
SUGARBEET_10	-0.0879015	0.0312136	0.0312906	2.809	0.00497 **
WHEAT_10	0.0194321	0.0118443	0.0118564	1.639	0.10122
WBARLEY_10	-0.0172964	0.0234134	0.0234457	0.738	0.46068
DEC_07	-0.0139182	0.0160320	0.0160531	0.867	0.38594
HORT_10	0.0054377	0.0117478	0.0117677	0.462	0.64402
TEMPG_10	-0.0235394	0.0299181	0.0299485	0.786	0.43187
URBAN_07	-0.0063617	0.0080050	0.0080122	0.794	0.42720
CON_07	-0.0052758	0.0148218	0.0148535	0.355	0.72245
OTHRPS_10	0.0024190	0.0113845	0.0114163	0.212	0.83220
PERMG_10	0.0004701	0.0054233	0.0054328	0.087	0.93105
TOSR_10	-0.0017551	0.0102252	0.0102525	0.171	0.86407

(conditional average)

	Estimate	Std. Error	Adjusted SE	z value	Pr(> z)
(Intercept)	0.673250	0.637365	0.637781	1.056	0.29114
GRSNFRM_10	0.027424	0.016100	0.016140	1.699	0.08931 .
POTS_10	0.082717	0.041903	0.042019	1.969	0.04900 *
SBARLEY_10	0.042962	0.019196	0.019247	2.232	0.02561 *
SUGARBEET_10	-0.089356	0.029333	0.029416	3.038	0.00238 **
WHEAT_10	0.022788	0.009383	0.009401	2.424	0.01535 *
WBARLEY_10	-0.034010	0.022572	0.022638	1.502	0.13301
DEC_07	-0.023236	0.014580	0.014619	1.589	0.11197
HORT_10	0.014811	0.015397	0.015439	0.959	0.33738
TEMPG_10	-0.042887	0.028302	0.028361	1.512	0.13048
URBAN_07	-0.011244	0.007640	0.007653	1.469	0.14179
CON_07	-0.016193	0.022305	0.022369	0.724	0.46912
OTHRPS_10	0.008631	0.020220	0.020284	0.426	0.67045
PERMG_10	0.001436	0.009404	0.009421	0.152	0.87889
TOSR_10	-0.006213	0.018505	0.018558	0.335	0.73778

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Relative variable importance:

	SUGARBEET_10	WHEAT_10	SBARLEY_10	POTS_10	GRSNFRM_10	DEC_07
Importance:	0.98	0.85	0.80	0.73	0.66	0.60
N containing models:	4858	4249	3442	2901	2866	3180
	URBAN_07	TEMPG_10	WBARLEY_10	HORT_10	PERMG_10	CON_07
Importance:	0.57	0.55	0.51	0.37	0.33	0.33
N containing models:	3657	3114	2771	2581	2683	2666
	TOSR_10	OTHRPS_10				
Importance:	0.28	0.28				
N containing models:	2461	2533				

5.2 Averaged Model - Yellowhammer

Model-averaged coefficients:
(full average)

	Estimate	Std. Error	Adjusted SE	z value	Pr(> z)
(Intercept)	2.0696903	0.2990117	0.2990447	6.921	< 2e-16 ***
CON_07	-0.0126550	0.0037569	0.0037578	3.368	0.000758 ***
DEC_07	-0.0203092	0.0036343	0.0036353	5.587	< 2e-16 ***
FWATER_07	-0.0358260	0.0128550	0.0128607	2.786	0.005341 **
GRSNFRM_10	-0.0102561	0.0057045	0.0057059	1.797	0.072261 .
MAIZE_10	-0.0249813	0.0118385	0.0118426	2.109	0.034905 *
OTHCER_10	0.0342733	0.0149485	0.0149542	2.292	0.021912 *
PERMG_10	-0.0084841	0.0035474	0.0035479	2.391	0.016789 *
POTS_10	-0.0479144	0.0115776	0.0115837	4.136	3.53e-05 ***
RGRAZ_10	-0.0160244	0.0044750	0.0044766	3.580	0.000344 ***
SBARLEY_10	0.0236887	0.0061030	0.0061050	3.880	0.000104 ***
URBAN_07	-0.0147849	0.0031893	0.0031899	4.635	3.60e-06 ***
HORT_10	0.0045690	0.0055612	0.0055621	0.821	0.411386
WBARLEY_10	0.0045260	0.0068890	0.0068905	0.657	0.511283
WHEAT_10	0.0026161	0.0047986	0.0047991	0.545	0.585672
OTHFRM_10	-0.0005739	0.0033470	0.0033483	0.171	0.863913
OTHCRPS_10	-0.0008534	0.0042602	0.0042620	0.200	0.841307

(conditional average)

	Estimate	Std. Error	Adjusted SE	z value	Pr(> z)
(Intercept)	2.069690	0.299012	0.299045	6.921	< 2e-16 ***
CON_07	-0.012773	0.003570	0.003571	3.577	0.000347 ***
DEC_07	-0.020309	0.003634	0.003635	5.587	< 2e-16 ***
FWATER_07	-0.036487	0.012008	0.012014	3.037	0.002389 **
GRSNFRM_10	-0.011668	0.004533	0.004535	2.573	0.010081 *
MAIZE_10	-0.026885	0.009982	0.009987	2.692	0.007103 **
OTHCER_10	0.036152	0.012954	0.012961	2.789	0.005282 **
PERMG_10	-0.009155	0.002727	0.002728	3.356	0.000792 ***
POTS_10	-0.047918	0.011570	0.011576	4.139	3.48e-05 ***
RGRAZ_10	-0.016121	0.004312	0.004313	3.737	0.000186 ***
SBARLEY_10	0.023689	0.006103	0.006105	3.880	0.000104 ***
URBAN_07	-0.014785	0.003189	0.003190	4.635	3.60e-06 ***
HORT_10	0.007753	0.005272	0.005273	1.470	0.141507
WBARLEY_10	0.009541	0.007225	0.007228	1.320	0.186849
WHEAT_10	0.005978	0.005703	0.005704	1.048	0.294611
OTHFRM_10	-0.001849	0.005808	0.005810	0.318	0.750346
OTHCRPS_10	-0.002914	0.007481	0.007485	0.389	0.697048

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Relative variable importance:

	DEC_07	SBARLEY_10	URBAN_07	POTS_10	RGRAZ_10	CON_07	FWATER_07	OTHCER_10
MAIZE_10								
Importance:	1.00	1.00	1.00	1.00	0.99	0.99	0.98	0.95
N containing models:	329	329	329	328	297	290	264	270
PERMG_10	GRSNFRM_10	HORT_10	WBARLEY_10	WHEAT_10	OTHFRM_10	OTHCRPS_10		
0.93	0.88	0.59	0.47	0.44	0.31	0.29		
236	201	242	207	235	156	140		

5.3 Averaged Model - Kestrel

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Model-averaged coefficients:
(full average)
      Estimate Std. Error Adjusted SE z value Pr(>|z|)
(Intercept) -0.1297853  0.0544987   0.0545244  2.380 0.017298 *
GRSNFRM_10  0.0097609  0.0029397   0.0029413  3.319 0.000905 ***
OTHFRM_10   0.0131090  0.0076724   0.0076748  1.708 0.087627 .
WHEAT_10    0.0024310  0.0026940   0.0026947  0.902 0.366982 .
WBARLEY_10  0.0062236  0.0089999   0.0090027  0.691 0.489373
URBAN_07   -0.0004429  0.0009821   0.0009825  0.451 0.652154

(conditional average)
      Estimate Std. Error Adjusted SE z value Pr(>|z|)
(Intercept) -0.129785  0.054499   0.054524  2.380 0.017298 *
GRSNFRM_10  0.009848  0.002804   0.002806  3.510 0.000449 ***
OTHFRM_10   0.015138  0.006104   0.006108  2.479 0.013189 *
WHEAT_10    0.003994  0.002384   0.002385  1.675 0.094004 .
WBARLEY_10  0.012680  0.009119   0.009125  1.390 0.164645
URBAN_07   -0.001208  0.001306   0.001307  0.924 0.355341
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Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Relative variable importance:
      GRSNFRM_10 OTHFRM_10 WHEAT_10 WBARLEY_10 URBAN_07
Importance:      0.99      0.87      0.61      0.49      0.37
N containing models: 16      16      15      14      16

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CONCLUDING REMARKS

Summary of the thesis

This thesis contributes to our understanding of the spatial dimension of biodiversity related values in the context of changing environment. It provides three empirical analyses, two that employ discrete choice modelling to examine how space impacts preferences for environmental change, and one that focuses on modelling the biodiversity impacts of land use change. The major contributions of the thesis are: 1) development of a novel methodology for choice experiments that incorporates space in the survey design, experimental design and present choice situations on individualised maps; 2) application of this methodology to test how adding such maps to commonly used Tabular format impacts on preferences and welfare values for land use change; 3) provision of evidence that political boundaries, such as state and country borders, have an impact on preferences for environmental change; and 4) development of predicting models that allow evaluating land use change impacts on farmland bird species; this includes assessment of model performance and variables importance for future integration of the models with economic analyses.

Starting with the literature review of economic valuation of preferences for ecosystem related goods and services, Chapter 1 makes a case for the use of valuation for improving decisions and halting the degradation of the biosphere. An overview of the main concepts in environmental valuation and methods is provided and the Chapter concludes with a discussion of the future of valuation research.

Chapter 2 and 3 provide empirical application of discrete choice modelling that focus on examining one particular issue raised in Chapter 1

– the impacts of space on preferences for environmental interventions. Both research applications use maps to portray location of environmental change to respondents and analyse the impact of the portrayed spatial attributes on respondents' choices. The choice experiment, presented in Chapter 2 examines preferences for sea grass restoration in South East Australia. This application of choice modelling looks at how state borders and specific city areas impacts on respondents' choices for interventions. It presents choice on maps of the region with explicit state borders and shows that, controlling for distance, respondents prefer sites locating in their states and strongly dislike sites that are located in close proximity to cities in the region. The chapter calls for more research into these issues as should these findings be more common, this might change policy implications of valuation studies that do not consider such political/specific attributes in the study design.

Building on experience gained from the work presented in Chapter 2, Chapter 3 proposes and implements a novel approach to bringing space into the design and presentation of stated preference choice experiments (CE). The study investigate preferences concerning agri-environmental interventions across Great Britain. CE scenarios are presented through individually generated maps, tailored to each respondent's home location. Each choice situation is underpinned by spatially optimal experimental designs relevant to the individual's spatial context and current British land uses. To the best of our knowledge, this represents the first case of a CE that integrates space into the survey design, presentation of options and data analysis. We test the effect of our map format for presenting spatial attributes against a commonly applied tabular approach, finding that the former yields both significantly different and more robust preference estimates than the latter. This effect is most pronounced for the spatial attributes, but seems to be also present for attributes related to use

motivations. Similarly to Chapter 2, evidence of preferences for sites located in one's country is provided.

Chapter 4 presents an application of generalised linear modelling to model bird abundance in Great Britain. Employing a vast dataset of land use covariates to explain bird abundances from Breeding Bird Survey dataset provided by the British Trust of Ornithology, the research allows understanding of how land use change impacts on biodiversity in biophysical terms. It provides an examination of how well could the data be used for modelling farmland bird species and which variables are the main drivers of modelled bird abundances. Using the same land use variable database, the Chapter discusses how the models could be used in parallel with the work presented in Chapter 3 to analyse simultaneously economic and biodiversity impacts of land use change.

PhD lessons learnt

The work presented in the thesis is underpinned by a number of research skills and knowledge I have gained over the course of past 4 years of my PhD that were applied in the context of assessment of environmental change. This included Discrete Choice modelling; stated preference survey design (experimental design using Ngene software and development of novel survey functionality with professional programmers); econometric WTP and preference analyses using Logit models and simulation; Generalised Linear Models and Model Averaging; use of High Performance Computing Facility at the University of East Anglia; use of Geographic Information Systems analysis of land use data; and data manipulation and statistical analysis implemented in R and Stata. The work on Chapter 4 deserves a special mention in learning new area of research, ecology, that I was not familiar with, but understanding of which I feel is crucial for any

environmental economist interested in valuation of ecosystem related goods and services.

More specifically, the PhD research presented in this thesis (and the supervisors involved) taught me a number of invaluable lessons about Choice experiments which constituted the majority of research presented in the thesis. A few such lessons stand out. First, based on the experience from the project described in Chapter 2, I would not again design a CE survey without a status quo (i.e. opt out) option, aside from situation when such set up would be significantly justified. It is in my opinion difficult to advocate for the robustness of a CE results where respondents are forced to choose from only options with payment options and cannot opt out from the payment. Indeed if I were to repeat the survey from Chapter 2, I would include a status quo option. Secondly, based on the work done on the project presented in Chapter 3, I have learnt the invaluable role of piloting the CE survey. The number of pilot waves that we have implemented made us realise a number of issues that needed to be improved which led to, what I hope, is a significantly better CE design and data analysis. I have since made multiple recommendations to other researchers to pilot the survey as much as possible and will surely do so in my future work. Thirdly, I have realised that I need to and will pay more attention to and spend more time on empirically testing the assumed distribution of the coefficients estimated within the mixed logit modelling framework. This is especially important for assumed distribution for the price coefficient. However, in the case of Chapter 2 (which was discussed in the viva) I have decided to use normally distributed price coefficient due to two reasons: 1) I was not interested in deriving WTP values which under normal distribution might be problematic; 2) I wanted to capture potential protesters by understanding the proportion of respondents exhibiting positive price coefficient. Finally,

I have learnt that the time and attention is paid to experimental and research design front is more than well spent. I was lucky to see the fruit of these efforts in the project presented in Chapter 3 that formed the major part of my PhD research and which has, so far, received positive feedback from other researchers.

The PhD research has also taught me how to write scientific publications and present my research in the leading conferences in the field. The First chapter of the thesis has been published (Badura et al. 2016) and Chapter 3 has been submitted to a leading scientific journal in the field. The research also greatly benefited from number of presentations and comments from participants' and debaters' points made at Envecon 2017 in London, international Choice Modelling Conference 2017 in Cape Town, EAERE 2017 in Athens, Bioecon 2017 in Tilburg, as well as great points made at the PhD colloquium in Birmingham 2013. In fact, the presentations were crucial in understanding of my research and what is novel and useful about it.

While there is always (infinite) room for improvement, looking at the past four years I feel that I have learnt a lot. These set of skills I hope will be put in goods years in the research years to come.

Way forward

Three general areas of further research arise from the work presented in this thesis. Further research into designing valuation surveys that incorporate space is needed. While majority of the stated preference literature incorporate spatial considerations in the data analysis (see Chapter 3 for an overview), a limited progress has been done on how we collect the data, how we sample it and how we present hypothetical

situations with regards to spatial considerations. I hope that the methodology presented in Chapter 3 is a step in this direction.

Relatedly, as demonstrated in this thesis, further exploration of different modes of spatial information presentation is crucial. Recent literature (esp. Johnston et al. 2016) has shown that, in line with our results, spatial information might have impact on both spatial, but also non-spatial attributes in stated preference studies. This has potential implications for welfare estimates and, in turn, policy advice. Understanding these effects is crucial for making valuation useful, relevant and reliable for policy use.

Further, exploration of the relevance of political dimension of space in preferences for environmental interventions is required. Number of environmental policies, particularly in the EU, are implemented on a transboundary scale. Given the findings presented in this thesis and the limited literature documenting presence of 'patriotic premiums' (Dallimer et al. 2013) or community association effects (Chapter 2) and similar preferences (Rogers and Burton 2017), further exploration of when such considerations is of relevance to stated preference surveys would be useful. In particular, exploration of different scales and definition of what we call political boundary might be interesting (e.g. counties, particular cities' areas etc.).

Personally, I plan to focus my attention on two areas of research. First, I will focus on the integration of models from Chapter 3 and 4. While this will require further work and expert consultation in selecting the correct bird species models to model trends in biodiversity, the resulting interdisciplinary analysis aims to illuminate the contrast between what is optimal economically and what is optimal ecologically. I hope that this analysis, together with additional collected survey data not reported in this

thesis, might be useful for informing the current debate in the UK regarding its future agri-environmental policies, following its decision to leave the EU. Further, I plan to expand the analysis presented in this thesis. We collected a very rich dataset that offers multiple avenues for analysis, including preference stability order effects (e.g. Day et al. 2012), attribute non-attendance (e.g. Hussen Alemu et al. 2013; Scarpa et al. 2009, 2013; Glenk et al. 2015), as well as analysing the difference between two scenarios that were presented in the survey and incorporating substitution effects in the data analysis (e.g. Schaafsma et al. 2013, Schaafsma and Brouwer 2013). Similarly, it would be interesting to try to incorporate distance in data analysis in a more advanced manner (e.g. Andrews et al. 2017; Czajkowski et al. 2016).

In long term I hope to be able to expand the work related to spatial choice experiments in terms of more complex survey design, as well as testing the different modes of how spatial information is presented to respondents. In particular, I am interested in developing more realistic stated preference surveys that incorporate real-world complexity in its design, for example, by better portrayal of spatial context through more advanced mapping interface than presented in this thesis. Similarly, I hope to further explore the issues surrounding incorporation of spatial attributes (both physical and political) in the experimental design. The goal would be to develop consistent methodology and interface that other researchers could easily use for spatially relevant stated preference surveys.

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